Montana
Board of Environmental Review
P. O. Box 200901 • Helena, MT 59620-0901 • (406) 444-2544 • Website: www.deq.state.mt.us

AGENDA

FRIDAY, JULY 25, 2014
METCALF BUILDING, ROOM 111
1520 EAST SIXTH AVENUE, HELENA, MONTANA

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NOTE: *Individual agenda items are not assigned specific times. For public notice purposes, the meeting will begin no earlier than the time specified; however, the Board might not address the specific agenda items in the order they are scheduled. The Board will make reasonable accommodations for persons with disabilities who wish to participate in this meeting. Please contact the Board Secretary by telephone at (406) 444-6701 or by e-mail at jwittenberg@mt.gov no later than 4 days prior to the meeting to advise her of the nature of the accommodation you need.*

9:00 A.M.

I. ADMINISTRATIVE ITEMS

A. REVIEW AND APPROVE MINUTES

1. The Board will vote on adopting the May 30, 2014, meeting minutes.

II. BRIEFING ITEMS

A. CONTESTED CASE UPDATE

1. Enforcement cases assigned to the Hearing Examiner

   a. **In the matter of violations of the Public Water Supply Laws by Trailer Terrace Mobile Park, LLC, Dennis Deschamps and Dennis Rasmussen at the Trailer Terrace, PWSID No. MT0000025, Great Falls, Cascade County, BER 2012-11 PWS.** A Fifth Order Granting Extension was issued on May 16, 2014, giving the parties through August 1, 2014, to settle the matter or file a joint proposed prehearing schedule.

   b. **In the matter of violations of the Sanitation in Subdivisions Act and Public Water Supply Laws by Roger Emery at the Sunrise Motel, Sidney, Richland County, BER 2013-06 SUB.** On June 4, 2014, the attorney for DEQ filed Department of Environmental Quality's Motion for Summary Judgment and Brief in Support.

2. Non-enforcement cases assigned to the Hearings Examiner

   a. **In the matter of the notice of appeal and request for hearing by Yellowstone Energy Limited Partnership (YELP) regarding issuance of MPDES Permit NO. MT0030180 for YELP's facility in Billings, MT, BER 2014-01 WQ.** On April 29, 2014, the attorney for YELP filed Unopposed Motion to Stay Proceedings, and on May 6, 2014, the Interim Hearings Examiner issued Order Granting Motion to Stay Proceedings, requiring a status report no later than August 1, 2014.

3. Contested Cases not assigned to a Hearing Examiner

   a. **In the matter of the notice of appeal and request for hearing by Western Energy Company (WECO) regarding its MPDES Permit No. MT0023965 issued for WECO's Rosebud Mine in Colstrip, BER 2012-12 WQ.** On April 9, 2014, the hearings examiner issued an Order Granting the Joint Unopposed Motion for Partial Remand of Permit to Department of Environmental Quality and for Suspension of Proceedings. On May 14, 2014, DEQ filed a Status Report regarding the matter. A modified permit will be made available for public comment on or before June 9, 2014.

   b. **In the matter of the notice of appeal for hearing by Montana Environmental Information Center regarding DEQ's approval of coal mine permit No. C1993017 issued to Signal Peak Energy, LLC, for Bull Mountain Mine No. 1 in Roundup, MT, BER 2013-07 SM.** The following documents have been filed in this matter since the March 21 Board meeting, and resolution of two motions is pending.

      - 4/11/14 – Appellant Montana Environmental Information Center's Motion for Summary Judgment and Brief in Support
      - 5/30/14 – Department of Environmental Quality Response Brief in Opposition to [MEIC's] Motion for Summary Judgment
      - 5/30/14 – Signal Peak Energy, LLC's Cross-Motion for Summary Judgment
      - 5/30/14 – Signal Peak Energy, LLC's Combined Response to MEIC's Motion for Summary Judgment and Brief in Support of Cross-Motion for Summary Judgment

III. ACTION ITEMS

A. INITIATION OF RULEMAKING

DEQ will propose that the Board initiate rulemaking to:

1. In the matter of the amendment of ARM 17.30.1101, 17.30.1102, 17.30.1105, 17.30.1106, 17.30.1107, 17.30.1111, 17.30.1341 and 17.30.1342 pertaining to Montana pollutant discharge elimination system (MPDES) permits, purpose and scope, definitions, permit requirements, exclusions, designation procedures: small municipal separate storm sewer systems (MS4s), application procedures, permit requirements, general permits and conditions applicable to all permits and repeal of ARM 17.30.1110, 17.30.1115 and 17.30.1117 application procedures: general, notice of intent procedures, and transfer of permit coverage.

B. REPEAL, AMENDMENT, OR ADOPTION OF FINAL RULES

1. Title 17, Chapter 36, Sub-Chapter 9, On-Site Subsurface Wastewater Treatment Systems by updating definitions and Table 1 Setback Distances to provide consistency between the subdivision rules in Title 17, Chapter 36 and DEQ Circular 4, 2013, edition. Title 17, Chapter 38, Sub-chapter 101(4)(d) to adopt by reference the proposed changes to Title 17, Chapter 36 for Subdivisions; specifically ARM 17.36.320 through 17.36.323, 17.36.325 and to remove the adoption by reference to ARM 17.36.327. Title 17 Chapter 38, Sub-chapters 106(2) (a), (d), and (e) to provide fee structure consistency for review of public water supply and sewage systems that correspond to the proposed changes to

Department Circular DEQ-1, the adopted changes to Department Circular DEQ-4, 2013 edition, and new proposed Department Circular DEQ-10. Title 17, Chapter 38, Sub-chapter 106(2) to add a provision (f) for the review of public water supply systems that corresponds to proposed Department Circular DEQ-16.

2. The Department recommends adoption of the proposed amendments to Title 17, Chapter 38, Sub-Chapter 1, Public Water and Sewer Plans, Cross Connections, and Drilling Water Wells, updating Department Circulars DEQ-1 and DEQ-3 related to public drinking water design standards to the 2014 edition, clarification of the requirements for the submission of plans and specifications, clarification of the engineering review fee tables, updating expedited checklists, adding new Department Circular DEQ-10 describing the use of spring's as a public source and new Department Circular DEQ-16 describing the use of cisterns for non-community public water systems, and amendments to Title 17, Chapter 36, Sub-chapter 3, Subdivisions/On-site Subsurface Wastewater Treatment, to adopt the 2014 editions of DEQ-1 and DEQ-3.

3. In the matter of new Department Circular DEQ-12A "Montana base numeric nutrient standards" for surface waters, and the amendment of rules in MAR Notice No. 17-356 to incorporate the base numeric nutrient standards into the water-quality standards. The Department is requesting the new circular and the rule amendments be adopted by the Board.

C. FINAL ACTION ON CONTESTED CASES

1. **In the matter of appeal and request for hearing by Missoula County and the Clark Fork Coalition regarding DEQ's issuance of MPDES Permit No. MT0000035 issued to M2Green Redevelopment's site in Frenchtown, MT, BER 2014-02/03 WQ.** On June 30, 2014, the Board received Stipulation for Dismissal of Administrative Appeal signed by the parties. An order to dismiss the appeal will be presented for the Chair to sign.

2. **In the matter of violations of the Clean Air Act of Montana by Myrstol Logging, Inc., Clyde Park, Park County, MT, BER 2014-04 AQ**. The Board received the appeal on June 2, 2014, and on June 11, a First Prehearing Order was issued. On June 25, 2014, the Board received an informal notice from the appellant that he was withdrawing his appeal. On July 14, 2014, the Board received a formal Withdrawal of Request for Contested Case from the appellant. An order dismissing the case will be presented for the Chair's signature.

IV. GENERAL PUBLIC COMMENT

Under this item, members of the public may comment on any public matter within the jurisdiction of the Board that is not otherwise on the agenda of the meeting. Individual contested case proceedings are not public matters on which the public may comment.

V. ADJOURNMENT

Montana
Board of Environmental Review
P. O. Box 200901 • Helena, MT 59620-0901 • (406) 444-2544

MINUTES

May 30, 2014

Call to Order

The Board of Environmental Review's regularly scheduled meeting was called to order by Madam Chair Shropshire at 9:02 a.m., on Friday, May 30, 2014, in Room 111 of the Metcalf Building, 1520 East Sixth Avenue, Helena, Montana.

Attendance

Board Members Present: Madam Chair Shropshire, Larry Mires, Marietta Canty, and Joan Miles

Board Members Present via Phone: Heidi Kaiser, Chris Tweeten, and Joe Russell

Board Attorney Present: Ben Reed, Attorney General's Office, Department of Justice

Board Attorney Present via Phone: Katherine Orr, Attorney General's Office, Department of Justice

Board Secretary Present: Joyce Wittenberg

Court Reporter Present: Laurie Crutcher, Crutcher Court Reporting

Department Personnel Present: Tom Livers (Deputy Director); John North, Norman Mullen, Kurt Moser – Legal; Jon Dilliard, Barb Kingery – Public Water Supply & Subdivisions Bureau; Jon Kenning, Kari Smith, Paul Skubinna, Water Protection Bureau; David Klemp, Eric Merchant, Charles Homer, Rebecca Harbage, Carolyn Arrington, Liz Ulrich, Hoby Rash, Annette Williams – Air Resources Management Bureau; John Arrigo – Enforcement Division; George Mathieus, Planning Division; Eric Urban, Mike Suplee – Water Quality Planning Bureau; Jeff Blend, Energy Pollution Prevention Bureau

Interested Persons Present *(Disclaimer: Names are spelled as best they can be read from the official sign-in sheet.)*: Mark Lambrecht – Treasure State Resource Industry Association; Jim Parker – PPL Montana

At the request of Chairman Shropshire, Mr. Livers took roll call of Board members present. Ms. Miles was not yet present.

Mr. Livers explained that the Board would address some of the agenda items out of order, taking the administrative items first, followed by the contested cases update. He said the Board would then initiate a motion to shift the cases currently assigned to Ms. Orr to Mr. Reed, before finishing with contested cases, followed by the remainder of the agenda.

I.A.1 Review and approve March 21, 2014, Board meeting minutes.

Chairman Shropshire asked if anyone had comments on the draft minutes. There were none.

Mr. Mires MOVED to approve the minutes as written. Ms. Canty SECONDED the motion. The motion CARRIED with a 6-0 vote.

(Ms. Miles now present.)

Ms. Orr introduced Mr. Reed as her replacement. Mr. Reed provided information regarding his background.

II.A.1.a In the matter of violations of the Public Water Supply Laws by Trailer Terrace Mobile Park, LLC, Dennis Deschamps and Dennis Rasmussen at the Trailer Terrace, PWSID No. MT0000025, Great Falls, Cascade County, BER 2012-11 PWS. *(No discussion took place regarding this matter.)*

II.A.1.b In the matter of violations of the Sanitation in Subdivision Act and Public Water Supply Laws by Roger Emery at the Sunrise Motel, Sidney, Richland County, BER 2013-06 SUB. *(No discussion took place regarding this matter.)*

II.A.2.a In the matter of the notice of appeal and request for hearing by Western Energy Company (WECO) regarding its MPDES Permit No. MT0023965 issued for WECO's Rosebud Mine in Colstrip, BER 2012-12 WQ. *(No discussion took place regarding this matter.)*

II.A.2.b In the matter of the notice of appeal for hearing by Montana Environmental Information Center regarding DEQ's approval of coal mine permit No. C1993017 issued to Signal Peak Energy, LLC, for Bull Mountain Mine No. 1 in Roundup, MT, BER 2013-07 SM.

Ms. Orr said there are two pending motions in this matter and that she and Mr. Reed will be working on those until her June 30 departure.

Chairman Shropshire thanked Ms. Orr for her years of service to the Board, as did Mr. Livers, Mr. Russell, Mr. Tweeten, Ms. Kaiser, Ms. Miles, Ms. Canty, Mr. Mires, and Mr. Arrigo.

Chairman Shropshire called for a motion to appoint Mr. Reed as the Board's interim hearing officer for all requests for contested case hearings that the Board receives in the future, and as replacement hearings officer for all contested cases for which Ms. Orr is currently acting as permanent or interim hearing officer. Ms. Miles so MOVED. Ms. Kaiser SECONDED the motion. The motion CARRIED with a 7-0 vote.

(Items from this point forward were taken out of order, in the order listed herein.)

III.C.1 In the matter of the notice of appeal and request for hearing by Yellowstone Energy Limited Partnership (YELP) regarding issuance of MPDES Permit NO. MT0030180 for YELP's facility in Billings, MT, BER 2014-01 WQ.

Ms. Orr described the case and said after the prehearing order was issued the parties filed a motion to stay all proceedings in an attempt to reach agreement regarding what the elements of the permit should be. She said a status report regarding progress of the negotiations is due August 1.

Chairman Shropshire called for a motion to assign Mr. Reed as the permanent hearings examiner. Mr. Mires so MOVED. Ms. Canty SECONDED the motion. The motion CARRIED with a 7-0 vote.

III.C.2 In the matter of appeal and request for hearing by Missoula County regarding DEQ's issuance of MPDES Permit No. MT0000035 issued to M2Green Redevelopment's site in Frenchtown, MT, BER 2014-02 WQ. *(see next item, III.C.3)*

III.C.3 In the matter of the notice of appeal and request for hearing by the Clark Fork Coalition regarding DEQ's issuance of MPDES Permit NO. MT0000035 issued to M2Green Redevelopment's site in Frenchtown, MT, BER 2014-03 WQ.

Ms. Orr said the department moved to consolidate this and the previous case (BER 2014-02 WQ), and that she granted the motion on May 28. These two cases will now be under one caption with both case numbers. She provided information regarding the details of the appeals.

Chairman Shropshire called for a motion to assign Mr. Reed as the permanent hearings examiner for this matter. Ms. Miles so MOVED. Ms. Kaiser SECONDED the motion. The motion CARRIED with a 7-0 vote.

(Ms. Orr concluded her participation.)

II.B.1 Numeric Nutrient Standards Briefing

Mr. Livers said that there had previously been multiple briefings on this and that this rulemaking is scheduled for final action at the July meeting. He noted that if the Board adopts the rulemaking at the July 25 meeting, the notice has to be filed with the Secretary of State on Monday, July 28, in order to meet the six-month deadline.

Mr. Mathieus reminded the Board that this effort has been ongoing since 2000 and provided details of the process. He noted that nonseverability was an issue, so the workgroup developed three provisions in the rule to deal with the issue, and that a fourth provision is being considered. He said stakeholders in the Flathead requested more time to review it and the department concluded that is a reasonable request, therefore the department recommends the Board not adopt the Flathead Lake numeric standards at this time.

III.A.2 In the matter of the proposal to initiate rulemaking to amend ARM 17.8.501 Definitions and 17.8.504 Air Quality Permit Fees, to adjust air quality permit application fees to more closely reflect the cost of processing a permit application, clarify relevant definitions, and make other housekeeping amendments.

Mr. Homer provided a brief history of the rulemaking, saying the last fee adjustment was in 2009. He said the department is now looking at the equity of the fee system, both the operating fees and the permit application fees. He said the department is requesting the Board initiate rulemaking, appoint a presiding officer, and schedule a hearing.

Mr. Livers and Mr. Homer responded to questions from Board members.

Chairman Shropshire asked if any members of the public wished to comment on the proposed rulemaking. Mr. Lambrecht spoke in support of the rule.

Chairman Shropshire called for a motion to initiate the rulemaking and to appoint Mr. Reed as the hearing officer. Ms. Canty so MOVED. Mr. Mires SECONDED the motion. The motion CARRIED with a 7-0 vote.

III.A.3 In the matter of the proposal to initiate rulemaking to amend ARM 17.8.818 Review of Major Stationary Sources and Major Modifications – Source Applicability and Exemptions, and 17.8.820 Source Impact Analysis, to reflect changes to Major New Source Review Prevention

Mr. Merchant proposed that the Board initiate rulemaking to remove and modify certain major air quality permitting provisions to be at least as stringent as federal requirements. He provided details of the changes being requested. He responded to questions from the Board.

Chairman Shropshire asked if any member of the public wanted to comment on the proposed rulemaking. No one responded.

Chairman Shropshire called for a motion to initiate the rulemaking and to appoint Mr. Reed as the hearing officer. Ms. Miles so MOVED. Ms. Canty SECONDED the motion. The motion CARRIED with a 7-0 vote.

III.B.1 In the matter of proposed final adoption of amended ARM 17.8.102 incorporating the air quality rules adopted in the 2013 edition of the Code of Federal Regulations and current updates to state statutes and regulations that are incorporated by reference in the rules.

Mr. Merchant said the department requests that the Board adopt the current editions of the federal and state statutes and regulations that are incorporated by reference in the Administrative Rules of Montana. He said the Board initiated rulemaking January 21, 2014, and a public hearing took place March 20, 2014, at which time the department commented on the proposed rule to remove redundant language.

Chairman Shropshire asked if any member of the public wanted to comment on the proposed rule. No one responded.

Chairman Shropshire called for a motion to adopt the Presiding Officer Report, the House Bill 311 and 521 analyses, and the responses to comments, and to amend 17.8.102 as provided in the notice. Mr. Mires so MOVED. Ms. Miles SECONDED the motion. The motion CARRIED with a 7-0 vote.

IV. General Public Comment

Chairman Shropshire asked if any member of the audience would like to speak to any matters before the Board. No one responded.

V. Adjournment

Chairman Shropshire called for a motion to adjourn. Ms. Canty so MOVED. Ms. Miles SECONDED the motion. The motion CARRIED 7-0.

The meeting adjourned at 10:32 a.m.

Board of Environmental Review May 30, 2014, minutes approved:

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ROBIN SHROPSHIRE
CHAIRMAN
BOARD OF ENVIRONMENTAL REVIEW

____

DATE

BOARD OF ENVIRONMENTAL REVIEW
AGENDA ITEM
EXECUTIVE SUMMARY FOR PROPOSED RULEMAKING

AGENDA ITEM # III.A.1.

AGENDA ITEM SUMMARY - The Department is requesting that the Board initiate rulemaking to amend and repeal certain rules governing the issuance of discharge permits under the Montana Pollutant Discharge Elimination System (MPDES) program. The Department is requesting these actions in order to maintain compliance with federal regulations governing discharge permits issued under the National Pollutant Discharge Elimination System (NPDES) program.

LIST OF AFFECTED RULES - This rulemaking would amend ARM 17.30.1101, 17.30.1102, 17.30.1105, 17.30.1106, 17.30.1107, 17.30.1111, 17.30.1341 and 17.30.1342, and repeal of 17.30.1110, 17.30.1115, and 17.30.1117.

AFFECTED PARTIES SUMMARY - This rulemaking would affect owners or operators of new or existing facilities that discharge storm water or other wastewater into state surface water, and are regulated under the Montana Pollutant Discharge Elimination System (MPDES) program, and persons or facilities who wish to obtain a discharge permit.

SCOPE OF PROPOSED PROCEEDING - The Department requests that the Board initiate rulemaking on the above-referenced rules and conduct a public hearing to take comment on the proposed rules.

BACKGROUND – The Department is delegated authority to issue discharge permits in Montana under the National Pollutant Discharge Elimination System pursuant to Section 402 of the federal Clean Water Act, 33 USC 1342. This rulemaking is necessary to maintain compliance and equivalency with federal regulations governing state programs that are delegated to implement the federal permitting program in accordance 40 CFR 123.25. Equivalent federal regulations governing the issuance of NPDES permits are found in 40 CFR 122. The federal rules applicable to this rulemaking are: storm water regulations found in 40 CFR 122.26, general permit requirements found in 40 CFR 122.28, and standard conditions for all NPDES permits found in 40 CFR 122.41.

The proposed amendments to storm water rules in ARM 17.30.1101, 17.30.1102, 17.30.1105, 17.30.1106, 17.30.1107, and 17.30.1111 will maintain consistency with equivalent federal rules and State MPDES rules in ARM 17.30.1301 - 1363. The proposed amendments to ARM 17.30.1101 will clarify the relationship between rules in Subchapter 11 and Subchapters 12 and 13 which collectively constitute the MPDES program. Amendments to ARM 17.30.1102 are necessary to amend some definitions, add new definitions, and delete terms which are obsolete or are no longer used in this subchapter. The amendments to 17.30.1105, 17.30.1107, and 17.30.1111 are necessary to clarify that the requirements for issuing general permits are given in ARM 17.30.1341 and to incorporate changes in the definitions of storm water discharges associated with industrial activity and small construction activities and associated cross references in the rules. Amendments to ARM 17.30.1105 are proposed to clarify that only the point source discharges identified in this rule are required to obtain permit coverage.

Amendments to ARM 17.30.1107 are necessary to incorporate a federal exemption for certain mining, and oil and gas activities from the requirement to obtain coverage under the MPDES program for discharges composed entirely of storm water, and to meet other requirements specified in the rule.

The amendments to ARM 17.30.1341 are proposed to clarify the requirements for issuing general permits including the area, scope of coverage, sources, content, and administration for all general permits, including storm water. The amendments also remove certain conditions and requirements that are not found in the equivalent federal rule. These amendments are necessary to maintain consistency with the federal rule governing the issuance of general permits under 40 CFR 122.28.

The amendments to ARM 17.30.1342 are necessary to make the rule consistent with the equivalent federal requirements in 40 CFR 122.41 and the Montana Water Quality Act. These amendments include updated penalty provisions for civil, criminal and administrative violations that may result from noncompliance with permit requirements.

Repeal of ARM 17.30.1110 is proposed because this rule describes application requirements and requirements for submittal of notices of intent which are now described in ARM 17.30.1322 and 17.30.1341 respectively and referenced in the proposed amendments to ARM 17.30.1105, 17.30.1107, and 17.30.1111. Repeal of ARM 17.30.1115 is proposed because procedures for submittal of notices of intent for all general permits are the same and are described in ARM 17.30.1341. Repeal of ARM 17.30.1117 is proposed to eliminate duplication and conflict between ARM 17.30.1341, 17.30.1360 and 17.30.1365, which set forth regulations governing transfer of all MPDES permit including general permits.

Hearing Information: The Department recommends that the Board appoint a hearing examiner, or presiding officer, and conduct a public hearing to take comment on the proposed amendments and repeal of the rules as described herein.

Board Options: The Board may:

1. Initiate rulemaking and issue the attached Notice of Public Hearing on Proposed Amendment and Repeal;
2. Modify the Notice and initiate rulemaking; or
3. Determine that amendment of the rules is not appropriate and deny the Department's request to initiate rulemaking

DEQ Recommendation: The Department recommends that the Board initiate rulemaking and appoint a presiding officer to conduct a public hearing as described in the attached rule notice.

Enclosures:

1. Draft Notice of Public Hearing on Proposed Amendment and Repeal

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.30.1101, 17.30.1102, 17.30.1105, 17.30.1106, 17.30.1107, 17.30.1111, 17.30.1341 and 17.30.1342 pertaining to Montana pollutant discharge elimination system (MPDES) permits, purpose and scope, definitions, permit requirements, exclusions, designation procedures: small municipal separate storm sewer systems (MS4s), application procedures, permit requirements, general permits and conditions applicable to all permits and repeal of ARM 17.30.1110, 17.30.1115 and 17.30.1117 application procedures: general, notice of intent procedures, and transfer of permit coverage pertaining to storm water discharges	NOTICE OF PUBLIC HEARING ON PROPOSED AMENDMENT AND REPEAL (WATER QUALITY)

TO: All Concerned Persons

1. On ___________, 2014, at __:__ __ p.m., the Board of Environmental Review will hold a public hearing [in/at address], Montana, to consider the proposed amendment and repeal of the above-stated rules.

2. The board will make reasonable accommodations for persons with disabilities who wish to participate in this public hearing or need an alternative accessible format of this notice. If you require an accommodation, contact Elois Johnson, Paralegal, no later than 5:00 p.m., _____________, 2014, to advise us of the nature of the accommodation that you need. Please contact Elois Johnson at Department of Environmental Quality, P.O. Box 200901, Helena, Montana 59620-0901; phone (406) 444-2630; fax (406) 444-4386; or e-mail ejohnson@mt.gov.

3. The rules proposed to be amended provide as follows, stricken matter interlined, new matter underlined:

17.30.1101 PURPOSE AND SCOPE (1) This subchapter is intended to be applied together with ARM Title 17, chapter 30, subchapters 12 and 13 to establish a system for regulating the discharge~~s~~ of ~~potential~~ pollutants from point sources ~~discharges of storm waters into surface~~ to state waters. This subchapter and subchapters 12 and13 of ARM Title 17, chapter 30~~, which regulate storm water discharges through Montana pollutant discharge elimination system (MPDES) general permits, permit authorizations, and notices of intent,~~ are intended to be compatible with the national pollutant discharge elimination system (NPDES) as established by the United States ~~e~~Environmental ~~p~~Protection ~~a~~Agency pursuant to section 402 of the federal Clean Water Act (CWA), 33 USC 1251, et seq. ~~Except as expressly modified in this subchapter, all requirements in ARM Title 17, chapter 30, subchapters 12 and 13 remain effective pertaining to point source discharges of storm water.~~

~~(2) The rules in this subchapter pertain to point source discharges of storm water that do not contain routine process wastewater and that do not contain non-storm water discharges except for the potential non-storm water discharges from MS4s that are listed in ARM 17.30.1111(6)(c)(iii). ARM Title 17, chapter 30, subchapter 13 contains additional requirements pertaining to point source discharges of storm water that routinely contain process wastewater or non-storm water discharges (other than the potential non-storm water discharges for MS4s listed in ARM 17.30.1111(6)(c)(iii)) that are regulated using an individual MPDES permit.~~

AUTH: 75-5-201, 75-5-401, MCA
IMP: 75-5-401, MCA

REASON: For the reasons set forth below, the board is proposing to amend (1) to clarify a person's duty to apply for an MPDES permit for any discharge of pollutants to state waters, unless the discharge is excluded under ARM 17.30.1310 or 17.30.1106. The term "discharge of pollutant" is defined in ARM 17.30.1102 and 17.30.1304 and means the addition of any pollutant or combination of pollutants to state waters from any point source. The board is also proposing to remove the term "potential" in reference to pollutants because the discharge of "potential pollutants" is not regulated under state or federal permit requirements. The board is proposing to remove the term "surface water" and replace it with "state water," as defined in 75-5-103, MCA. The board is also proposing to remove text from (1) stating that the requirements in subchapter 11 modify the requirements in subchapters 12 and 13. This change is necessary because subchapters 12 and 13 apply to all MPDES permits and are not modified by subchapter 11.

The board is also proposing to remove (2) to provide consistency between storm water discharge permit requirements and ARM 17.30.1322 (pertaining to all MPDES permit application requirements) and to clarify that storm water discharge permits are subject to the provisions of subchapter 13, which pertain to all MPDES permits. These amendments to (2) are necessary to provide storm water discharge permit requirements that are consistent with the applicable federal regulations and board rules pertaining to all discharge permits.

17.30.1102 DEFINITIONS (1) through (4) remain the same.

~~(5) "Final stabilization" means the time at which all soil-disturbing activities at a site have been completed and a vegetative cover has been established with a density of at least 70% of the pre-disturbance levels, or equivalent permanent, physical erosion reduction methods have been employed. Final stabilization using vegetation must be accomplished using seeding mixtures or forbs, grasses, and shrubs that are adapted to the conditions of the site. Establishment of a vegetative cover capable of providing erosion control equivalent to pre-existing conditions at the~~

~~site will be considered final stabilization.~~

(6) through (21) remain the same, but are renumbered (5) through (20).

(21) "Significant materials" includes, but is not limited to:

(a) raw materials;

(b) fuels;

(c) materials such as solvents, detergents, and plastic pellets;

(d) finished materials such as metallic products;

(e) raw materials used in food processing or production;

(f) substances designated as hazardous under section 101(14) of the federal Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) 42 U.S.C. 9601(14);

(g) any chemical the facility is required to report pursuant to the reporting requirements under section 313 of the federal Emergency Planning and Community Right to Know Act (EPCRA) created under the Superfund Amendments and Reauthorization Act (SARA)also known as SARA Title III, 42 U.S.C. 11001 - 11050;

(h) fertilizers;

(i) pesticides; and

(j) waste products such as ashes, slag and sludge that have the potential to be released with storm water discharges.

(22) through (27) remain the same.

~~(28) "Storm water discharge associated with construction activity" means a discharge of storm water from construction activities including clearing, grading, and excavation that result in the disturbance of equal to or greater than one acre of total land area. For purposes of these rules, construction activities include clearing, grading, excavation, stockpiling earth materials, and other placement or removal of earth material performed during construction projects. Construction activity includes the disturbance of less than one acre of total land area that is a part of a larger common plan of development or sale if the larger common plan will ultimately disturb one acre or more.~~

~~(a) Regardless of the acreage of disturbance resulting from a construction activity, this definition includes any other discharges from construction activity designated by the department pursuant to ARM 17.30.1105(1)(f).~~

~~(b) For construction activities that result in disturbance of less than five acres of total land area, the acreage of disturbance does not include routine maintenance that is performed to maintain the original line and grade, hydraulic capacity, or original purpose of the facility.~~

~~(c) For construction activities that result in disturbance of five acres or more of total land area, this definition includes those requirements and clarifications stated in (29)(a), (b), (d) and (e).~~

~~(29)~~ (28) "Storm water discharge associated with industrial activity" means a discharge from any conveyance that is used for collecting and conveying storm water and that is directly related to manufacturing, processing or raw materials storage areas at an industrial plant.

(a) remains the same.

~~(b) For the categories of industries identified in (e)(ix) of this definition, the term includes only storm water discharges from all the areas (except access roads and rail lines) that are listed in the previous sentence where material handling~~

~~equipment or activities, raw materials, intermediate products, final products, waste materials, by-products, or industrial machinery are exposed to storm water.~~

(c) remains the same, but is renumbered (b).

~~(d)~~ (c) Industrial facilities<u>,</u> ~~(~~including industrial facilities that are federally, state, or municipally owned or operated that meet the description of the facilities listed in ~~(e)~~ <u>(d)</u>(i) through ~~(ix) and (30))~~ <u>(x),</u> include those facilities designated under the provisions of ARM 17.30.1105(1)~~(f)~~ <u>(d)</u>.

(e) remains the same, but is renumbered (d).

(i) facilities subject to storm water effluent limitations guidelines, new source performance standards, or toxic pollutant effluent standards under 40 CFR subchapter N <u>(Effluent Guidelines and Standards Part 405-471),</u> ~~(~~except facilities with toxic pollutant effluent standards that are exempted under category ~~(e)~~ <u>(d)</u>~~(ix)~~ <u>(x)</u> of this definition~~)~~;

(ii) remains the same.

<u>(iii) facilities classified as standard industrial classifications 10 through 14 (mineral industry) including active and inactive mining operations, except for areas of coal mining operations no longer meeting the definition of a reclamation area under 40 CFR 434.11(l) because the performance bond issued to the facility by the appropriate SMCRA authority has been released, or areas of non-coal mining operations that have been released from applicable state or federal reclamation requirements after December 17, 1990; oil and gas exploration, production, processing, or treatment operations; and transmission facilities that discharge storm water by contact with, or that come into contact with, any overburden, raw material, intermediate material, finished products, byproducts or waste products located on the site of such operations. Inactive mining operations are mining sites that are not being actively mined, but which have an identifiable owner/operator. Inactive mining operations do not include sites where mining claims are being maintained prior to disturbances associated with the extraction, beneficiation, or processing of mined materials nor sites where minimal activities are undertaken for the sole purpose of maintaining a mining claim;</u>

(iii) remains the same, but is renumbered (iv)

~~(iv)~~ <u>(v)</u> landfills, land application sites, and open dumps that receive or have received any industrial wastes (waste that is received from any of the facilities described under this definition~~, or under the definitions of "storm water discharge associated with mining and oil and gas activities," and "storm water discharge associated with construction activity" that will result in construction-related disturbance of five acres or more of total land area~~) including those that are subject to regulation under subtitle D of RCRA;

(v) through (ix) remain the same, but are renumbered (vi) through (x).

<u>(xi) construction activities including clearing, grading, and excavating except operations that result in the disturbance of less than five acres of total land area. Construction activity also includes the disturbance of less than five acres of total land area that is part of a larger common plan of development or sale if the larger plan will ultimately disturb five acres or more.</u>

~~(30) "Storm water discharge associated with mining and oil and gas activity" means the same as the definition for "storm water discharges associated with industrial activity" except that the term pertains only to discharges from facilities~~

~~classified as standard industrial classifications 10 through 14 (mineral industry) that discharge storm water contaminated by contact with or that has come into contact with, any overburden, raw material, intermediate products, finished products, byproducts, or waste products located on the site of such operations. Such facilities include active and inactive mining operations (except for areas of coal mining operations no longer meeting the definition of a reclamation area under 40 CFR 434.11(1) because the performance bond issued to the facility by the appropriate SMCRA authority has been released, and except for areas of non-coal mining operations that have been released from applicable state or federal reclamation requirements after December 17, 1990); and oil and gas exploration, production, processing, or treatment operations; and transmission facilities. "Inactive mining operations" are mining sites that are not being actively mined but that have an identifiable owner/operator, but do not include sites where mining claims are being maintained prior to disturbances associated with the extraction, beneficiation, or processing of mined materials, nor sites where minimal activities are undertaken for the sole purpose of maintaining a mining claim.~~

(29) "Storm water discharge associated with small construction activity" means:

(a) the discharge of storm water from construction activities including clearing, grading, and excavating that result in land disturbance of equal to or greater than one acre and less than five acres. Small construction activity also includes the disturbance of less than one acre of total land area that is part of a larger common plan of development or sale if the larger common plan will ultimately disturb equal to or greater than one and less than five acres. Small construction activity does not include routine maintenance that is performed to maintain the original line and grade, hydraulic capacity, or original purpose of the facility. The department may waive the otherwise applicable requirements in a general permit for a storm water discharge from construction activities that disturb less than five acres where the conditions given in ARM 17.30.1105(3) are satisfied; and

(b) any other construction activity designated by the department under ARM 17.30.1105, or by the EPA regional administrator, based on the potential of the discharge to contribute to a violation of a water quality standard or to contribute significant pollutants to state surface water.

~~(31) "Storm water pollution prevention plan (SWPPP)" means a document developed to help identify sources of pollution potentially affecting the quality of storm water discharges associated with a facility or activity, and to ensure implementation of measures to minimize and control pollutants in storm water discharges associated with a facility or activity. The department determines specific requirements and information to be included in a SWPPP based on the type and characteristics of a facility or activity, and on the respective MPDES permit requirements.~~

~~(32) "Surface waters" means any waters on the earth's surface including, but not limited to, streams, lakes, ponds, and reservoirs, and irrigation and drainage systems discharging directly into a stream, lake, pond, reservoir, or other surface water. Water bodies used solely for treating, transporting, or impounding pollutants shall not be considered surface water.~~

(33) through (35) remain the same, but are renumbered (30) through (32).

AUTH: 75-5-201, 75-5-401, MCA
IMP: 75-5-401, MCA

REASON: The board is proposing to amend definitions found in ARM 17.30.1102 to add several new definitions found in 40 CFR 122.26(b), the federal rule defining terms used in the federal storm water regulations, and to remove several definitions that are no longer used in this subchapter. The board is also proposing to modify several definitions to ensure consistency with federal storm water regulations found at 40 CFR 122.26. The board's specific reasons for amending these definitions follow.

The board is proposing to remove the definition of final stabilization as this term does not appear in federal storm water regulations found at 40 CFR 122.26. The General Permit for Storm Water Discharges Associated with Construction Activity (General Permit No. MTR1000000) covers storm water discharges associated with construction activity from initiation of construction-related ground disturbance to "final stabilization" of that disturbance. The term "final stabilization" is defined in Part 5 of General Permit No. MTR1000000, to describe the point at which coverage under General Permit No. MTR1000000 may be terminated, but the term does not appear in subchapter 11.

The board is proposing to add a definition of "significant materials" to define materials that may be discharged with storm water and have the potential to impact human health or the environment. The proposed definition at (21) is consistent with the federal definition of "significant materials" at 40 CFR 122.26(b)(12).

The board is proposing to remove the definition of storm water discharges associated with construction activity at (28) and replace it with two new definitions. The first of these definitions is at proposed (28)(d)(xi) and would place construction activities that disturb more than five acres of total land area under the definition of storm water discharges associated with industrial activity. The second definition pertains to storm water discharges associated with small construction activity at proposed (29), which would include the disturbance of less than five acres of total land area. These amendments are necessary to ensure consistency with the federal definitions of storm water discharge associated with construction activities at 40 CFR 122.26(b).

The board is proposing the amendments at current (29) (proposed to be renumbered (28)) to define the term "storm water discharges associated with industrial activity" to include mining and oil and gas activities, currently defined in (30), and construction activities greater than five acres. The board is also proposing to make other minor editorial changes and to renumber the definitions in this rule. This amendment is necessary to provide consistency with the federal definition of industrial activities at 40 CFR 122.26(b)(14). The board is proposing to delete what is currently numbered (29)(b) as the text is not part of the federal definition of industrial activities in 40 CFR 122.26(b)(14). The board is proposing to amend current (29)(e) (proposed to be renumbered (28)(d)) to make minor editorial changes and to correct internal references. The board is proposing to amend current (29)(e)(i) (proposed to be renumbered (28)(d)(i)) to change the reference to subparts of this definition. The board is also proposing to amend current (29)(d)(iv) (proposed to be renumbered (28)(d)(v)) to remove language that is no longer necessary due the inclusion of mining and oil and gas activities that were defined in (30) and are now defined in proposed (28) as amended.

The board is proposing to remove the definition of "storm water discharges associated with mining and oil and gas activities," currently at (30), and to include this category of industrial discharge in proposed (28), along with other similar industrial activities. This amendment will provide consistency between the state and federal definition of storm water discharge associated with industrial activity.

The board is proposing a new definition in (29) to define "storm water discharges associated with small construction activity" consistent with 40 CFR 122.26(b)(15). This amendment is necessary to maintain consistency with federal regulations defining storm water discharges and different application and permitting requirements for small construction in ARM Title 17, chapter 30, subchapter 13.

The board is proposing to delete the definition of "storm water pollution prevention plan" (SWPPP), currently in (31), as this term is no longer used in this subchapter and does not appear in federal storm water regulations found at 40 CFR 122.26. The General Permit for Storm Water Discharges Associated with Construction Activity (General Permit No. MTR1000000) covers storm water discharges associated with construction activity. In order to achieve compliance with the conditions of General Permit No. MTR1000000, the permittee is required to develop a Storm Water Pollution Prevention Plan (SWPPP). The term "SWPPP" is defined in Part 5 of General Permit No. MTR1000000 to describe a document developed to identify sources of pollution potentially affecting the quality of storm water discharges associated with a facility or activity and to ensure implementation of measures to minimize and control pollutants in storm water discharges associated with a facility or activity. The department determines specific requirements and information to be included in a SWPPP based on the type and characteristics of a facility or activity and on the respective MPDES permit requirements.

The board is proposing to remove the definition of "surface waters," currently at (32), because this definition is unnecessary. Surface waters are included in the definition of state water at 75-5-103, MCA. The provisions of this subchapter apply to discharges of storm water to state water unless excluded under ARM 17.30.1106. The board is proposing to renumber current (33) through (35) as (30) through (32). The proposed amendments to these definitions are necessary to ensure consistency and equivalency with the federal definitions found in 40 CFR 122.2 and 40 CFR 122.26(b) and with the definitions found in the board rules at ARM 17.30.1304 and 17.30.1202.

17.30.1105 PERMIT REQUIREMENT (1) ~~Any person who discharges or proposes to discharge storm water from a point source must obtain coverage under an MPDES general permit or another MPDES permit for discharges~~ On or after October 1, 1994, operators must obtain an MPDES permit for discharges composed entirely of storm water that are not required by (4) to obtain a permit only if:

(a) the discharge is associated with small construction activity as defined in ARM 17.30.1102;

~~(b) associated with industrial activity;~~

~~(c) associated with mining and oil and gas activity;~~

~~(d)~~ (b) the discharge is from a small municipal separate storm sewer systems

~~that are~~ as ~~identified~~ defined in ARM 17.30.1102 or as designated pursuant to ARM 17.30.1107;

~~(e)~~ (c) for which the department determines that storm water controls are needed based on wasteload allocations that are part of TMDLs that address the pollutants of concern; ~~and~~ or

(f) remains the same, but is renumbered (d).

~~(2) For point source discharges of storm water identified in (1)(a) through (f) that are routinely composed entirely of storm water, authorization under an MPDES general permit must be obtained pursuant to this subchapter, unless the discharge is covered under an individual MPDES permit that is issued pursuant to ARM Title 17, chapter 30, subchapter 13 to the same owner or operator for other point source discharges.~~

~~(3) For point source discharges of storm water identified in (1)(a) through (f) that are not routinely composed of storm water, and that routinely discharge pollutants, coverage under an individual MPDES storm water permit or under an MPDES general permit must be obtained pursuant to ARM Title 17, chapter 30, subchapter 13.~~

(4) remains the same, but is renumbered (2).

~~(5)~~ (3) The department may waive the permit requirements in this subchapter for a storm water discharge associated with construction activity that disturbs less than five acres of total land area if either of the following two conditions exist:

(a) the value of the rainfall erosivity factor ("R" in the revised universal soil loss equation) is less than five during the period of construction activity. ~~The period of construction activity extends through to final stabilization.~~ The rainfall erosivity factor must be determined using a state-approved method. The owner or operator must certify to the department that the construction activity will take place only during a period when the value of the rainfall erosivity factor is less than five. If unforeseeable conditions occur that are outside of the control of the waiver applicant, and which will extend the construction activity beyond the dates initially applied for, the owner or operator shall reapply for the waiver or obtain authorization under the general permit for storm water discharges associated with construction activity. The waiver reapplication or notice of intent must be submitted within two business days after the unforeseeable condition becomes known; or

(b) remains the same.

~~(6)~~ (4) Prior to October 1, 1994, discharges composed entirely of storm water are not required to obtain an MPDES permit except for:

(a) discharges with respect to which an ~~individual~~ MPDES permit has been issued prior to February 4, 1987; ~~and~~

(b) ~~discharges listed in (1)(a), (b), (c), and (f), except that, for discharges listed in (1)(a), this requirement applies only to storm water discharges associated with construction activity that will result in construction-related disturbance of five acres or more of total land area~~ a discharge associated with an industrial activity; or

(c) a discharge that the department or EPA regional administrator determines contributes to a violation of a water quality standard or is a significant contributor of pollutants to state waters.

~~(7)~~ (5) For storm water discharges designated by the department under (1) ~~(e)~~ (c) and ~~(f)~~ (d) or (4)(c), the owner or operator shall apply for a permit within 180 days of receipt of the department's notice of designation, unless the department grants a later date.

~~(8)~~ (6) Except as provided in ~~(9)~~ (7), ~~if not authorized under a storm water general permit, a permit application or notice of intent must be submitted to the department for storm water discharges existing as of~~ any storm water discharge associated with an industrial activity as defined in ARM 17.30.1102 that is not covered under an existing MPDES permit must submit a permit application to the department by October 1, 1992~~, that are associated with:~~.

~~(a) industrial activity;~~

~~(b) mining and oil and gas activity; and~~

~~(c) construction activity that will result in construction-related disturbances of five acres or more of total land area and for which storm water discharges are not authorized by a storm water general permit.~~

~~(9)~~ (7) The permit requirements in this subchapter are effective beginning March 10, 2003, ~~F~~for ~~discharges identified in (8)(a) through (c)~~ ~~that are not authorized by a general or individual MPDES permit,~~ ~~and which~~ ~~are~~ any storm water discharge associated with industrial activity from a facility~~, other than an airport, powerplant, or uncontrolled sanitary landfill,~~ that is owned or operated by a municipality with a population of under 100,000~~,~~ that is not authorized by a general or individual permit, other than an airport, powerplant, or uncontrolled sanitary landfill~~, the permit requirements in this subchapter are effective beginning March 10, 2003~~.

(10) and (11) remain the same, but are renumbered (8) and (9).

AUTH: 75-5-201, 75-5-401, MCA
IMP: 75-5-401, MCA

REASON: The board is proposing to amend the permit requirements for discharges composed entirely of storm water in ARM 17.30.1105 to maintain consistency with the equivalent federal regulations set forth in 40 CFR 122.26(a) and the permit requirements set forth in ARM Title 17, chapter 30, subchapter 13. The proposed amendments to the definitions of "storm water discharges associated with small construction" and "storm water discharges associated with industrial activity" in ARM 17.30.1102 allow for streamlining and better alignment of this subchapter with the applicable federal regulations and board rules in ARM Title 17, chapter 30, subchapter 13. Under 40 CFR 123.25, the permit requirements in ARM 17.30.1105 are a required element of a delegated state's NPDES permit program. The board is also proposing minor changes to wording, punctuation, formatting, and renumbering the provisions in this rule. The board's specific reasons for proposing these amendments follow.

The board is proposing to amend (1) to maintain consistency with the equivalent federal rules at 40 CFR 122.26(a)(9). This federal rule requires permit coverage for certain discharges that are composed entirely of storm water after October 1, 1994. Permit coverage under this rule is limited to: discharges associated with small construction activity; discharges from designated small municipal separate storm sewer systems; storm water discharges which require a waste load allocation; and discharges which contribute to a violation of water quality standards or are required by current (6) (proposed to be renumbered (4)) to obtain permit coverage. Unless specifically required by (1) or proposed (4) of this rule, discharges composed entirely of storm water are not subject to permit requirements under this subchapter. The board is proposing to amend (1)(a) to reflect the proposed change in the definition of small construction activity in ARM 17.30.1102, which would include construction activities that are greater than one acre and less than five acres. The board is proposing to delete industrial facilities from (1) since they are addressed in (4). The board is also proposing to delete mining and oil and gas activities from (1) since these activities are proposed to be included in the definition of industrial activity in ARM 17.30.1102. The board is also proposing to amend (1)(d) to make minor wording changes and to renumber it (1)(b).

The board is proposing to delete (2) which requires a storm water discharger to obtain coverage under a general permit unless the discharge is covered under an individual permit because this requirement is not found in equivalent federal rules set forth in 40 CFR 122.21, 122.26, and 122.28. When it qualifies for general permit coverage, a facility may obtain coverage under that general permit unless directed by the department to obtain coverage under an individual permit. A facility may also request to be excluded from coverage under the general permit, in accordance with 40 CFR 122.28(b)(3) or ARM 17.30.1341 and obtain an individual permit.

The board is proposing to delete (3) because the board rules in ARM Title 17, chapter 30, subchapter 13, have been updated to include storm water discharges as well as discharges of process wastewater and other types of wastewater making the requirements set forth in (3) unnecessary. A facility that discharges both storm water and other forms of wastewater must submit the applicable information as specified in a general permit issued under ARM 17.30.1341 or individual permit under ARM 17.30.1322.

The board is proposing to amend (6) and renumber it (4). The proposed amendments reflects changes to the definition of "discharges from small construction activity and industrial activity" proposed in ARM 17.30.1102 and are necessary to maintain consistency with 40 CFR 122.26(a)(1). In order to maintain consistency with the equivalent federal rule at 40 CFR 122.26(a)(i), which does not restrict this permitting requirement to discharges for which individual permits were issued prior to February 4, 1987, the board is proposing to delete the word "individual." The board is also proposing to amend (b) to maintain consistency with 40 CFR 122.26(a)(ii) to reflect the proposed amendment in the definition which will include discharges from mining, oil and gas, and construction activities greater than five acres. The board is also proposing a new (c) to maintain consistency with 40 CFR 122.26(a)(v) which is a federal rule requiring discharges of storm water that contribute to a violation of water quality standards, or are a significant contributor of pollutants, to obtain permit coverage.

The board is proposing to amend (7) to make minor word changes and renumber (7) as (5). This amendment is necessary to incorporate changes in the definitions, to incorporate the proposed deletion and renumbering of two subsections in (1), and to incorporate the proposed addition of (4)(c).

The board is proposing to amend (8) and renumber it as (6). The proposed amendment also deletes (8)(a) through (c), which would no longer be necessary if the amendments are adopted as proposed. The proposed amendments to the definition of "storm water discharges associated with industrial activity" at ARM 17.30.1102 will incorporate the activities described in existing (a) through (c).

The board is proposing to amend (9) and renumber it as (7). The proposed amendments maintains consistency with 40 CFR 122.26(e)(1)(ii), which is a federal rule establishing application deadlines for certain categories of industrial activities.

17.30.1106 EXCLUSIONS (1) In addition to the exclusions stated in ARM 17.30.1310, the following storm water discharges do not require MPDES permits:

(a) remains the same.

(b) existing or new discharges composed entirely of storm water from oil or gas exploration, production, processing, or treatment operations, or transmission facilities, unless the operation or facility:

(i) has had, at any time since November 16, 1987, a discharge of storm water resulting in the discharge of a reportable quantity for which notification is or was required pursuant to 40 CFR 110.6, 40 CFR 117.21 or 40 CFR 302.6; or

(ii) contributes to a violation of a water quality standard; ~~or~~

~~(iii) has a storm water discharge associated with construction activity, as defined in this subchapter~~;

(c) existing or new discharges ~~composed entirely of storm water from mining operations, unless the discharge has come into contact with any overburden, raw material, intermediate products, finished products, byproducts or waste products located on the site of such operations~~ of storm water runoff from mining operations, from oil and gas exploration, production, processing, treatment operations, or transmission facilities, if such existing or new discharges are composed entirely of flows which are from conveyances or systems of conveyances including, but not limited to, pipes, conduits, ditches, and channels, used for collection and conveying precipitation runoff and which have not come into contact with any overburden, raw material, intermediate products, finished product, byproduct, or waste products located on the site of such operation. For purposes of this rule only, "oil and gas exploration, production, processing, treatment operations or transmission facilities" means all field activities or operations associated with exploration, production, processing, or treatment operations or transmission facilities, including activities necessary to prepare a site for drilling and for the movement and placement of drilling equipment, whether or not such field activities or operations may be considered to be construction activity.

AUTH: 75-5-201, 75-5-401, MCA
IMP: 75-5-401, MCA

REASON: The board is proposing to amend ARM 17.30.1106 to maintain consistency with 40 CFR 122.26(c)(1)(iii) and 40 CFR 122.26(a)(2), which are federal rules that exclude field activities or operations associated with oil and gas exploration, production, processing , or treatment from the permit coverage requirements of ARM Title 17, chapter 30, subchapters 11 and 13 for certain discharges composed entirely of storm water.

The board is proposing to amend (1)(b)(i) and (ii) to make minor editorial changes to reflect the proposed deletion of (1)(b)(iii). The board is proposing to remove (1)(b)(iii) (the exception from the exclusion for oil and gas operations when the activity is associated with construction) because it is not found in the equivalent federal rules at 40 CFR 122.26(c)(1)(iii) and 40 CFR 122.26(a)(2) and the board has determined that it is unnecessary to maintain the exception for construction activities to protect human health and the environment because the proposed rule amendments maintain the authority to require an MPDES permit for storm water discharges associated with oil or gas exploration, production, processing, treatment operations, or transmission facilities when the operation or facility has had a storm water discharge resulting in a reportable quantify for which notification is or was required; or a storm water discharge that contributes to a violation of a water quality standard. These federal rules exclude storm water discharges from mining and from operations associated with oil and gas exploration, production, processing, treatment, or transmission facilities, including construction and other field activities from storm water discharge permit requirements provided these discharges do not come into contact with overburden, raw material, intermediate products, finished product, byproduct, or waste products located on the site of such operation.

The board is proposing to amend (1)(c) to expand the scope of this exclusion to oil and gas operations, consistent with 40 CFR 122.26(a)(2) (July 1, 2005) (the later version was vacated by the Ninth Circuit in NRDC v. U.S. EPA, 526 F.3d 591 (2008)) and with 33 USC 1342(l)(2) (CWA § 402(l)(2)), which exclude these activities from regulation under the national pollutant discharge elimination system. The board is proposing new (1)(c)(i) to clarify, for the purposes of this exclusion, the meaning of the term "oil and gas exploration, production, processing, treatment operations, or transmission facilities." This definition is consistent with 33 USC 1362(24) (§ 323 of the Energy Policy Act). These amendments are necessary to implement the storm water discharge permitting exclusions for mining and oil and gas activities provided under the federal Clean Water Act.

17.30.1107 DESIGNATION PROCEDURES: SMALL MS4S (1) through (3) remain the same.

(4) The department may designate an MS4 other than those identified in ARM 17.30.1102(23) pursuant to the criteria in ARM 17.30.1105(1) ~~(e)~~ (c) or ~~(f)~~ (d).

(5) through (13) remain the same.

AUTH: 75-5-201, 75-5-401, MCA
IMP: 75-5-401, MCA

REASON: The board is proposing to amend ARM 17.30.1107(4) to reflect changes that are proposed in ARM 17.30.1105(1). The changes in 1105(1) reflect the changes in the definition of storm water discharges associated with industrial activity to include mining, oil and gas activities, and large construction activities. The proposed changes are also necessary to maintain the correct internal cross reference to this rule and to maintain consistency with the federal definitions at 40 CFR 122.26(a)(9) and 122.26(b).

17.30.1111 APPLICATION PROCEDURES, PERMIT REQUIREMENTS: SMALL MS4S (1) Owners or operators of small MS4s shall ~~apply for authorization~~

~~under an MPDES permit as provided in ARM 17.30.1110 and this rule~~ obtain coverage under an MPDES general or individual permit and are subject to the following requirements~~.~~:

(a) and (b) remain the same.

(2) Small MS4s shall complete ~~an application for authorization~~ a notice of intent in accordance with the requirements ~~in ARM 17.30.1110~~ specified in a general permit issued pursuant to ARM 17.30.1341 or submit an application for an individual permit and comply with the application requirements set forth in (19). The ~~application~~ general permit must, ~~also include~~ at a minimum, require the following information:

(a) through (18) remain the same.

(19) An operator of a small MS4 that does not obtain coverage under a general permit must obtain coverage by the dates established in (1) and submit an application for an individual permit that includes the required permit application information specified in 40 CFR 122.26(d).

(20) The board adopts and incorporates by reference 40 CFR Part 122.26(d) (July 1, 2013), which sets forth application requirements for large or medium municipal separate storm sewers or for municipal separate storm sewers that are designated subject to permit requirements, as part of the Montana pollutant discharge elimination system. Copies of these federal regulations may be obtained from the Department of Environmental Quality, P.O. Box 200901, Helena, MT 59620-0901.

AUTH: 75-5-201, 75-5-401, MCA
IMP: 75-5-401, MCA

REASON: The board is proposing to amend the application procedures in ARM 17.30.1111 to reflect proposed changes in the general permit rule in ARM 17.30.1341, the proposed repeal of ARM 17.30.1110, and to maintain consistency with the federal regulations related to permit applications for small MS4s at 40 CFR 122.33. This federal rule sets forth application procedures and timeframes for small MS4s and is a required element of a delegated state's permit program as required by 40 CFR 123.25(a)(42). The proposed amendments are necessary to correct citations to the appropriate state regulation governing the application requirements for general permits. The permit requirements for small MS4 operators given in this rule remain unchanged. The board's specific reasons for proposing these amendments follow.

The board is proposing to amend (1) to provide that an operator of an MS4 must obtain permit coverage under a general permit or an individual permit and to remove the reference to ARM 17.30.1110 which is proposed for repeal. The proposed amendment is necessary to maintain consistency with 40 CFR 122.28(b), which sets forth the general administrative requirements for all general permits.

The board is proposing to amend (2) to provide that the operator of a small MS4 may submit a notice of intent to be covered under the general permit or submit an application for an individual permit and that the application requirements are found in new (19). ARM 17.30.1341 and the federal rule at 40 CFR 122.28(b) requires the contents of the notice of intent to be specified in the general permit.

The board is also proposing to amend (2) to clarify that the general permit must at minimum contain the elements in (a) through (c). These amendments are necessary to maintain consistency with the equivalent federal rule related to permit applications for small MS4s found at 40 CFR 122.33 and 122.34, which are required by 40 CFR 123.25(42) to be part of a delegated state's permit program.

The board is proposing to add new (19) to the application requirements for a small MS4 to maintain consistency with the equivalent federal rule found at 40 CFR 122.33, which is required by 40 CFR 123.25(42) to be part of a delegated state's permit program.

The board is proposing new (20) to incorporate 40 CFR 122.26(d) and the federal application requirements applicable to large or medium MS4s, or to small MS4s that are either designated or choose to obtain a permit for their discharges in order to retain state primacy under the federal clean water act.

17.30.1341 GENERAL PERMITS (1) The department may issue general permits ~~for the following categories of point sources which the board has determined are appropriate for general permitting under the criteria listed in 40 CFR 122.28 as stated in ARM 17.30.1105~~ in accordance with the following:

~~(a) cofferdams or other construction dewatering discharges;~~

~~(b) ground water pump test discharges;~~

~~(c) fish farms;~~

~~(d) placer mining operations;~~

~~(e) suction dredge operations using suction intakes no larger than four inches in diameter;~~

~~(f) oil well produced water discharges for beneficial use;~~

~~(g) animal feedlots;~~

~~(h) domestic sewage treatment lagoons;~~

~~(i) sand and gravel mining and processing operations;~~

~~(j) point source discharges of storm water;~~

~~(k) treated water discharged from petroleum cleanup operations;~~

~~(l) discharges from public water supply systems, as determined under Title 75, chapter 6, MCA;~~

~~(m) discharges to wetlands that do not contain perennial free surface water;~~

~~(n) discharges from road salting operations;~~

~~(o) asphalt plant discharges;~~

~~(p) discharges of hydrostatic testing water;~~

~~(q) discharges of noncontact cooling water;~~

~~(r) swimming pool discharge;~~

~~(s) septic tank pumper disposal sites; and~~

~~(t) pesticide application.~~

(a) The general permit must be written to cover one or more categories or subcategories of discharges or facilities described in the permit under (b), except those covered by individual permits, within a geographic area. The area should correspond to existing geographic or political boundaries such as:

(i) designated panning area under sections 208 and 303 of the federal Clean Water Act;

(ii) sewer districts or sewer authorities;

(iii) city, county or state political boundaries;

(iv) state highway systems;

(v) standard metropolitan statistical areas as defined by the federal Office of Management and Budget;

(vi) urbanized areas as designated by the U.S. Bureau of Census; or

(vii) any other appropriate division or combination of boundaries.

(b) the general permit may be written to regulate one or more categories or subcategories of discharges or facilities, within the area described in (1)(a), where the sources within a covered subcategory of discharges are either:

(i) storm water point sources; or

(ii) one or more categories or subcategories of point sources, other than storm water point sources, if the sources within each category or subcategory all:

(A) involve the same or substantially similar types of operations;

(B) discharge the same types of wastes;

(C) require the same effluent limitations or operating conditions;

(D) require the same or similar monitoring; and

(E) in the opinion of the department, are more appropriately controlled under a general permit than under individual permits.

(c) Where sources within a specific category or subcategory of discharges are subject to water quality-based limits imposed pursuant to 40 CFR122.44(d), the sources in that specific category or subcategory shall be subject to the same water quality-based effluent limitations.

(d) The general permit must clearly identify the applicable conditions for each category or subcategory of discharges covered by the permit.

(e) The general permit may exclude specified sources or areas from coverage.

(2) ~~Although MPDES general permits may be issued for a category of point sources located throughout the state, they may also be restricted to more limited geographical areas.~~ General permits may be issued, modified, revoked and reissued, or terminated by the department in accordance with applicable requirements of ARM 17.30.1363 through 17.30.1365, and ARM 17.30.1370 through 17.30.1378. In accordance with 40 CFR 123.44(a)(2), EPA has 90 days from the date it receives a proposed general permit to comment upon, object to, or make any recommendations with respect to the proposed general permit. The effective date of an MPDES general permit is 90 days after the receipt of the proposed permit by EPA.

(3) Prior to issuing a ~~MPDES~~ general permit, the department shall ~~prepare~~ provide a public notice ~~which includes the equivalent of information listed in ARM 17.30.1372(6) and shall publish the same as follows:~~ in accordance with the requirements of ARM 17.30.1372 and shall adhere to the requirements of ARM 17.30.1373 through 17.30.1377 regarding public comments and public hearings. The department shall provide a copy of the public notice~~, as follows:~~

~~(a) prior to publication, notice to the U.S. Environmental Protection Agency;~~

~~(b) direct mailing of notice~~ to the Water Pollution Control Advisory Council and to any persons who may be affected by the proposed general permit~~;~~

~~(c) publication of notice in a daily newspaper in Helena and in other daily newspapers of general circulation in the state or affected area;~~

~~(d) after publication, a hearing must be held and a 30-day comment period allowed as provided in ARM 17.30.1372 through 17.30.1377 and 17.30.1383~~.

(4) A person owning or proposing to operate a point source who wishes to ~~operate~~ obtain coverage under a ~~MPDES~~ general permit shall ~~complete~~ submit to the department a ~~standard MPDES application or~~ written notice of intent ~~form available from the department for the particular~~ to be covered by the general permit. A discharger who fails to submit a written notice of intent in accordance with the terms of the general permit may not discharge under the permit. A complete and timely notice of intent to be covered in accordance with general permit requirements fulfills the requirements for permit application for purposes of ARM 17.30.1023, 17.30.1105, 17.30.1313, and 17.30.1322. ~~Except for notices of intent, the department shall, within 30 days of receiving a completed application, either issue to the applicant an authorization to operate under the MPDES general permit, or shall notify the applicant that the source does not qualify for authorization under a MPDES general permit, citing one or more of the following reasons as the basis for denial:~~

~~(a) the specific source applying for authorization appears unable to comply with the following requirements:~~

~~(i) effluent standards, effluent limitations, standards of performance for new sources of pollutants, toxic effluent standards and prohibitions, and pretreatment standards;~~

~~(ii) water quality standards established pursuant to 75-5-301, MCA;~~

~~(iii) prohibition of discharge of any radiological, chemical, or biological warfare agent or high-level radioactive waste;~~

~~(iv) prohibition of any discharge which the secretary of the army acting through the chief of engineers finds would substantially impair anchorage and navigation;~~

~~(v) prohibition of any discharges to which the regional administrator has objected in writing;~~

~~(vi) prohibition of any discharge which is in conflict with a plan or amendment thereto approved pursuant to section 208(b) of the Act; and~~

~~(vii) any additional requirements that the department determines are necessary to carry out the provisions of 75-5-101, et seq., MCA.~~

~~(b) the discharge is different in degree or nature from discharges reasonably expected from sources or activities within the category described in the MPDES general permit;~~

~~(c) an MPDES permit or authorization for the same operation has previously been denied or revoked;~~

~~(d) the discharge sought to be authorized under a MPDES general permit is also included within an application or is subject to review under the Major Facility Siting Act, 75-20-101, et seq., MCA;~~

~~(e) the point source will be located in an area of unique ecological or recreational significance. Such determination must be based upon considerations of Montana stream classifications adopted under 75-5-301, MCA, impacts on fishery resources, local conditions at proposed discharge sites, and designations of wilderness areas under 16 USC 1132 or of wild and scenic rivers under 16 USC 1274.~~

(5) ~~Where authorization to operate under a MPDES general permit is denied,~~

~~or a notice of intent under ARM 17.30.1115 is not applicable, the department shall proceed, unless the application or notice of intent is withdrawn, to process the application or notice of intent through the individual MPDES permit requirements under this subchapter.~~ Subject to (a) and (b), the contents of the written notice of intent must be specified in the general permit and must contain information necessary for adequate program implementation including, at a minimum, the legal name and address of the owner or operator, the facility name and address, type of facility or discharges, and the receiving stream(s). A notice of intent must be signed in accordance with ARM 17.30.1323. In addition to these general requirements, the following specific provisions apply:

(a) Subject to the department's approval, a general permit for storm water discharges associated with industrial activity from inactive mining, inactive oil and gas operations, or inactive landfills occurring on federal lands where an operator cannot be identified may contain alternative information and meet notice of intent requirements.

(b) Notices of intent for coverage under a general permit for concentrated animal feeding operations must include the information required in the Notice of Intent for MPDES Application for New and Existing Concentrated Animal Feeding Operation (CAFO Notice of Intent) provided by the department and the information specified in 40 CFR122.21(i)(1), including a topographic map of the area in which the CAFO is located.

(6) ~~Every MPDES general permit must have a fixed term not to exceed five years. Except as provided in (10), every authorization to operate under a MPDES general permit expires at the same time the MPDES general permit expires.~~ Each general permit must specify the deadline for submitting notices of intent to be covered and the dates(s) when a discharger is authorized to discharge under the permit.

(7) A general permit must specify, by one of the following methods, whether a discharger that has submitted a complete and timely notice of intent to be covered under the general permit is authorized to discharge under the permit:

(a) upon receipt of the notice of intent by the department;

(b) after a waiting period specified in the general permit;

(c) on a date specified in the general permit; or

(d) upon receipt of written notification of authorization from the department.

~~(7)~~ (8) Where authorization to ~~operate~~ discharge under a ~~MPDES~~ general permit is denied solely because the source is already ~~issued to, or a notice of intent received from, a point source~~ covered by an individual ~~MPDES~~ permit, the ~~department~~ owner or operator may request ~~shall, upon issuance of the authorization to operate or receipt of the notice of intent under~~ termination of the ~~MPDES~~ individual ~~general~~ permit~~, terminate the individual MPDES permit~~ and coverage for that ~~point~~ source under the general permit. Upon termination of the individual permit, the general permit applies to the source.

~~(8)~~ (9) Any person authorized or eligible to ~~operate~~ discharge under a ~~MPDES~~ general permit may at any time, upon providing reasons supporting the request or application, apply for an individual ~~MPDES~~ permit ~~according to the procedures in this subchapter~~. Upon issuance of the individual ~~MPDES~~ permit, the ~~department shall terminate any MPDES general permit authorization or notice of~~

~~intent held by such person~~ authorization to discharge under the general permit automatically terminates.

~~(9)~~ (10) The department, on its own initiative or upon the petition of any interested person, may ~~modify, suspend, or revoke in whole or in part a MPDES general permit or an authorization or notice of intent to operate under a MPDES general permit during its term in accordance with the provisions of ARM 17.30.1361 for any cause listed in ARM 17.30.1361~~ require any discharger authorized by a general permit to obtain an individual permit ~~for~~ under any of the following ~~causes~~ circumstances:

(a) ~~the approval of~~ a water quality management plan has been approved that contains requirements applicable to categories or subcategories of discharges or facilities ~~point sources~~ covered in ~~the MPDES~~ a general permit;

(b) ~~determination by~~ the department has determined that the discharge from ~~any~~ the authorized source is a significant contributor to pollution as determined by the factors set forth in 40 CFR ~~122.26(c)(2)~~ 122.28(b)(3) including the location of the discharge, the size of the discharge, the quantity and nature of the pollutants discharged, and other relevant factors; ~~or~~

(c) a change has occurred in the availability of demonstrated technology or practices for the control or abatement of pollutants applicable to ~~a~~ the source or to a category ~~of sources~~ or subcategory of discharges or facilities;

(d) ~~occurrence of one or more of the following circumstances:~~ the discharger is not in compliance with the conditions of the general permit;

~~(i) violation of any conditions of the permit; or~~

~~(ii) obtaining an MPDES permit by misrepresentation or failure to disclose fully all relevant facts;~~

(e) circumstances have changed since the time of the request to be covered by the general permit so that the discharger is no longer appropriately controlled under the general permit;

(f) effluent limitations guidelines (ELGs) have been promulgated for the source, or a category or subcategory of discharges or facilities covered under the general permit; or

~~(iii)~~ (g) there is a change in any condition that requires either a temporary or permanent reduction or elimination of the ~~authorized~~ discharge authorized under the general permit~~; or~~

~~(iv) a failure or refusal by the permittee to comply with the requirements of 75-5-602, MCA~~.

~~(10)~~ (11) ~~The department may reissue an authorization to operate under a MPDES general permit provided that the requirements for reissuance of MPDES permits specified in ARM 17.30.1322 are met.~~ The department may require any owner or operator authorized to discharge under a general permit to apply for an individual permit as provided in (10) only upon written notice to the owner or operator that an individual permit application is required. This notice must include a brief statement of the reasons for this decision, an application form, a statement setting a time for the owner or operator to file the application, and a statement that on the effective date of the individual permit the general permit as it applies to the individual permittee will automatically terminate. The department may grant additional time upon request of the applicant.

~~(11)~~ (12) The department shall maintain and make available to the public a register of all sources and activities authorized to discharge ~~operate, or with notices of intent to discharge,~~ under each ~~MPDES~~ general permit, including the location of such sources and activities, and shall provide copies of such registers upon request.

(12) remains the same, but is renumbered (13).

~~(13)~~ (14) For purposes of this rule, the board hereby adopts and incorporates by reference ~~(see ARM 17.30.1303 for complete information about all materials incorporated by reference~~):

(a) ~~40 CFR 122.28 (July 26, 2012), which sets forth criteria for selecting categories of point sources appropriate for general permitting;~~

(b) ~~40 CFR 124.10 (July 26, 2012) which sets forth minimum contents of public notices; and~~

~~(c)~~ (a) 40 CFR 122.23(h) (July 1, 2012), which sets forth procedures for CAFOs seeking coverage under a general permit~~.~~;

(b) 40 CFR 122.44 (July 1, 2013), which sets forth procedures for establishing limitations, standards, and other permit conditions;

(c) 40 CFR 123.44(a)(2) (July 1, 2013), which sets forth timeframes for EPA to object to general permits; and

(d) 40 CFR 122.21(i)(1) (July 1, 2013), which sets forth application requirements for new and existing concentrated animal feeding operations.

AUTH: 75-5-201, 75-5-401, MCA
IMP: 75-5-401, MCA

REASON: The board is proposing to amend the general permit requirements in ARM 17.30.1341 in order to maintain consistency with the federal requirements set forth in 40 CFR 122.28 and provide a uniform rule for the issuance and administration of general permits under both the MPDES and ground water pollution control system (GWPCS) programs. The board is proposing to adopt these federal requirements because they are required elements of a delegated state's permit program and are required to implement the federal Clean Water Act's national pollutant discharge elimination system (NPDES) program. See 40 CFR 123.25. In general, the proposed amendments add criteria for coverage and administrative requirements, clarify public notice and public hearing requirements, and update incorporations by reference to applicable federal rules. The board's specific reasons for adopting the federal requirements into various sections of ARM 17.30.1341 follow. The proposed amendments also make minor changes to wording and punctuation to conform to standard practices for rule formatting.

The board is proposing to amend (1) by adding, consistent with 40 CFR 122.28(a)(1) and (2), criteria with which the department can issue general permits and by removing the specific categories of discharges, which had been listed in (1), as general permits were not developed for some of those categories and some of the categories are not subject to permits such as discharges from road salting and septic systems. Categories of discharges currently listed may still be covered by a general permit provided they meet the criteria now proposed in (1).

The board is proposing new language in (2) to provide that general permits are subject to the same requirements for issuance, modification, revocation and reissuance, and termination as set forth in ARM Title 17, chapter 30, subchapter 13 except that the issuance date is delayed for 90 days to allow EPA to review and object to state issued general permits. The amendment is necessary to maintain consistency with 40 CFR 123.44(a)(2), which is the equivalent federal rule. Existing text in (2) that is redundant with the requirements and categories for issuing general permits given in (1) and in ARM Title 17, chapter 30, subchapter 13 is proposed to be stricken.

It is necessary to amend the public notice requirements for general permits, as the board proposed in (3), in order for the rules in subchapter 11 to reference the public comment and public hearing provisions in subchapter 13 and to be consistent with the board's public notice rules in ARM 17.30.1372 and 17.30.1343 through 17.30.1379, which set forth procedures for responding to public comment and for holding public hearings. After these amendments become effective, permits issued under subchapter 11 will follow the public comment and public hearing provisions in subchapter 13. The board is proposing to retain the requirement in (3) that notice of the general permit be provided to the Water Pollution Control Advisory Council (WPCAC) and to any person affected by the general permit.

The board is proposing to amend the requirements to obtain coverage under a general permit, set forth in (4), to be consistent with the federal rule at 40 CFR 122.28(b)(2). The proposed amendments are necessary to remove the requirement that an owner or operator submit a complete application form because these proposed amendments will instead require an owner or operator wishing to obtain coverage under a general permit to submit a notice of intent. Standardizing the format and procedure serves an objective of general permitting, which is to expedite permitting and lessen the department's administrative burden for groups of similar discharges. The board is also proposing to remove the current rule's requirement to cite one of several specifically-listed reasons when coverage is denied. Many of these "reasons" appear in 40 CFR 122.4 and ARM 17.30.1311 and are not specific to general permits. Federal regulations at 40 CFR 122.28 do not include any such requirements for denial of general permit coverage. Instead, conditions for requiring an individual permit, the equivalent of denial of coverage under a general permit, are given in (11), as amended.

The board proposes amendments to (5) to set forth the contents of a notice of intent that are necessary for the program to identify the owner or operator and the discharging facility, properly implement the storm water program, and specify that the signatory requirements for a notice of intent are given in ARM 17.30.1323. It is also necessary that the board propose removal of language regarding denial of general permit coverage as coverage under a general permit is not denied, rather the discharger is required to obtain individual permit coverage. Proposed (5)(a) is necessary to address specific situations where alternative notice of intent requirements may be necessary for certain storm water discharges from inactive facilities on federally-owned lands. Proposed (5)(b) is necessary to provide that notices of intent to obtain coverage under a general permit for concentrated animal feeding operations (CAFOs) must be consistent with the federal rule at 40 CFR 122.21(i)(1).

The board is proposing to amend (6) to remove duplicative language and the condition that all authorizations expire on the date the general permit expires and replace it with new language to clarify that the general permit must specify the deadline for submitting a notice of intent and when permit coverage begins. ARM 17.30.1346 specifies that all MPDES permits are effective for a fixed term not to exceed five years, which applies to general permits as well. ARM 17.30.1313 addresses the continuation of expiring permits. The new language is necessary to maintain consistency with the federal requirements at 40 CFR 122.28(b)(2)(iii).

The board is proposing new (7) to specify the method in the general permit by which the permitee will be informed that it is authorized to discharge. The four methods for informing a permittee that it is authorized to discharge under a general permit are: upon receipt of the notice of intent; after a waiting period specified in the permit; on a specific date; or upon written notification by the department. These provisions are necessary to maintain consistency with 40 CFR 122.28(b)(2)(iv).

The board is proposing to amend (7) and renumber it as (8). The proposed amendments remove language that is specific to MPDES permits in order to include and accommodate ground water permits and to remove language that refers to a notice of intent, because that requirement is not consistent with the federal or state regulations governing individual permits and adds nothing to the intent of the rule, which is to provide a process for transferring coverage from an individual to a general permit. The board is also proposing to change the term 'operate' to 'discharge' to clarify that permits only authorize the discharge of pollutants and do not control other aspects of the facilities operations. These provisions are necessary to maintain consistency with the federal requirements for transferring coverage from an individual permit to a general permit in 40 CFR 122.28(b)(3)(v).

The board is proposing to amend (8) and renumber it as (9). The proposed amendments are necessary remove language specific to MPDES permits, remove language referring to receipt of a notice of intent, and add a requirement that the permittee submit the reasons for requesting an individual permit along with the permit application. Such a requirement for "reasons" is consistent with 40 CFR 122.28 (b)(3)(iii), which provides a process for an owner or operator to request exclusion from the coverage of a general permit by applying for an individual permit. The request will be granted by issuing an individual permit if the reasons cited by the owner or operator are adequate to support the request. This provides the department reasonable_discretion to deny coverage under an individual permit in the case where a discharger is already properly covered by a general permit. An objective of general permitting is to ease the department's administrative burdens. Therefore, dischargers should not be able to routinely opt out of coverage by requesting an individual permit. The new language also specifies that the authorization to discharge under the general permit is terminated upon issuance of the individual permit.

The board is proposing to amend (9) and renumber it as (10). The proposed amendments are necessary to remove language that allows the department, on its own initiative or upon request by any interested person, to modify, suspend, or revoke, in whole or in part, a general permit, an authorization, or notice of intent to operate under a general permit. In accordance with (2), general permits are issued, modified, revoked and reissued, or terminated in accordance with applicable provisions of ARM Title 17, chapter 30, subchapter 13. The proposed language in (10) is consistent with the federal rule at 40 CFR 122.28(b)(3),which specifies the conditions under which an individual discharger authorized under a general permit may be required to obtain an individual permit. The result of the proposed change to (10) is that interested persons may petition the department to require that a discharger, covered by a general permit, be required to obtain an individual permit where the conditions in (10)(a) through (g) are present. These provisions are necessary to maintain consistency with the federal requirements in 40 CFR 122.28(b)(3) for requiring a discharger authorized by a general permit to obtain an individual permit.

The board is proposing to amend (10) and renumber it as (11). The proposed amendments are necessary to remove provisions related to reissuance of an authorization to discharge under a general permit when the requirements of ARM 17.30.1322 are met. This proposed amendment is necessary because ARM 17.30.1322 establishes extensive application requirements for MPDES permits, but excludes "persons covered by general permits under ARM 17.30.1341" from the application requirements. The equivalent federal rule at 40 CFR 122.28(b)(2)(i) states that "[a] complete and timely notice of intent (NOI) to be covered in accordance with general permit requirements fulfills the requirements for permit applications." The equivalent language is proposed in the amendments to (4). The board is proposing new language in (11) that will require written notification from the department when a discharger under a general permit is required to submit an application for an individual permit. This notification must include the basis for the decision, appropriate application form(s), and timeframes for submittal of the individual permit application. The new language also specifies that coverage under the general permit terminates upon the effective date of the individual permit. This requirement is consistent with 40 CFR 122.28(b)(3)(ii) for EPA-issued permits.

The board is proposing to amend (11) and renumber it as (12). This amendment is necessary to make technical corrections, to remove language referring to the notice of intent, and to remove language specific to MPDES permits.

The board is proposing to amend (13) and renumber it as (14). The amendments are necessary to incorporate by reference federal rules that support ARM 17.30.1341 and are proposed for incorporation by reference. Two of the federal rules currently incorporated by reference are no longer necessary to support this rule and will be deleted because the criteria for categories of point sources appropriate for general discharge permits are now set forth in (1) and the criteria for public notice in subchapter 13 will apply to general permits. In order to maintain state primacy, the board is proposing to incorporate by reference the following federal rules: 40 CFR 122.44, which sets forth procedures for establishing limitations, standards, and other permit conditions necessary to support the categories of general permits in proposed (1)(c); 40 CFR 123.44(a)(2), which sets forth timeframes for EPA to object to state-issued general permits necessary to support general permit actions by the department under proposed (2); and 40 CFR 122(i)(1), which sets forth application requirements for CAFOs necessary to define notice of intent requirements for such facilities in proposed (5)(b).

<u>17.30.1342 CONDITIONS APPLICABLE TO ALL PERMITS</u> <u>(1)</u> The ~~following~~ conditions <u>described in this rule</u> apply to all MPDES permits. Additional conditions applicable to MPDES permits are set forth in ARM <u>17.30.1343</u>

~~17.30.1344~~. All conditions applicable to MPDES permits must be incorporated into the permits either expressly or by reference. If incorporated by reference, a specific citation to these rules must be given in the permit.

~~(1)~~ (2) The permittee shall comply with all standard conditions in 40 CFR 122.41 and all conditions of this permit. Any permit noncompliance constitutes a violation of the Act and is grounds for enforcement action; for permit termination, revocation and reissuance, or modification; or denial of a permit renewal application.

~~(a)~~ (3) The permittee shall comply with effluent standards or prohibitions established under the Act and rules adopted thereunder including limitations ~~ARM 17.30.1206~~ for toxic pollutants in ARM 17.30.1206 and is required by federal law to comply with technology-based effluent limitations for solids, sludge, and other pollutants removed in the course of wastewater treatment set forth in ARM Title 17, chapter 30, subchapter 12 within the time provided in the rules that establish these standards or prohibitions, even if the permit has not yet been modified to incorporate the requirement.

~~(b)~~ (4) The Act provides that any person who violates a permit condition or limitation is subject to a civil penalty not to exceed $~~10,000~~ 25,000 per day ~~of such~~ for each violation. Any person who willfully or negligently violates 75-5-605, MCA, including a permit condition or limitation, is subject to ~~a fine~~ criminal penalties not to exceed $25,000 per day of violation, ~~or~~ imprisonment for not more than one year, or both. In the case of a second or subsequent conviction for a willful or negligent violation, a person is subject to a fine of not more than $50,000 per day of violation, imprisonment of not more than two years, or both. The Act provides that any person who violates a permit condition or limitation may be assessed an administrative penalty by the department not to exceed $10,000 per violation per day, with the maximum penalty assessed not to exceed $100,000 for any related series of violations.

(2) remains the same, but is renumbered (5).

~~(3)~~ (6) It ~~may~~ is not ~~be~~ a defense for a permittee in an enforcement action that it would have been necessary to halt or reduce the permitted activity in order to maintain compliance with the conditions of this permit.

(4) through (8) remain the same, but are renumbered (7) through (11).

~~(9)~~ (12) The permittee shall allow the department, or an authorized representative, including an authorized contractor acting as a representative of the department, upon the presentation of credentials and other documents as may be required by law, to:

(a) through (d) remain the same.

~~(10)(a)~~ (13) Samples and measurements taken for the purpose of monitoring must be representative of the monitored activity.

~~(b)~~ (14) Except for records and monitoring information required by this permit that are related to the permittee's sewage sludge use and disposal activities, which must be retained for a period of at least five years, or longer, ~~T~~the permittee shall retain records of all monitoring information, including all calibration and maintenance records and all original strip chart recordings for continuous monitoring instrumentation, copies of all reports required by this permit, and records of all data used to complete the application for this permit, for a period of at least three years from the date of the sample, measurement, report or application. This period may be extended by request of the department at any time.

(c) Records of monitoring information must include:

(i) through (vi) remain the same, but are renumbered (a) through (f).

(d) (15) Monitoring must be conducted according to test procedures approved under 40 CFR Part 136, unless other test procedures have been specified in this permit another method is required under 40 CFR 503.8 or Subchapter N.

(16) The Act provides that any person who falsifies, tampers with, or knowingly renders inaccurate any monitoring device or method required to be maintained under this permit shall, upon conviction, be punished by a fine of not more than $25,000, imprisonment for not more than six months, or both.

(11) (17) All applications, reports, or information submitted to the department must be signed and certified. (See ARM 17.30.1323.) as required by ARM 17.30.1323.

(12)(a) (18) The permittee shall give notice to the department, as soon as possible, of any planned physical alterations or additions to the permitted facility. Notice is required only when:

(i) (a) the alteration or addition to a permitted facility may meet one of the criteria for determining whether a facility is a new source in ARM 17.30.1340(2); or

(ii) remains the same, but is renumbered (b).

(b) remains the same, but is renumbered (19).

(c) (20) This permit is not transferable to any person, except after notice to the department. The department may require modification or revocation and reissuance of the permit to change the name of the permittee and incorporate such other requirements as may be necessary under the Act. (See ARM 17.30.1360; in some cases, modification or revocation and reissuance is mandatory.) or mandatory, as required by ARM 17.30.1360 and the Act.

(d) (21) Monitoring results must be reported at the intervals specified elsewhere in this permit and subject to the following requirements.:

(i) remains the same, but is renumbered (a).

(ii) (b) If the permittee monitors any pollutant more frequently than required by the permit, using test procedures approved under 40 CFR 136, or as using procedures specified in the permit for any pollutant for which an analytical method is not established by 40 CFR Part 136, or by another method required for an industry-specific waste stream under 40 CFR 503.8 or 40 CFR subchapter N, the results of such monitoring must be included in the calculation and reporting of the data submitted in the DMR.

(iii) (c) Calculations for all limitations, which require averaging of measurements, must utilize an arithmetic mean unless otherwise specified by the department in the permit.

(e) (22) Reports of compliance or noncompliance with, or any progress reports on, interim and final requirements contained in any compliance schedule of this permit must be submitted no later than 14 days following each schedule date.

(f)(i) (23) The permittee shall report any noncompliance which may endanger health or the environment. Any information must be provided orally within 24 hours from the time the permittee becomes aware of the circumstances. A written submission must also be provided within five days of the time the permittee becomes aware of the circumstances. The written submission must contain a description of the noncompliance and its cause; the period of noncompliance, including exact dates and times, and if the noncompliance has not been corrected, the anticipated time it is expected to continue; and steps taken or planned to reduce, eliminate, and prevent reoccurrence of the noncompliance.

(ii) (24) The following must be included as information which must be reported within 24 hours under this rule:

(A) and (B) remain the same, but are renumbered (a) and (b).

(C) (c) violation of a maximum daily discharge limitation for any of the pollutants listed by the department in the permit to be reported within 24 hours (see ARM 17.30.1344 and as required by 40 CFR 122.44(g) and 40 CFR 122.41).

(iii) (25) The department may waive the written report on a case-by-case basis for reports under (ii) (24), above, if the oral report has been received within 24 hours.

(g) (26) The permittee shall report all instances of noncompliance not reported under (a) (18)(a), (d) (21), (e) (22), and (f) (23), at the time monitoring reports are submitted. The reports must contain the information listed in (f) (23).

(h) remains the same, but is renumbered (27).

(13)(a) (28) The permittee may allow any bypass to occur which does not cause effluent limitations to be exceeded, but only if it also is for essential maintenance to assure efficient operation. These bypasses are not subject to the provisions of (b) and (c) (29)(a) and (30).

(29) Bypasses are subject to the following notification requirements:

(b) (a) If the permittee knows in advance of the need for a bypass, it shall submit prior notice to the department, if possible at least ten days before the date of the bypass. The permittee shall submit notice of an unanticipated bypass as required in (12)(f) (24-hour notice).

(b) The permittee shall submit notice of an unanticipated bypass as required in (23), except as provided in (28).

(c) (30) Except as provided in (29), Bbypass is prohibited, and the department may take enforcement action against a permittee for bypass, unless:

(i) and (ii) remain the same, but are renumbered (a) and (b).

(iii) (c) the permittee submitted notices as required under (c) (30).

(d) (31) The department may approve an anticipated bypass, after considering its adverse effects, if the department determines that it will meet the three conditions listed above in (c)(i) (30)(a).

(14)(a) (32) An upset constitutes an affirmative defense to an action brought for noncompliance with such technology-based permit effluent limitations if the requirements of (b) (33) are met. No determination made during administrative review of claims that noncompliance was caused by upset, and before an action for noncompliance, is final administrative action subject to judicial review.

(b) (33) A permittee who wishes to establish the affirmative defense of upset shall demonstrate, through properly signed, contemporaneous operating logs, or other relevant evidence that:

(i) and (ii) remain the same, but are renumbered (a) and (b).

(iii) (c) the permittee submitted notice of the upset as required in (12)(f)(ii)(B) (24)(b) (24-hour notice); and

(iv) (d) the permittee complied with any remedial measures required under

~~(4)~~ (7).

(c) remains the same, but is renumbered (34).

~~(15)~~ (35) The board ~~hereby adopts and incorporates herein by reference (see ARM 17.30.1303 for complete information about all materials incorporated by reference)~~ adopts and incorporates by reference the following federal regulation as part of the Montana pollutant discharge elimination system. Copies of these federal regulations may be obtained from the Department of Environmental Quality, P.O. Box 200901, Helena, MT 59620-0901:

(a) 40 CFR Part 136 (July 1, 2013), which ~~is a series of federal agency rules setting~~ sets forth guidelines establishing test procedures for the analysis of pollutants;~~ and~~

(b) 40 CFR 122.41 (conditions applicable to all discharge permits);

~~(b)~~ (c) 40 CFR 122.44(g) (July 1, 2013), which ~~is a federal agency rule~~ sets forth notification requirements requiring 24-hour notice of any violation of maximum daily discharge limits for toxic pollutants or hazardous substances;

(d) 40 CFR 503.8 (July 1, 2013), which sets forth sampling and analytical methods for sewage sludge that are approved for use in NPDES permits; and

(e) 40 CFR Subchapter N (July 31, 2013), which sets forth technology-based effluent limitations and specific analytical methods applicable to these limitations.

AUTH: 75-5-201, 75-5-401, MCA
IMP: 75-5-401, MCA

REASON: The board is proposing to amend the conditions applicable to all permits in ARM 17.30.1342 in order to make the rule consistent with the equivalent federal requirements set forth in 40 CFR 122.41 and the Montana Water Quality Act. ARM 17.30.1342 defines and establishes certain conditions which apply to all MPDES permits and must be incorporated into the permits either expressly or by reference. The proposed amendments update the standard permit language to incorporate changes in the Montana Water Quality Act for assessment of civil and administrative penalties for noncompliance with permit conditions. The proposed amendments make minor changes to wording and punctuation to conform to standard practices for rule formatting. The board's specific reasons for deletions and amendments to ARM 17.30.1342 follow. The board has also renumbered the rule to simplify the rule and make it more readable.

The board is proposing to amend the language in new (1) to correct the reference for additional conditions applicable to certain categories of permits from ARM 17.30.1344 to 17.30.1343. ARM 17.30.1343 is the board's rule that is equivalent to 40 CFR 122.42 in federal rule, which contains the additional conditions that are applicable to certain categories of permits. This amendment is necessary to maintain consistency with the federal rule at 40 CFR 122.41 and to correct formatting.

The board is proposing to amend current (a) to add language requiring compliance with the limitations and timeframes for toxic pollutants and for sewage sludge use and disposal in the Act and rules adopted thereunder, to provide that failure to comply with these standards and limitations is a violation of the permit even if the permit has not been modified to include these requirements, and to renumber

(a) to (1)(a)(i). The federal CWA requires the administrator of the EPA to identify and promulgate effluent standards for toxic pollutants and to periodically revise and update the list of toxic pollutants and applicable standards for each listed toxic pollutant. Section 405(d) of the federal CWA requires the administrator of the EPA to develop and promulgate regulations governing the use and disposal of sewage sludge and identify and regulate toxic pollutants which may be present in such material. The state incorporates these requirements as standard permit conditions by incorporating 40 CFR 122.41 by reference. The permittee must comply with both of these federal provisions even if the permit has not been modified to incorporate these requirements. This amendment is necessary to maintain consistency with the federal requirements and standard conditions at 40 CFR 122.41(a)(1) and to correct formatting. Current (a) is proposed to be renumbered (3).

The board is proposing to amend current (b), regarding a permittee's duty to comply with the Montana Water Quality Act (the Act) and all permit conditions, by clarifying what civil, criminal, and administrative penalties may result from non-compliance with the permit or the applicable requirements under the Act or administrative rules and by renumbering. These changes are necessary to provide notice of penalties for non-compliance with permit conditions, the Act, and rules and to correct erroneous language. The board is also proposing to add language addressing administrative penalties that may be assessed under 75-5-611, MCA, for permit violations or violations of the Act. Administrative penalties may be assessed in the amount of up to $10,000 per day for each violation, but not exceed $100,000 for a series of related violations. These amendments are necessary to maintain consistency with the Act and 40 CFR 122.41(a) and 123.27(a). Current (b) is proposed to be renumbered (4).

The board is proposing to amend current (3), regarding compliance responsibilities for permittees, to make a minor word change and to renumber (3) to (6).

The board is proposing to amend current (9), which adopts and incorporates federal requirements regarding inspection and entry of permitted facilities by the department, to authorize a contractor, who presents appropriate credentials and is acting as a representative of the department, to access a permittee's premises and inspect and perform sampling to determine permit compliance. This amendment is necessary to maintain consistency with the federal rule at 40 CFR 122.41(i). Current (9) is proposed to be renumbered (12).

The board is proposing to amend current (10)(a), which incorporates federal requirements regarding monitoring and records, and to renumber (10)(a) to (13). The board is proposing to amend current (10)(b) to include language requiring monitoring records related to sludge use and disposal to be kept for five years and to renumber (10)(b) to (14). This amendment is necessary to maintain consistency with 40 CFR 122.41(j). The board is proposing a minor word change to current (10)(c) and is proposing to renumber (i) through (vi) as (a) through (f). The board is also proposing to amend current (10)(d), which specifies approved testing procedures to include methods specified in 40 CFR 503.8 and subchapter N, which are federal regulations governing sewage sludge monitoring requirements and technology-based effluent limitation guidelines, respectively. This amendment is necessary to maintain consistency with 40 CFR 122.41(j). Current (10)(d) is proposed to be renumbered (15). The board is also proposing to add a new (16) to establish penalties that are consistent with 75-5-633, MCA, for falsifying, tampering with, or knowingly altering monitoring equipment or test methods causing inaccurate monitoring results. This amendment is necessary to maintain consistency with 40 CFR 122.41(j)(5).

The board is proposing to amend current (11), regarding signatory requirements, to make minor editorial changes, and renumber (11) to (17).

The board is proposing to amend current (12)(a), regarding the permittee's reporting and notification requirements, to correct minor changes to wording and punctuation, and to renumber (12)(a) to (18). The board is proposing to clarify when the permittee is required to notify the department of alterations or additions to permitted facilities and to correct formatting. Current (12)(a)(i) is proposed to be renumbered (18)(a) and the reference to ARM 17.30.1340(2), regarding new sources, is corrected to make the reference applicable to the entire rule. The board is also proposing to modify current (12)(c) and (12)(d) to make minor editorial changes and to renumber (12)(c) to (20) and (12)(d) to (21). The board is also proposing to amend current (12)(d)(ii) to include analytical results obtained using test methods that are specified in 40 CFR 136, the permit, 40 CFR 503.8, or 40 CFR subchapter N in permit calculations that are reported to the department in the DMR. Current (12)(d)(ii) is proposed to be renumbered (21)(b). Federal regulations at 40 CFR 136, 40 CFR 503.8, and 40 CFR subchapter N address effluent limitations that are adopted by the board at ARM 17.30.1207 and are required to be included in all MPDES permits issued by the department. In some cases, the effluent limitations given in these subchapters require specific analytical methods that are not included in 40 CFR 136, but are applicable to a specific industrial category. The board is proposing to make minor editorial changes to current (12)(d)(iii), (12)(e), (12)(f)(i), and (12)(f)(ii) and renumber them to (21)(c), (22), (23), and (24), respectively. The board is also proposing to modify current (12)(f)(ii)(C) to eliminate language directing permittees to ARM 17.30.1344, because the discharge limitations requiring 24-hour reporting are not contained in ARM 17.30.1344, and to renumber (12)(f)(ii)(C) to (24)(c). This provision requires permittees to report exceedances or violations, within 24 hours, of maximum daily discharge limitations for pollutants, which are listed by the department in an MPDES permit. 40 CFR 122.44(g) places the burden on the department to list those pollutants in an MPDES permit for which this 24-hour reporting requirement must be required. ARM 17.30.1344 adopts by reference 40 CFR 122.44(g). For clarification, the board is proposing text which points the permittee directly to 40 CFR 122.44(g). The board is also proposing to amend current (12)(f)(iii) and (12)(g) to correct internal references and to renumber (12)(f)(iii) to (25) and (12)(g) to (26).

The board is proposing to make minor amendments to current (13)(a), regarding bypass reporting requirements, to make editorial changes, correct formatting, correct internal references, and to renumber (13)(a) to (28). Bypass is the intentional diversion of waste streams from any portion of a treatment facility, as defined in ARM 17.30.1303 and 40 CFR 122.41(m). These proposed changes are necessary to maintain consistency with 40 CFR 122.41(m). The board is proposing a new (29) to describe the department's bypass notification requirements. Current (13)(b), renumbered (29)(a), is proposed to be amended to provide notification requirements for anticipated bypass. New (29)(b) is being proposed to provide notification requirements for unanticipated bypass. These amendments are being proposed to make the rule consistent with the federal rule. The board is also proposing to amend current (13)(c)(iii) and (13)(d) to correct internal cross references and to renumber (13)(c)(iii) to (30)(c) and (13)(d) to (31).

The board is proposing to make minor amendments to current (14)(a) regarding upset requirements to make editorial changes, correct formatting, and to renumber (14)(a) to (32). An upset occurs when there is unintentional and temporary noncompliance with technology-based effluent limitations due to factors beyond the reasonable control of the permitee and is defined in ARM 17.30.1303 and 40 CFR 122.41(n). These changes are necessary to maintain consistency with 40 CFR 122.41(n) and to correct formatting.

In (33)(c), the board is proposing to reference the general 24-hour notice provision for permit non-compliance.

The board is proposing to incorporate and update all applicable federal rules necessary to support the provisions in ARM 17.20.1342 in proposed amendments to current (15), which is proposed to be renumbered (35). These amendments will also correct formatting and provide consistency with other MPDES rules. The proposed amendments to current (15)(a), proposed to be renumbered (35)(a), incorporate the most recent federal guidelines establishing testing procedures for the analysis of pollutants as given in 40 CFR 136 and the proposed amendments to current (15)(b), proposed to be renumbered (35)(b), clarify the notification requirements for permittees under this rule. The board is further proposing to add a new (35)(c) incorporating 40 CFR 503.8, which addresses additional analytical methods for sewage sludge and new (35)(d), which incorporates analytical methods that are assigned to specific technology-based limitations in 40 CFR subchapter N. The board has adopted federal technology based effluent limitations as permit requirements in ARM 17.30.1207.

4. The rules proposed for repeal are as follows:

<u>17.30.1110 APPLICATION PROCEDURES: GENERAL</u> (AUTH: 75-5-201, 75-5-401, MCA; <u>IMP</u>, 75-5-401, MCA), located at pages 17-2871 and 17-2872, Administrative Rules of Montana. The board is proposing to repeal ARM 17.30.1110, which sets forth application procedures for storm water discharges other than storm water discharges associated with construction activity. This rule is no longer necessary because application procedures for all individual MPDES permits, including storm water, are found in ARM 17.30.1322. The procedures for issuing and administering MPDES general permits, including storm water general permits, are found in ARM 17.30.1341, as amended. These procedures require filing a notice of intent for coverage under a general permit and are common to all general permits issued under the MPDES rules. ARM 17.30.1322 and 17.30.1341 are equivalent to federal regulations set forth at 40 CFR 122.21, 122.26(c), for individual permits, and 122.28, for general permits. Repeal of ARM 17.30.1110 will eliminate duplication and potential conflicts between this rule and other rules adopted by the board in ARM Title 17, chapter 30, subchapters 11 through 13 and provide a uniform system for the administration of general permits.

17.30.1115 NOTICE OF INTENT PROCEDURES: CONSTRUCTION ACTIVITY (AUTH: 75-5-201, 75-5-401, MCA; IMP, 75-5-401, MCA), located at pages 17-2883 and 17-2884, Administrative Rules of Montana. The board is proposing to repeal ARM 17.30.1115, which sets forth application procedures for construction activity. This rule is no longer necessary because application procedures for all MPDES individual permits, including storm water, are found in ARM 17.30.1322. The procedures for issuing and administering MPDES general permits, including procedures for filing a notice of intent for coverage under a general permit, are found in ARM 17.30.1341, as amended. ARM 17.30.1322 and 17.30.1341 are equivalent to federal regulations set forth at 40 CFR 122.21, 122.26(c), for individual permits, and 122.28, for general permits. Repeal of ARM 17.30.1115 will eliminate duplication and potential conflicts between this rule and other rules adopted by the board in ARM Title 17, chapter 30, subchapters 11 through 13 and provide a uniform system for the administration of general permits.

17.30.1117 TRANSFER OF PERMIT COVERAGE (AUTH: 75-5-201, 75-5-401, MCA; IMP, 75-5-401, MCA), located at page 17-2884.4, Administrative Rules of Montana. The board is proposing to repeal ARM 17.30.1117, which sets forth procedures for transferring permit coverage for storm water discharges regulated under subchapter 11. This rule is not necessary because storm water permits are MPDES permits and may be transferred in accordance with the applicable provisions of ARM Title 17, chapter 30, subchapter 13, specifically ARM 17.30.1360. Repeal of ARM 17.30.1117 will eliminate duplication and potential conflicts between this rule and other rules adopted by the board in ARM Title 17, chapter 30, subchapters 11 through 13 and provide a uniform system for the administration of general permits.

5. Concerned persons may submit their data, views, or arguments, either orally or in writing, at the hearing. Written data, views, or arguments may also be submitted to Elois Johnson, Paralegal, Department of Environmental Quality, 1520 E. Sixth Avenue, P.O. Box 200901, Helena, Montana 59620-0901; faxed to (406) 444-4386; or e-mailed to ejohnson@mt.gov, no later than 5:00 p.m., ___________, 2014. To be guaranteed consideration, mailed comments must be postmarked on or before that date.

6. The attorney for the board, or another attorney for the Agency Legal Services Bureau, has been designated to preside over and conduct the hearing.

7. The board maintains a list of interested persons who wish to receive notices of rulemaking actions proposed by this agency. Persons who wish to have their name added to the list shall make a written request that includes the name, e-mail, and mailing address of the person to receive notices and specifies that the person wishes to receive notices regarding: air quality; hazardous waste/waste oil; asbestos control; water/wastewater treatment plant operator certification; solid waste; junk vehicles; infectious waste; public water supply; public sewage systems regulation; hard rock (metal) mine reclamation; major facility siting; opencut mine reclamation; strip mine reclamation; subdivisions; renewable energy grants/loans;

wastewater treatment or safe drinking water revolving grants and loans; water quality; CECRA; underground/above ground storage tanks; MEPA; or general procedural rules other than MEPA. Notices will be sent by e-mail unless a mailing preference is noted in the request. Such written request may be mailed or delivered to Elois Johnson, Paralegal, Department of Environmental Quality, 1520 E. Sixth Ave., P.O. Box 200901, Helena, Montana 59620-0901, faxed to the office at (406) 444-4386, e-mailed to Elois Johnson at ejohnson@mt.gov, or may be made by completing a request form at any rules hearing held by the board.

8. The bill sponsor contact requirements of 2-4-302, MCA, do not apply.

9. With regard to the requirements of 2-4-111, MCA, the board has determined that the amendment of the above-referenced rules will not significantly and directly impact small businesses.

Reviewed by: BOARD OF ENVIRONMENTAL REVIEW

___________________________ BY: ________________________________
JOHN F. NORTH ROBIN SHROPSHIRE
Rule Reviewer Chairman

Certified to the Secretary of State, ____________________, 2014.

BOARD OF ENVIRONMENTAL REVIEW
AGENDA ITEM
EXECUTIVE SUMMARY FOR RULE ADOPTION

AGENDA # III.B.1

AGENDA ITEM SUMMARY - The department requests that the board adopt the proposed amendments to the public water and sewage rules to:

1. Adopt by reference proposed Subdivision Rules ARM 17.36.320 through 17.36.323 and 17.36.325;
2. Amend existing public sewage rules to remove the adoption by reference to Subdivision rule ARM 17.36.327;
3. Amend existing public sewage rules establishing requirements for professional engineer submission of onsite sewage treatment systems;
4. Amend existing public water supply rules to provide fee structure consistency for review of public water supply systems that correspond to the proposed changes to Department Circular DEQ-1, 2014 edition which sets forth the requirements for the design and preparation of plans and specifications for public water supply systems;
5. Amend existing public sewage rules to provide fee structure consistency for review of public sewage systems that correspond to the adopted changes to Department Circular DEQ-4, 2013 edition which sets forth standards for the design and preparation of plans and specifications for subsurface wastewater treatment systems;
6. Amend existing public water supply rules to provide fee structure consistency for review of public water supply systems that correspond to the proposed changes to Department Circular DEQ-10, 2014 edition which sets forth the standards for development of springs to serve public water supply systems;
7. Amend existing public water supply rules to provide fee structure consistency for review of public water systems that correspond to proposed Department Circular DEQ-16, 2014 edition which sets forth standards for cisterns to serve non-community public water supply systems;
8. Amend Title 17, Chapter 36, Subchapter 9, On-Site Subsurface Wastewater Treatment Systems by updating definitions to provide consistency between the Subdivision Rules in Title 17, Chapter 36 and Department Circular DEQ-4, 2013 edition; and
9. Amend Title 17, Chapter 36, Subchapter 9, On-Site Subsurface Wastewater Treatment Systems by updating the minimum setback distances provided in Table 1 to provide consistency with the Subdivision Rules in Title 17, Chapter 36.

LIST OF AFFECTED RULES - ARM 17.36.320 through 325, 17.36.912, 17.36.918, and 17.38.101

AFFECTED PARTIES SUMMARY - The proposed rule amendments will affect designers and owners of systems that discharge sewage to subsurface treatment systems, and local boards of health and health departments that have regulations for such systems

SCOPE OF PROPOSED PROCEEDING - The board is considering final action on adoption of amendments to the above-referenced rules as proposed in the Montana Administrative Register.

BACKGROUND - The legislature requires the Board of Environmental Review to adopt rules related to the review and approval of subsurface sewage treatment systems.

Title 17, Chapter 36, Sub-Chapter 9, On-Site Subsurface Wastewater Treatment Systems, are board rules for the state minimum standards used by local health departments to permit onsite septic systems under Title 50, Chapter 2. The proposed revisions to the On-Site Subsurface Wastewater Treatment Systems rules update definitions and setback requirements to provide consistency between the proposed subdivision rules, Department Circular DEQ-4 and the state minimum standards.

Title 17 Chapter 38, Sub-chapter 1 Public Water and Sewage System Requirements, are board rules outlining the requirements for public sewage treatment systems. The Public Water and Sewage System Requirements adopt several Subdivision rules by reference for subsurface sewage treatment systems. The proposed Subdivision rule revisions outline allowable new and replacement system types, discuss site evaluation requirements and provide minimum setback requirements applicable in both proposed subdivisions and for public wastewater treatment systems not part of a subdivision. The proposed Public Water and Sewage System Requirements also outline the requirements for plan submission by a professional engineer and the fees for review of all public onsite sewage treatment systems.

In addition, the legislature requires the Board of Environmental Review to adopt rules related to the review and approval of public water supply systems. This includes the fees necessary for the review of public water supply plans and specifications.

HEARING INFORMATION - Katherine Orr conducted a public hearing on May 19, 2014, on the proposed amendments. The Presiding Officer's Report and the draft Notice of Amendment are attached to this executive summary. Draft responses to comments received are incorporated into the proposed notice.

BOARD OPTIONS - The board may:

1. Adopt the proposed amendments as set forth in the attached Notice of Public Hearing on Proposed Amendment;
2. Adopt the proposed amendments with revisions that the board finds are appropriate and that are consistent with the scope of the Notice of Public Hearing on Proposed Amendment and the record in this proceeding; or
3. Decide not to adopt the amendments.

DEQ RECOMMENDATION - The department recommends adoption of the proposed amendments as set forth in the attached Notice of Public Hearing on Proposed Amendment.

ENCLOSURES -

1. Notice of Public Hearing on Proposed Amendment
2. Presiding Officer's Report
3. HB521 and 311 Analysis
4. Small Business Impact Analysis
5. Public Comments
6. Draft Notice of Amendment

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
AND THE DEPARTMENT OF ENVIRONMENTAL QUALITY
OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.36.320, 17.36.321, 17.36.322, 17.36.323, 17.36.325, 17.36.912, 17.36.918, 17.38.101, and 17.38.106 pertaining to sewage systems, definitions, horizontal setbacks, floodplains, plans for public sewage system, and fees	NOTICE OF PUBLIC HEARING ON PROPOSED AMENDMENT (SUBDIVISIONS/ON-SITE SUBSURFACE WASTEWATER TREATMENT) (PUBLIC WATER AND SEWAGE SYSTEM REQUIREMENTS)

TO: All Concerned Persons

1. On May 19, 2014, at 1:30 p.m., the Board of Environmental Review and the Department of Environmental Quality will hold a public hearing in Room 111, Metcalf Building, 1520 East Sixth Avenue, Helena, Montana, to consider the proposed amendment of the above-stated rules.

2. The board and department will make reasonable accommodations for persons with disabilities who wish to participate in this public hearing or need an alternative accessible format of this notice. If you require an accommodation, contact Elois Johnson, Paralegal, no later than 5:00 p.m., May 5, 2014, to advise us of the nature of the accommodation that you need. Please contact Elois Johnson at Department of Environmental Quality, P.O. Box 200901, Helena, Montana 59620-0901; phone (406) 444-2630; fax (406) 444-4386; or e-mail ejohnson@mt.gov.

3. The rules proposed to be amended provide as follows, stricken matter interlined, new matter underlined:

17.36.320 SEWAGE SYSTEMS: DESIGN AND CONSTRUCTION (1) All components of ~~subsurface~~ sewage treatment systems must be designed and installed in accordance with ~~d~~Department Circular DEQ-4, Department Circular DEQ-2, or other applicable department circular and are subject to the following restrictions~~.~~:

(a) systems designed in accordance with Department Circular DEQ-2 may not be used for individual, shared, or multiple-user systems, except as provided in Department Circular DEQ-4; and

(b) experimental systems are allowed only pursuant to a waiver granted in accordance with ARM 17.36.601.

(2) ~~As indicated on Table 2 of this rule, public systems and multi-~~ Multiple-user systems with design flows greater than or equal to 2500 gallons per day must be designed by a ~~registered~~ professional engineer and are subject to the requirements in [New Rule II, proposed in MAR Notice No. 17-358 published in this register].

~~(2)~~ (3) ~~A~~ For subsurface systems, a minimum separation of at least four feet of natural soil must exist between the infiltrative surface or the liner of a lined system and a limiting layer, except that at least six feet of natural soil must exist on a ~~steep~~ slope ~~(~~of greater than 15~~%~~ percent ~~to 25%)~~.

~~(3)~~ (4) The proposed subsurface sewage treatment area must include an area for 100~~%~~ percent replacement of the system, except that the replacement area for elevated sand mounds may be allowed as provided in Department Circular DEQ-4. If a size reduction is approved for a system, the replacement area must have area sufficient for the system without the size reduction. Unless a waiver is approved by the department pursuant to ARM 17.36.601, the replacement area must meet the same requirements as the primary area. If the replacement area is not immediately adjacent to the primary area, or if ~~the department indicates to the applicant that it has reason to believe~~ there is evidence that site conditions for the replacement area may vary from those for the primary area, the applicant shall submit adequate evidence of the suitability of the replacement area.

~~TABLE 2~~
~~ALLOWABLE SYSTEMS, REQUIREMENTS~~

	~~YES - Systems that are allowed~~ ~~NO - Systems that are not allowed~~			
~~DEQ-4 System~~	~~Public: > 5000 gpd (1) (7)~~	~~Public or Multiple-user: ≥ 2500 gpd and ≤ 5000 gpd (2) (7)~~	~~Public or Multiple-user: < 2500 gpd (3)~~	~~Individual/ Shared: (6)~~
~~Standard Absorption Trench~~	~~NO~~	~~NO~~	~~YES~~	~~YES~~
~~At-Grade Systems~~	~~NO~~	~~NO~~	~~YES~~	~~YES~~
~~Gravelless~~	~~YES~~	~~YES~~	~~YES~~	~~YES~~
~~Deep Trench~~	~~NO~~	~~NO~~	~~NO~~	~~YES~~
~~Elevated Sand Mound~~	~~YES~~	~~YES~~	~~YES~~	~~YES~~
~~Evapotranspiration (ET) Systems~~	~~NO~~	~~NO~~	~~NO~~	~~NO (5)~~
~~ET-Absorption~~	~~NO~~	~~YES~~	~~YES~~	~~YES~~

Intermittent Sand Filters	YES	YES	YES	YES
Recirculating Sand Filters	YES	YES	YES	YES
Recirculating Trickling Filters	YES	YES	YES	YES

	YES - Systems that are allowed NO - Systems that are not allowed			
DEQ-4 System	Public: > 5000 gpd (1)	Public or Multiple-user: ≥ 2500 gpd and ≤ 5000 gpd (2)	Public or Multiple-user: < 2500 gpd (3)	Individual/ Shared: (6)
Chemical Nutrient Reduction; Aerobic Sewage Treatment Systems	NO (5)	NO (5)	NO (5)	NO (4)(5)
Pressure Distribution	YES	YES	YES	YES
Sand-lined Absorption Trenches	NO	YES	YES	YES
Experimental Systems	NO (5)	NO (5)	NO (5)	NO (5)

(1) Public systems with design flow greater than 5000 gallons per day (gpd).

(2) Public or multiple-user systems with design flow greater than or equal to 2500 gpd and less than or equal to 5000 gpd.

(3) Public or multiple-user systems with design flow less than 2500 gpd.

(4) Means of securing continuous operation and maintenance of these systems must be approved by the reviewing authority prior to DEQ approval.

(5) May be allowed by waiver, pursuant to ARM 17.36.601.

(6) Individual or shared commercial sewage systems that have a design flow greater than 700 gpd shall be considered multi-user.

(7) Must be designed by a professional engineer.

AUTH: 76-4-104, MCA
IMP: 76-4-104, MCA

REASON: The department is proposing to eliminate Table 2 and replace it with a narrative format. Table 2 shows sewage systems that are allowed by DEQ-4, but the systems currently listed in Table 2 do not include all of the systems addressed in the most recent edition (2013) of the Circular. Table 2 also adds some restrictions and requirements for Department Circular DEQ-4 (DEQ-4) systems. The department is proposing to eliminate some of these additional restrictions. With the proposed elimination of some of the restrictions in Table 2, and because Table 2 otherwise simply lists systems allowed by DEQ-4, it has limited use. The restrictions and requirements that are retained are proposed to be set out in a narrative format that is easier to understand.

The department is proposing to eliminate the restrictions imposed by Table 2 on standard absorption trenches, at-grade systems, deep trenches, evapotranspiration (ET) systems, ET-absorption systems, and chemical nutrient reduction and aerobic sewage treatment systems. The restrictions are not necessary because, if the systems are designed in accordance with DEQ-4, they will provide adequate treatment of wastewater. The proposed amendments would retain the restriction in Table 2 that experimental systems may be allowed only through a waiver. The amendments also would retain the requirement that multiple-user systems with a design flow greater than or equal to 2500 gallons per day be designed by a professional engineer. The amendments require that multiple-user systems designed by a professional engineer comply with the requirements of New Rule II, proposed in MAR Notice No. 17-358 and published in this register.

ARM 17.36.320(1) requires that components of sewage systems be designed in accordance with DEQ-4. The proposed amendments would delete the term "subsurface." This is necessary because DEQ-4 is not limited to subsurface systems. DEQ-4 also addresses systems such as waste segregation and incinerator toilets. The proposed amendments also add a reference to Department Circular DEQ-2 (DEQ-2). This is necessary because DEQ-2 requirements may be applicable to some public sewage systems.

Proposed ARM 17.36.320(1)(a) prohibits use of DEQ-2 for individual, shared, and multiple-user systems, except as provided in DEQ-4. A similar restriction currently exists in ARM 17.36.321(2), and it is proposed to be restated here for clarity. Because DEQ-4 requires some components to be designed in accordance with DEQ-2, the amendments will allow use of DEQ-2 when required by DEQ-4.

Proposed ARM 17.36.320(1)(b) sets out the requirement, currently in Table 2, that experimental systems are allowed only pursuant to a waiver.

The proposed amendments create a new ARM 17.36.320(2) to state the existing requirement that a professional engineer design multiple-user systems with a design capacity equal to or greater than 2500 gallons per day. The amendments delete the reference in this sentence to public systems. The provisions requiring design by professional engineers of public sewage systems will now be consolidated in the rules for public water and sewer systems. See proposed amendments to ARM 17.38.101. The amendments delete the reference to a "registered" professional engineer. The term "registered" is not necessary because "professional engineer" is proposed to be defined, in proposed amendments to the department's Sanitation in Subdivisions Act rules, as a person licensed pursuant to Title 37, chapter 67, MCA. This definition already appears in the public water supply rules.

See ARM 17.38.101(3)(m). The proposed amendments provide a cross-reference to the requirements in New Rule II, proposed in MAR Notice No. 17-358 and published in this register, for engineer-designed multiple-user systems. New Rule II, proposed in MAR Notice No. 17-358 and published in this register, requires the applicant to commit to retaining a professional engineer to certify that construction was completed in accordance with the approved design and requires that an engineer certify, before the system is operated, that it was completed in accordance with approved plans. It also requires an engineer to submit to the department, within 90 days after completion, certified "as-built" plans, and requires that plans and specifications be re-submitted if construction is not completed within three years after approval.

The proposed amendments to renumbered ARM 17.36.320(3) clarify that this section is applicable only to subsurface systems. It is not necessary to apply the requirements of this section to systems not addressed in DEQ-4. The proposed amendment eliminates the 25 percent maximum. Under proposed ARM 17.36.322(2), slopes of up to 35 percent are allowed with a variance and there is no need to state a maximum in this rule. This amendment follows proposed amendments to ARM 17.36.322 that would allow pressure-dosed systems on slopes up to 35 percent through a waiver process. This amendment is necessary to clarify that, if a waiver is granted under ARM 17.36.322 to allow a pressure-dosed system on a slope greater than 25 percent, the six-foot soil requirement applies.

The proposed amendments to renumbered ARM 17.36.320(4) clarify that the reviewing authority has discretion whether to require replacement areas for elevated sand mounds, pursuant to DEQ-4. See DEQ-4 Section 6.7.2.5. The amendments also provide that a replacement area must provide space for a full-size system, even when the original approved system qualified for a size reduction. This is necessary to ensure adequate space in the event that the replacement system does not qualify for a size reduction. The amendments also make minor changes for clarification.

17.36.321 SEWAGE SYSTEMS: ALLOWABLE NEW AND REPLACEMENT SYSTEMS (1) ~~The allowable new sewage treatment systems, together with certain other requirements for such systems, are indicated in Table 2 of ARM 17.36.320.~~ All systems must be designed and installed in accordance with ~~d~~Department Circular DEQ-4, Department Circular DEQ-2, or other applicable department circular. ~~The use of sewage systems for replacement systems shall be in accordance with department Circular DEQ-4.~~ Requirements applicable to review of existing sewage treatment systems are set out in ARM 17.36.327.

(2) Systems designed in accordance with ~~d~~Department Circular DEQ-2, may not be used for individual, shared, or multiple-user systems, except as provided in Department Circular DEQ-4.

(3) The following sewage systems may not be used for new systems:

(a) through (f) remain the same.

(g) holding tanks, except that~~.~~:

(i) ~~T~~the department may grant a waiver, pursuant to ARM 17.36.601, to allow holding tanks ~~for recreational vehicle dump stations~~ in facilities owned and operated by a local, state, or federal unit of government, or in facilities licensed by the Department of Public Health and Human Services and inspected by the local health department. Holding tanks must be designed and maintained in accordance with the requirements in ~~d~~Department Circular DEQ-4 and all other requirements imposed by the department and local health department~~.~~; and

(ii) the department may grant a waiver, pursuant to ARM 17.36.601 and with concurrence by the local health department, to allow holding tanks to replace a failed system when no other alternative that meets these rules is reasonably available.

(4) through (5) remain the same.

AUTH: 76-4-104, MCA
IMP: 76-4-104, MCA

REASON: The proposed amendments to ARM 17.36.321(1) delete the reference to Table 2 in ARM 17.36.320. This is necessary because the proposed amendments to ARM 17.36.320 would delete Table 2. The proposed amendments would also add a reference to DEQ-2. This is necessary because DEQ-2 requirements may be applicable to some sewage systems. The amendments would delete the sentence identifying requirements for replacement systems. The sentence is unnecessary because the preceding sentence identifies requirements for "all systems," which include replacement systems.

ARM 17.36.321(2) prohibits use of DEQ-2 for individual, shared, and multiple-user systems. The proposed amendment clarifies that DEQ-2 requirements may apply in some cases, as specified in DEQ-4.

ARM 17.36.321(3)(g)(i) allows the department to allow, through waiver, holding tanks for recreational vehicle dump stations in facilities owned and operated by a local, state, or federal unit of government, or in facilities licensed by the Department of Public Health and Human Services (DPHHS). The proposed amendment would also allow waivers for holding tanks in other types of government-owned or licensed facilities. It is not necessary to limit waivers under this section to recreational vehicle dump stations.

The proposed amendments add a new ARM 17.36.321(3)(g)(ii), which allows the department to allow, through waiver, holding tanks in any situation where a system has failed and no other alternative that meets the rules is reasonably available. The new provision is necessary to allow for continued use of a parcel when the existing sewage system has failed and cannot be replaced with any system other than a holding tank.

17.36.322 SEWAGE SYSTEMS: SITING (1) ~~Subsurface~~ Gravity-fed subsurface sewage treatment systems may not be used if natural slopes are greater than 15~~%~~ percent~~;~~. ~~however, the department may, by waiver granted pursuant to ARM 17.36.601, allow a~~ A pressure-dosed sewage treatment system with a design flow of 5000 gallons per day or less may be used on slopes ~~between~~ greater than 15~~%~~ percent and up to 25~~%~~ percent, if a ~~registered~~ professional engineer or a person qualified to evaluate and identify soil in accordance with ~~ASTM standard D5921-96el (Standard Practice for Subsurface Site Characterization of Test Pits for On-Site Septic Systems)~~ Department Circular DEQ-4 submits adequate evidence that there will be no visible outflow of liquid downslope from the subsurface sewage treatment system.

(2) The department may grant a waiver, pursuant to ARM 17.36.601 and after consultation with the local health department, to allow pressure-dosed subsurface sewage treatment systems on slopes greater than 25 percent and up to 35 percent if a professional engineer or a person qualified to evaluate and identify soil in accordance with Department Circular DEQ-4 submits adequate evidence that there will be no visible outflow of liquid downslope from the subsurface sewage treatment system.

~~(2)~~ (3) Subsurface sewage treatment systems may not be installed on unstable landforms, as defined in ARM ~~17.36.320~~ 17.36.101.

(3) and (4) remain the same, but are renumbered (4) and (5).

~~(5)~~ (6) For lots ~~one~~ two acres in size or less, the applicant shall physically identify the drainfield location by staking or other acceptable means of identification. For lots greater than ~~one~~ two acres in size, the department may require the applicant to physically identify the drainfield location.

(6) remains the same, but is renumbered (7).

AUTH: 76-4-104, MCA
IMP: 76-4-104, MCA

REASON: The proposed amendments delete the reference to a "registered" professional engineer. See Reason for ARM 17.36.320. The proposed amendments to ARM 17.36.322(1) retain the 15 percent slope limitation for gravity-fed subsurface systems, and allow, without a waiver, pressure-dosed systems on slopes greater than 15 percent and up to 25 percent if a qualified person performs a soil evaluation. Gravity-fed systems are not suitable on slopes greater than 15 percent due to the tendency of these systems to load effluent over small areas, which creates the potential for soil sloughing or effluent outfall. However, pressure-dosed systems can be used on those slopes, and the waiver process is not needed to ensure that the pressure-dosed systems are properly designed. For slopes greater than 15 percent and up to 25 percent, the amendments require that soil evaluations be conducted in accordance with DEQ-4 instead of ASTM standard D5921-96el. The reference to the ASTM standard is not necessary because the procedures in the standard are substantially addressed in DEQ-4.

The proposed new ARM 17.36.322(2) allows, through a department waiver, use of pressure-dosed systems on slopes greater than 25 percent and up to 35 percent, if a qualified person performs a soil evaluation. The department has found that in some situations pressure-dosed systems can be installed on these slopes without adverse consequences. The use of the waiver process will allow for consideration of the special circumstances in each case.

The proposed amendment to renumbered ARM 17.36.322(3) is necessary to correct an erroneous cross reference.

The proposed amendment to renumbered ARM 17.36.322(6) expands, from one to two acres, the size of lots in which approved drainfield locations must be staked or otherwise identified. This amendment is necessary to conform to revisions to DEQ-4, 2013 edition (Section 2.1.4.). Physical identification of approved drainfield sites is necessary to prevent other construction improvements from interfering with the drainfield site. Identification may be by physical staking, or by a method such as electronic identification using GPS coordinates. The increase in lot size from one to two acres is necessary because the potential for interference is not limited to one-acre lots. The amendments also give the department discretion to require drainfield site identification on lots larger than two acres. This is necessary to allow the reviewing authority to prevent interference with an approved drainfield site where a significant amount of ground disturbance is proposed.

17.36.323 ~~SEWAGE SYSTEMS: HORIZONTAL~~ SETBACKS~~; WAIVERS~~

(1) Minimum ~~horizontal~~ setback distances, ~~(~~in feet~~)~~, shown in Table ~~3~~ 2 of this rule must be maintained, except as provided in the table footnotes or as allowed through a deviation granted under ARM Title 17, chapter 38, subchapter 1. The setbacks in this rule are not applicable to gray water irrigation systems that meet the setbacks and other requirements of ARM 17.36.319.

~~(2) A waiver of the setback distance for a cistern may be granted by the department, pursuant to ARM 17.36.601, if the applicant demonstrates that the elevation of the cistern is higher than the elevation of the septic tank, other components, or drainfield/sand mound.~~

~~(3) A waiver of the setback distance between drainfields/sand mounds and surface waters, springs, and floodplains may be granted by the department, pursuant to ARM 17.36.601, only if:~~

~~(a) the applicant demonstrates that ground water flow at the drainfield site cannot flow into the surface water or spring; or~~

~~(b) the surface water or spring seasonally high water level is a minimum of 100 feet horizontal distance from the drainfield and the bottom of the drainfield will be at least two feet above floodplain elevation.~~

~~(4) The department may require more than 100 feet of separation from the floodplain or from surface water or springs if it determines that site conditions or water quality nondegradation requirements indicate a need for the greater distance.~~

TABLE ~~3~~ 2
SETBACK DISTANCES
(in feet)

From	To Drinking Water ~~Supply~~ Wells	To Sealed Components (1) and Other Components (2)	To Drainfields/~~Sand Mounds~~ Soil Absorption Systems
Public or multiple-user drinking water wells/springs	-	100 (3)	100
Individual and shared drinking water wells	-	50 (3)	100
Other wells (4)	-	50 (3)	100 (3)

Suction lines	-	50	100
Cisterns	-	25	50
Roadcuts, escarpment	-	10 ~~(3)~~ (5)	25
Slopes > ~~25%~~ 35 percent ~~(4)~~ (6)	-	10 ~~(3)~~ (5)	25
Property boundaries	10 (7)	10 (7)	10 (7)
Subsurface drains	-	10	10
Water ~~Lines~~ mains	-	10 (8)	10
Drainfields/~~Sand Mounds~~ Soil absorption systems	100	10	-
Foundation walls	-	10	10
Surface water (9), springs	100 ~~(5)~~ (3) (10) (11)	50 (3) (10)	100 (3) (10) (12)
Floodplains	10 (10)	- Sealed components - no setbacks (1) Other components - 100 (2) (3) (10)	100 (10) (13)
Mixing zones	100 (3)	-	-
Storm water ponds and ditches	25 (14)	10	25

(1) Sealed components include ~~sewer lines, sewer mains, septic tanks, grease traps, dosing tanks, and pumping chambers~~ holding tanks, sealed pit privies, and the components addressed in Department Circular DEQ-4, Chapters 4 and 5. Sealed components must meet the requirements of ARM 17.36.322(4).

(2) Other components include ~~intermittent and recirculating sand filters, package plants, and evapotranspiration systems~~ the components addressed in Department Circular DEQ-4, chapter 7.

(3) A waiver of this requirement may be granted by the department pursuant to ARM 17.36.601.

(4) Other wells include, but are not limited to, irrigation and stock watering, but do not include observation wells as addressed in Department Circular DEQ-4.

(3) remains the same, but is renumbered (5).

~~(4)~~ (6) Down-gradient of the sealed component, other component, or drainfield/~~sand mound~~ soil absorption system.

~~(5) A waiver of this requirement may be granted by the department pursuant to ARM 17.36.601.~~

(7) Easements may be used to satisfy the setback to property boundaries.

(8) Unless a waiver is granted by the department pursuant to ARM 17.36.601, sewer mains that cross water mains must be laid with a minimum vertical separation distance of 18 inches between the mains.

(9) For purposes of this rule, "surface water" does not include intermittent storm water.

(10) The department may require more separation from the floodplain or from surface water or springs if it determines that site conditions or water quality requirements indicate a need for the greater distance.

(11) Pursuant to ARM 17.36.331, the reviewing authority may require greater than a 100-foot horizontal separation between a well and surface water if there is a potential that the well may be influenced by contaminants in the surface water.

(12) A waiver may be granted by the department, pursuant to ARM 17.36.601, if the applicant demonstrates that ground water flow at the drainfield site cannot flow into the surface water or spring. The setback between drainfields or soil absorption systems to irrigation ditches does not apply if the ditch is lined with a full culvert.

(13) A waiver may be granted by the department, pursuant to ARM 17.36.601, if the applicant demonstrates that the surface water or spring seasonally high water level is at least a 100-foot horizontal distance from the drainfield and the bottom of the drainfield will be at least two feet above the maximum 100-year flood elevation.

(14) The setback is 100 feet for public wells, unless a deviation is granted under ARM Title 17, chapter 38, subchapter 1.

AUTH: 76-4-104, MCA
IMP: 76-4-104, MCA

REASON: The proposed amendment to the title of the rule deletes "Sewage Systems." This is necessary because the setbacks in Table 2 apply to other features besides sewage systems. The proposed amendment to the title also deletes the term "horizontal." This is necessary because proposed new footnote (8) to Table 2 establishes vertical setbacks between water and sewer mains.

The proposed amendments move ARM 17.36.323(2) through (4) into the Table 2 footnotes. The current format is confusing in that some allowable waivers are shown on Table 2 and others are not. These amendments will ensure that all allowable waivers are indicated on the table and described in the table footnotes. The proposed amendment to ARM 17.36.323(1) indicates that all waivers to the setbacks in Table 2 are shown in the footnotes. The proposed amendments to ARM 17.36.323(1) also allow a waiver to a setback in the table if the department has allowed a lesser distance through the deviation process under the public water and sewer (PWS) rules in ARM Title 17, chapter 38, subchapter 1 and related department circulars. This "reciprocal" waiver process is necessary to prevent a conflict between these rules and a deviation for a proposed subdivision facility that is granted under the PWS rules.

At the top of Table 2, column 4, the proposed amendments replace the term "sand mounds" with "soil absorption systems." This is necessary to clarify that the setback table applies to other systems besides sand mounds. The proposed amendments also replace "water supply wells" with "drinking water wells." This is necessary to clarify that the referenced setbacks apply only to water wells proposed to be used for human drinking water supply.

Existing footnotes (1) and (2) of Table 2 identify sealed and "other" components that are subject to the table. The proposed amendments to footnotes (1) and (2) delete the lists of components in the footnotes and replace them with a reference to DEQ-4, Chapters 4, 5, and 7. The components currently listed in the footnotes are addressed in DEQ-4, but DEQ-4 includes other components as well. It is not practical to list all of the components in the footnote. To provide a more complete identification of components that are subject to Table 2, it is necessary to identify them by reference.

In the first row of Table 2, the proposed amendments allow a waiver of the setback between public or multiple-user wells or springs and sealed or other components of sewage systems. A 100-foot setback is not always necessary when the sewage system component is designed to prevent contamination of the water supply. The current table allows waivers under footnote (5). The proposed amendments renumber the waiver footnote as footnote (3) throughout Table 2.

The proposed amendments insert a new second row in Table 2 for individual and shared water supply wells. The current table addresses these wells under "other wells." The new category is proposed in order to distinguish between drinking water wells and non-drinking water wells. Under the proposed amendment, setbacks to non-drinking water wells will be addressed under "other wells." The setbacks are the same for drinking water wells and other wells, except that a waiver is allowed for the setback between other wells and drainfields/soil absorption systems. Because other wells no longer include wells for drinking water, it is appropriate to adjust this setback in some cases through waiver. The proposed amendments would also allow a waiver of the setback between individual, shared, and other wells and sealed and "other" components of sewage systems. A 100-foot setback is not always necessary when the sewage system component is designed to prevent contamination of the water supply or other well. Proposed footnote (4) provides that the setbacks for other wells do not apply to monitoring wells. This is necessary to allow the use of monitoring wells in subdivisions. Compared with wells for irrigation or stockwater, monitoring wells do not present a significant risk of surfacing sewage, and in some cases monitoring wells must be installed close to a sewage source to determine potential impacts to water quality.

The proposed amendments to the setbacks for roadcuts, escarpments, and slopes greater than 25 percent renumber the existing footnote from (3) to (5). The amendments increase, from 25 percent to 35 percent, the slope to which the slope setback applies. This is necessary to be consistent with the proposed amendments to ARM 17.36.322, which allow, through waiver, pressure-dosed sewage treatment systems on slopes between 25 percent and 35 percent. The amendment also renumbers, from (4) to (6), the footnote that clarifies that the slope setback applies down-gradient of the sealed component, other component, or drainfield/soil absorption system.

The proposed amendments add a new footnote (7) to the 10-foot setback for property boundaries to provide that easements may be obtained to satisfy the setback. The purpose of the setback is to allow owners adequate access to their facilities for purposes of repairs and maintenance. In some cases, usually involving a change to a previously approved facility, the 10-foot buffer from the property boundary may be unavailable. In those cases, an easement from the adjoining landowner will provide adequate assurance that access is available.

The proposed amendments modify the current 10-foot setback for "water lines" so that it would apply only to "water mains." Ten feet of horizontal separation is not needed between sewage system components and water service lines. This amendment will also provide consistency with a comparable setback in the Uniform Plumbing Code. The proposed amendments add a new footnote (8) to the setback that requires an 18-inch vertical separation between water and sewer mains, unless the department grants a waiver. The 18-inch vertical separation requirement is currently found in Department Circulars DEQ-1 (DEQ-1) and DEQ-2 (DEQ-2), and is included in footnote (8) to ensure that subdivision applicants are aware of it. The waiver process will provide a method for considering special circumstances that may affect the need for the 18-inch vertical setback.

The proposed amendments add several new footnotes to the setbacks for surface water and springs. Footnote (9) provides that this setback is not applicable to intermittent storm water. Footnote (9) is added because the amendments add, in the last row of Table 2, a new setback for storm water ponds and ditches. The proposed amendments add footnote (3), which will allow waivers from the setbacks from surface water and springs. Special circumstances can affect whether these setbacks are necessary. The waiver process will provide a method for considering these circumstances on a case-by-case basis. Footnote (10) allows the department to require more separation from surface water or springs, based on site conditions or water quality needs. This footnote incorporates the provisions that are currently in (4) of the rule. Footnote (11) provides a cross-reference to ARM 17.36.331, which allows the reviewing authority to require a greater than 100-foot separation between a well and surface water if there is a potential that the well may be influenced by contaminants. Footnote (11) is necessary to indicate that the setback shown in Table 2 can be modified in those circumstances. Footnote (12) provides that the department may waive the drainfield setback if the applicant demonstrates that ground water flow at the drainfield site cannot flow into the surface water or springs. This footnote incorporates the provisions that are currently in (3)(b). Footnote (12) also states that the setback between drainfields or soil absorption systems and irrigation ditches does not apply if the ditch is lined with a full culvert. This provision reflects an existing department interpretation of former (3)(a). Including it in footnote (12) will provide guidance to applicants about this setback requirement.

The proposed amendments add several footnotes to the floodplain setbacks. The proposed amendments add footnote (3), which allows waivers, to the setback between the floodplain and wells. This is necessary to allow, through the waiver process, consideration of special construction or siting circumstances that minimize the potential for commingling between flood waters and a water supply. Footnote (10) provides that the reviewing authority may require more separation from the floodplain, based on site conditions or water quality needs. This footnote incorporates the provisions that are currently in (4) of the rule. Proposed footnote (13) provides that the department may waive the setback between floodplains and drainfields/soil absorption systems if the applicant demonstrates that the surface water or spring seasonally high water level is at least 100 feet horizontal distance from the drainfield and that the bottom of the drainfield will be at least two feet above the maximum flood elevation. This footnote incorporates the provisions that are currently in (3)(b) of the rule. The proposed amendments also add footnote (3), which allows waivers, to the setback between the flood plain and "other" sewage components. Under the proposed amendments to footnote (2), "other" sewage components are the advanced treatment systems addressed in chapter 7 of DEQ-4. Some of these systems are sealed units that would not create a contamination risk during a flood event. The waiver process will provide a method for considering these circumstances on a case-by-case basis.

The proposed amendments insert a new row in Table 2 establishing a 100-foot setback between mixing zones and water supply wells. This is necessary to ensure that drinking water wells are isolated from potential sources of contamination. A waiver provision is provided to allow for department consideration of unique circumstances.

The proposed amendments insert a new row in Table 2 establishing setbacks from storm water ponds and ditches. The proposed setbacks are less than those for non-storm surface water and springs. Because storm water facilities have intermittent flows, they are less likely to impact wells or be impacted by sewage disposal facilities. Consequently, it is not necessary to apply the larger setbacks that apply to more permanent surface water sources. Proposed footnote (14) clarifies that the setback remains 100 feet between storm water facilities and public wells. This is necessary to be consistent with the requirements for public wells set out in DEQ-1 and Department Circular DEQ-3 (DEQ-3). Section 3.2.3.1 of DEQ-1 and DEQ-3 requires that public wells be located at least 100 feet from sewer lines, septic tanks, holding tanks, and any structure used to convey or retain industrial, storm, or sanitary waste.

17.36.325 SEWAGE SYSTEMS: SITE EVALUATION (1) remains the same.

(2) If the applicant or the department has reason to believe that ground water will be within seven feet of the surface at any time of the year within the boundaries of the treatment system, ~~the applicant shall install ground water level observation pipes to a depth of at least eight feet to determine the seasonally high ground water level. The applicant shall monitor the observation pipes through the seasonally high ground water period~~ ground water monitoring must be conducted in accordance with Department Circular DEQ-4.

(3) The applicant shall provide descriptions of the soils within 25 feet of the boundaries of each proposed drainfield. ~~Soil descriptions must address the characteristics used in the U.S. Department of Agriculture's National Soil Survey Handbook (USDA, NRCS, September 1999), and the Soil Survey Manual (USDA, October 1993). These characteristics include, but are not limited to, soil texture, soil structure, soil consistence, and indicators of redoximorphic features.~~ Soil descriptions for the proposed subdivision must meet the following requirements:

(a) Soil descriptions must be done in accordance with Department Circular DEQ-4. The characteristics that must be addressed include, but are not limited to, soil texture, soil structure, soil consistence, and indicators of redoximorphic features.

~~(a)~~ (b) Soil descriptions ~~for the proposed subdivision~~ must be based on data obtained from test holes. Test holes must be ~~at least eight feet in depth~~ dug in accordance with Department Circular DEQ-4~~;~~. The number of test holes must be as provided in (c), unless a waiver is granted by the department pursuant to ARM

17.36.601. Before a waiver is granted, the applicant shall complete test holes for 25 percent of the proposed drainfield locations in the proposed subdivision, shall demonstrate that the soils are consistent throughout the area requested for a waiver, and shall obtain the approval of the local reviewing authority. The department may require additional test holes than are required in (c) if the department determines that there is significant variability of the soils in the proposed drainfield areas. Each test hole must be keyed by a number on a copy of the lot layout or map with the information provided in the application.

~~(b)~~ (c) At least one test hole must be dug for each individual drainfield and for each shared (two-user) drainfield~~, unless a waiver is approved by the department pursuant to ARM 17.36.601. Before a waiver is requested and granted, the applicant must complete test holes for 25% of the proposed drainfield locations in the subdivision, demonstrate that the soils are consistent throughout the area requested for a waiver, and must obtain the approval of the local reviewing authority for reduction in number of test holes~~. At least three test holes must be dug for each multiple-user and public drainfield~~, unless a waiver is approved by the department pursuant to ARM 17.36.601~~. At least one test hole must be dug ~~in~~ for each zone of a pressure-dosed drainfield~~, unless a waiver is approved by the department pursuant to ARM 17.36.601. The department shall require additional test holes if it determines that there is significant variability of the soils in the proposed drainfield area;~~.

~~(c) Test holes must be located within 25 feet of the boundaries of the proposed drainfield. The locations must be established by a person qualified to evaluate and identify soil in accordance with ASTM standard D5921-96el (Standard Practice for Subsurface Site Characterization of Test Pits for On-Site Septic Systems);~~

(d) If the applicant or the department has reason to believe that a limiting layer is within seven feet of the ground surface at the site of a proposed subsurface sewage treatment system~~s~~, the department may require additional test ~~pits~~ holes and soil descriptions sufficient to describe the suitability of the soil ~~must be provided; and~~.

~~(e) Each test hole must be keyed by a number on a copy of the lot layout or map with the information provided in the report.~~

(4) Sewage systems that are subject to the design requirements of Department Circular DEQ-2 must meet the siting requirements of that circular.

AUTH: 76-4-104, MCA
IMP: 76-4-104, MCA

REASON: The proposed amendment to ARM 17.36.325(2) deletes the existing description of required ground water monitoring procedures and replaces it with a reference to DEQ-4. DEQ-4 contains a more complete statement of procedures and the amendment is necessary to inform subdivision applicants of all applicable ground water monitoring procedures.

The proposed amendments to ARM 17.36.325(3) reorganize the section to consolidate the waiver provisions into a single subsection. This is necessary to eliminate repetition and to clearly indicate which requirements are subject to waiver. The proposed amendments add a reference to DEQ-4 to renumbered ARM

17.36.325(3)(b). DEQ-4 contains a more complete statement of test hole requirements, and the amendment is necessary to inform subdivision applicants of all applicable procedures. The amendment in new (c) is necessary to allow test holes to be dug near, but not in, the zone if disruption by the test hole could interfere with the function of the system. The amendment is also necessary to be consistent with procedures in DEQ-4, 2013 edition. The proposed amendments delete existing ARM 17.36.325(3)(c) because it unnecessarily duplicates other provisions in the rule. The amendment to (d) is proposed because additional holes and descriptions may not always be necessary in this situation. Subsection (e) is eliminated because this requirement will now be found in the new language in (b).

The proposed amendments add a reference to the siting requirements of DEQ-2. This is necessary to identify applicable siting requirements for sewage systems that are subject to DEQ-2.

17.36.912 DEFINITIONS For purposes of this subchapter, the following definitions apply:

(1) through (4) remain the same.

(5) "Commercial unit" means the area under one roof that is occupied by a business or other nonresidential use. A building housing two businesses is considered two commercial units.

(5) and (6) remain the same, but are renumbered (6) and (7).

~~(7) "Dwelling" or "residence" means any structure, building or portion thereof, which is intended or designed for human occupancy and supplied with water by a piped water system.~~

(8) and (9) remain the same.

(10) "Floodplain" means the area adjoining the watercourse or drainway that would be covered by ~~the floodwater of a flood of 100-year frequency except for sheet flood areas that receive less than one foot of water per occurrence and are considered zone b areas by the federal Emergency Management Agency~~ a flood that is expected to recur on the average of once every 100 years or by a flood that has a one percent chance of occurring in any given year. The floodplain consists of the floodway and the flood fringe, as defined in ARM Title 36, chapter 15.

(11) through (13) remain the same.

(14) "Impervious layer" means any layer of material in the soil profile that has a percolation rate slower than ~~120~~ 240 minutes per inch.

(15) "Individual wastewater system" means a wastewater system that serves one living unit or commercial ~~structure~~ unit. The ~~total number of people served may not exceed 24~~ term does not include a public sewage system as defined in 75-6-102, MCA.

(16) remains the same.

(17) "Living unit" means the area under one roof ~~occupied by a family~~ that can be used for one residential unit and which has facilities for sleeping, cooking, and sanitation. For example, a duplex is considered two living units.

(18) "Multiple-user wastewater system" means a ~~non-public~~ wastewater system that serves or is intended to serve ~~three through 14 living units or three through 14 commercial structures~~ more than two living units or commercial units or a combination, but which is not a public sewage system as defined in 75-6-102, MCA.

~~The total number of people served may not exceed 24.~~ In estimating the population that will be served by a proposed residential system, the reviewing authority shall multiply the number of living units ~~times the county average of persons per living unit based on the most recent census data~~ by 2.5.

(19) remains the same.

~~(20) "Package plants" means wastewater treatment systems that are sealed within a watertight container and contain components for the secondary and tertiary treatment of wastewater.~~

~~(21)~~ (20) "Percolation test" means a standardized test used to assess the infiltration rate of soils, performed in accordance with Appendix A in Department Circular DEQ-4.

~~(22)~~ (21) "Piped water ~~system~~ supply" means a plumbing system that conveys water into a structure from any source including, but not limited to, wells, cisterns, springs, or surface water.

(23) through (28) remain the same, but are renumbered (22) through (27).

~~(29)~~ (28) "Septic tank" means a ~~storage~~ wastewater settling tank in which settled sludge is in immediate contact with the wastewater flowing through the tank while the organic solids are decomposed by anaerobic action.

~~(30)~~ (29) "Shared wastewater system" means a wastewater system that serves or is intended to serve two living units or commercial ~~structures~~ units or a combination of both. The ~~total people served may not exceed 24~~ term does not include a public sewage system as defined in 75-6-102, MCA. ~~In estimating the population served, the reviewing authority shall multiply the number of living units times the county average of persons per living unit based on the most recent census data.~~

(31) and (32) remain the same, but are renumbered (30) and (31).

~~(33)~~ (32) "Soil profile" means a description of the soil strata to a depth of eight feet using the United States Department of Agriculture (USDA) soil classification system method in Appendix B, Department Circular DEQ-4.

(34) and (35) remain the same, but are renumbered (33) and (34).

~~(36)~~ (35) "Wastewater" means water-carried ~~waste that is discharged from a dwelling, building, or other facility, including~~ wastes including, but not limited to:

(a) through (d) remain the same.

~~(37)~~ (36) "Wastewater treatment system" or "wastewater disposal system" means a system that receives wastewater for purposes of treatment, storage, or disposal. The term includes, but is not limited to, ~~pit privies and experimental systems~~ all disposal methods described in Department Circular DEQ-4.

AUTH: 75-5-201, MCA
IMP: 75-5-305, MCA

REASON: The term "commercial unit" is defined in new ARM 17.36.912(5). The term is used in the definitions of individual, shared, and multiple-user wastewater systems. The proposed definition of "commercial unit" is the same as the definition in these rules and DEQ-4, 2013 edition. The definition is necessary to clarify how shared and multiple-user systems are defined.

The proposed amendments delete the definition of "dwelling." The term "dwelling" is currently used only in the definition of "wastewater" to refer to wastewater discharged from a dwelling. The proposed amendments would modify the definition of "wastewater" to delete the reference to discharge from a dwelling. Consequently, the definition of "dwelling" is no longer necessary.

The proposed amendments to the definition of "floodplain" in ARM 17.36.912(10) eliminate the exception for areas that receive less than one foot of water per occurrence that are considered "zone b" areas by the Federal Emergency Management Agency (FEMA). The defined term "floodplain" is used in rules that restrict the construction of drainfields in and near floodplains. The exception for FEMA "zone b" in the current definition could allow construction of drainfields in areas that are inundated by floodwaters less than one foot deep during the 100-year flood. Because any inundation of drainfields by flood waters during a 100-year flood could interfere with proper drainfield operation, it is necessary to eliminate the exception, in the definition of "floodplain," for FEMA "zone b" areas.

The proposed amendments to the definition of "impervious layer" in ARM 17.36.912(14) change, from 120 to 240 minutes per inch, the percolation rate at which material is considered impervious. The amendment conforms this definition to that in DEQ-4, 2013 edition, and is necessary because adequate wastewater treatment can be achieved in soils with slower percolation rates.

The proposed amendments to the definition of "individual wastewater system" in ARM 17.36.912(15) replace the term "commercial structure" with "commercial unit." This is necessary in order to use the term "commercial unit" as defined in these rules and in DEQ-4, 2013 edition. The amendments also delete the limitation to 24 people served, and replace it with a reference to the statutory definition of public water supply and public sewage systems. This amendment is necessary because the 24-person limit does not accurately identify the threshold between a non-public and a public system contained in 75-6-102, MCA.

The proposed amendment to the definition of "living unit" in ARM 17.36.912(17) deletes the reference to "family" and replaces it with "residential." This is necessary because not all residential uses involve use by a family. The amendments also identify the basic features of a living unit, which are that it has facilities for sleeping, cooking, and sanitation. The amendments conform this definition to that in the Sanitation in Subdivisions Act rules and DEQ-4 and are necessary to identify which structures constitute living units for the purposes of these rules.

The proposed amendments to the definition of "multiple-user wastewater system" in ARM 17.36.912(18) replace the term "commercial structure" with "commercial unit." This is necessary in order to use the term "commercial unit" as defined in these rules and in DEQ-4, 2013 edition. The proposed amendments provide that multiple-user systems can consist of two or more living units, commercial units, or a combination of residential and commercial units. This is necessary to provide guidance about the meaning of the rules. The amendments also delete the limitation to 24 people served and replace it with a reference to the statutory definition of public water supply and public sewage systems. This amendment is necessary because the 24-person limit does not accurately identify the threshold between a non-public and a public system. The amendments also modify the formula for determining when proposed residential water and sewer systems will be subject to the requirements for public systems. The current rule multiplies the number of proposed living units times the county average of persons per living unit, based on the most recent census data. The amendments standardize the persons per living unit to 2.5. This is necessary to ensure that the requirements for public systems are applied consistently across the state to developments of a certain size.

The proposed amendments delete the definition of "package plants" in ARM 17.36.912(20). The term is used in a list of sewage system components in footnote (2) of the setback table in ARM 17.36.918. Because the proposed amendments delete the term from the footnote to the setback table, this definition is no longer necessary.

The proposed amendment to the definition of "percolation test" in ARM 17.36.912(21) references the procedures for performing percolation tests set out in DEQ-4 Appendix A. This amendment conforms to the definition in DEQ-4 and is necessary to clarify that tests must be done in accordance with Appendix A to meet the requirements of these rules.

The proposed amendments modify the definition of "piped water system" in ARM 17.36.912(22). This is necessary because the term "piped water system" is used only in the definition of "dwelling," which the proposed amendments would replace with the term "living unit." The modification replaces the term with "piped water supply," which is used in ARM 17.36.916(6).

The proposed amendments to the definition of "septic tank" in ARM 17.36.912(29) make minor changes for clarification and are necessary to conform to the definition in DEQ-4, 2013 edition.

The proposed amendments to the definition of "shared wastewater system" in ARM 17.36.912(30) replace the term "commercial structure" with "commercial unit." This is necessary in order to use the term "commercial unit" defined in these rules and in DEQ-4, 2013 edition. The amendments also clarify that shared user systems can consist of two or more living units, commercial units, or a combination of residential and commercial units. This is necessary to provide guidance about the meaning of the rules. The amendments also delete the limitation to 24 people served, and replace it with a reference to the statutory definition of public water supply and public sewage systems. This amendment is necessary because the 24-person limit does not accurately identify the threshold between a non-public and a public system. The amendment conforms to the definition of "shared wastewater system" in DEQ-4, 2013 edition. The amendments also delete the reference to the formula for determining when a shared system is subject to the design standards for public systems. The reference is not necessary because shared systems can be public based on the definitions in 75-6-102, MCA, but will not reach the public threshold based on the county average of persons per living unit.

The proposed amendment to the definition of "soil profile" in ARM 17.36.912(33) adds a reference to the soil classification method set out in Appendix B of DEQ-4. The amendment is necessary to provide guidance to permit applicants about where the required procedures can be found.

The proposed amendments to the definition of "wastewater" in ARM 17.36.912(36) delete the reference to wastewater that is discharged from a dwelling, building, or other facility. The amendment is necessary to include systems that do not discharge from a building, such as waste segregation systems and incinerator toilets. The proposed amendments also conform this definition to that in DEQ-4, 2013 edition.

The proposed amendments to the definition of "wastewater treatment system" in ARM 17.36.912(37) replace the reference to pit privies and experimental systems with a reference to all disposal methods described in DEQ-4. Pit privies and experimental systems are addressed in DEQ-4, together with a number of other types of systems. The amendment is necessary to provide a more complete reference to the types of wastewater treatment systems.

17.36.918 HORIZONTAL SETBACKS, FLOODPLAINS (1) Minimum horizontal setback distances (in feet) are as follows:

TABLE 1
SETBACK DISTANCES
(in feet)

From	To Sealed components (1) and other components (2)	To Absorption systems (3)
Public or multiple-user drinking water wells/springs	100	100
Individual and shared drinking water supply	50	100
Other wells (4)	50	100
Suction lines	50	100
Cisterns	25	50
Roadcuts, escarpments	10 ~~(4)~~ (5)	25
Slopes > ~~25%~~ 35 percent ~~(5)~~ (6)	10 ~~(4)~~ (5)	25
Property boundaries (7)	10	10
Subsurface drains	10	10
Water ~~lines~~ mains (8)	10	10
Drainfields/sand mounds (3)	10	-
Foundation walls	10	10
Surface water, Springs	50	100
Floodplains	~~-~~Sealed components - no setbacks (1) Other components - 100 (2)	100

(1) Sealed components include ~~sewer lines, sewer mains, septic tanks, grease traps, dosing tanks, pumping chambers,~~ holding tanks, ~~and~~ sealed pit privies, and

the components addressed in Department Circular DEQ-4, Chapters 4 and 5. Holding tanks and sealed pit privies must be located at least ~~10~~ ten feet outside the floodplain or any openings must be at least two feet above the floodplain elevation.
(2) Other components include ~~intermittent and recirculating sand filters, package plants, and evapotranspiration systems~~ the components addressed in Department Circular DEQ-4, Chapter 7.
(3) Absorption systems include ~~absorption trenches, absorption beds, sand mounds, and other drainfield type systems that are not lined or sealed. This term also includes seepage pits and unsealed pit privies~~ the systems addressed in Department Circular DEQ-4, Chapter 6.
(4) Other wells include, but are not limited to, irrigation and stock watering, but do not include observation wells as addressed in Department Circular DEQ-4.
Footnotes (4) and (5) remain the same, but are renumbered (5) and (6).
(7) Easements may be used to satisfy the setback to property boundaries.
(8) Sewer mains that cross water mains must be laid with a minimum vertical separation distance of 18 inches between the mains.

(2) The reviewing authority may require greater horizontal separation distances than those specified in Table 1, if it determines that site conditions or water quality ~~nondegradation~~ requirements indicate a need for the greater distance.
(3) through (5) remain the same.

AUTH: 75-5-201, MCA
IMP: 75-5-305, MCA

REASON: Existing footnotes (1), (2), and (3) of Table 1 identify sealed components, "other" components, and absorption systems that are subject to Table 1. The proposed amendments to footnotes (1), (2), and (3) delete the lists of components and systems in the footnotes and replace them with a reference to DEQ-4, Chapters 4, 5, 6, and 7. The components and systems currently listed in the footnotes are addressed in DEQ-4, but DEQ-4 includes other components and systems as well. It is not practical to list all of the components and systems in the footnote. To provide a more complete identification of components and systems that are subject to Table 1, it is necessary to identify them by reference.

The proposed amendments clarify that the setback row referring to "Public or multiple-user wells/springs" applies to "drinking water" supplies. This is necessary to clarify that the referenced setbacks apply only to water wells proposed to be used for a human drinking water supply.

Proposed new footnote (4) clarifies that the setbacks for other wells do not apply to monitoring wells. Compared with wells for irrigation or stockwater, monitoring wells do not present a significant risk of surfacing sewage, and in some cases monitoring wells must be installed close to a sewage source to determine potential impacts to water quality.

A new setback row is proposed for "Individual and shared water supply wells." Because new footnote (4) designates "other wells" as non-drinking water wells, the new row is necessary to provide a setback for individual and shared drinking water wells.

The proposed amendments to the setbacks for roadcuts, escarpments, and slopes renumber the existing footnotes from (4) to (5). The amendments increase, from 25 percent to 35 percent, the slope to which the slope setback applies. This is necessary to be consistent with the proposed amendments to ARM 17.36.322, which allow, through a Department of Environmental Quality waiver, pressure-dosed sewage treatment systems on slopes between 25 percent and 35 percent. The amendments also renumber, from (5) to (6), the existing footnote that states that the slope setback applies down-gradient of the sealed component, other component, or drainfield/soil absorption system.

The proposed amendments add a new footnote (7) to the ten-foot setback for property boundaries, to clarify that easements may be obtained to satisfy the setback. The purpose of the setback is to allow owners adequate access to their facilities for purposes of repairs and maintenance. In some cases the ten-foot buffer from the property boundary may be unavailable. In those cases, an easement from the adjoining landowner will provide adequate assurance that access is available.

The proposed amendments modify the current ten-foot setback for "water lines" so that it would apply only to "water mains." Ten feet of horizontal separation is not needed between sewage system components and water service lines. This amendment will also provide consistency with a comparable setback in the Sanitation in Subdivisions Act rules and the Uniform Plumbing Code.

The proposed amendments add a new footnote (8) to the setback, for water mains, that requires an 18-inch vertical separation between water and sewer mains. The 18-inch vertical separation requirement is currently found in DEQ-1 and the requirement is included in footnote (8) to ensure that permit applicants are aware of it.

<u>17.38.101 PLANS FOR PUBLIC WATER SUPPLY OR PUBLIC SEWAGE SYSTEM</u> (1) through (3)(n)(ii) remain the same.

(4) A person may not commence or continue the construction, alteration, extension, or operation of a public water supply system or public sewage system until the applicant has submitted a design report along with the necessary plans and specifications for the system to the department or a delegated division of local government for its review and has received written approval. Three sets of plans and specifications are needed for final approval. Approval by the department or a delegated division of local government is contingent upon construction and operation of the public water supply or public sewage system consistent with the approved design report, plans, and specifications. Failure to construct or operate the system according to the approved plans and specifications or the department's conditions of approval is an alteration for purposes of this rule. Design reports, plans, and specifications must meet the following criteria:

(a) through (c) remain the same.

(d) the board adopts and incorporates by reference ARM 17.36.320 through 17.36.325 ~~and 17.36.327~~. The design report, plans, and specifications for public subsurface sewage treatment systems must be prepared in accordance with ARM 17.36.320 through 17.36.325 ~~and 17.36.327,~~ and ~~in accordance~~ with the format and criteria set forth in Department Circular DEQ-4, "Montana Standards for Subsurface Wastewater Treatment Systems~~;~~<u>.</u>" <u>For public subsurface sewage treatment systems</u>

with a design flow greater than or equal to 2500 gallons per day, the design report, plans, and specifications must be prepared by a professional engineer.

(e) through (20) remain the same.

AUTH: 75-6-103, MCA
IMP: 75-6-103, 75-6-112, 75-6-121, MCA

REASON: ARM 17.38.101 sets out requirements for plans for public water supply and public sewage systems. The rule is promulgated under the board's authority under the public water and sewer (PWS) statutes in Title 75, chapter 6, part 1, MCA. ARM 17.38.101(4)(d) incorporates by reference sewage system rules that are promulgated by the Department of Environmental Quality (department) under the Sanitation in Subdivisions Act, Title 76, chapter 4, MCA. In this joint department/board rule notice, the department is proposing amendments to some of the Sanitation in Subdivisions Act rules incorporated by reference in ARM 17.38.101(4)(d). See department's proposed amendments to ARM 17.36.320 through 17.36.323 and ARM 17.36.325 above. If, after public comment, the department amends those Sanitation in Subdivisions Act rules, the board is proposing to incorporate the department's amendments in ARM 17.38.101. The incorporation of the Sanitation in Subdivisions Act rules within the PWS rules is necessary to maintain consistency between board PWS requirements for subsurface sewage systems and department requirements for subsurface sewage systems in proposed subdivisions.

The board is proposing to amend ARM 17.38.101(4)(d) to delete the incorporation by reference of ARM 17.36.327, which sets out provisions applicable to existing sewage systems in proposed subdivisions. The requirements in ARM 17.36.327 are less stringent than the requirements in the rules pertaining to public sewage systems. Because of the volume of sewage with which to deal, it is not appropriate for ARM 17.36.327 to apply to public sewage systems.

The proposed amendments to ARM 17.38.101(4)(d) also add a requirement that professional engineers design public subsurface sewage treatment systems with design flows greater than, or equal to, 2500 gallons per day. This requirement is currently codified in Sanitation in Subdivisions Act rules at ARM 17.36.320, but the proposed amendments will delete the requirement from ARM 17.36.320 and add it to ARM 17.38.101(4)(d). These amendments are necessary to consolidate, in the PWS rules, the requirements for design of public sewage systems by professional engineers.

17.38.106 FEES (1) remains the same.

(2) Department review will not be initiated until fees calculated under (2)(a) through ~~(e)~~ (f) and (5) have been received by the department. If applicable, the final approval will not be issued until the calculated fees under (3) and (4) have been paid in full. The total fee for the review of a set of plans and specifications is the sum of the fees for the applicable parts or subparts listed in these ~~citations.~~ subsections:

(a) The fee schedule for designs requiring review for compliance with Department Circular DEQ-1 is set forth in Schedule I, as follows:

SCHEDULE I

Policies	
ultra violet disinfection	$ 700
point-of-use/point-of-entry treatment	$ 700
Section 1.0 Engineering Report	$ 280
Section 3.1 Surface water	
quality and quantity	$ 700
structures	$ 700
Section 3.2 Ground water	$ 840
Section 4.1 Microscreening	$ 280
Section 4.~~1~~ 2 Clarification	
standard clarification	$ 700
solid contact units	$ 1,400
Section 4.~~2~~ 3 Filtration	
rapid rate	$ 1,750
pressure filtration	$ 1,400
diatomaceous earth	$ 1,400
slow sand	$ 1,400
direct filtration	$ 1,400
biologically active filtration	$ 1,400
membrane filtration	$ 1,400
micro and ultra filtration	$ 1,400
bag and cartridge filtration	$ 420
Section 4.~~3~~ 4 Disinfection	$ 700
Section 4.4 5 Softening	$ 700
Section 4.6 Ion Exchange	$ 700
Section 4.~~5~~ 7 Aeration	
natural draft	$ 280
forced draft	$ 280
spray/pressure	$ 280
packed tower	$ 700
Section 4.~~6~~ 8 Iron and manganese	$ 700
Section 4.~~7~~ 9 Fluoridation	$ 700
Section 4.~~8~~ 10 Stabilization	$ 420
Section 4.~~9~~ 11 Taste and odor control	$ 560
~~Section 4.10 Microscreening~~	~~$ 280~~
~~Section 4.11 Ion exchange~~	~~$ 700~~
Section 4.12 Adsorptive media	$ 700
Chapter 5 Chemical application	$ 980
Chapter 6 Pumping facilities	$ 980
Section 7.1 Plant storage	$ 980
Section 7.2 Hydropneumatic tanks	$ 420
Section 7.3 Distribution storage	$ 980
~~Section 7.4 Cisterns~~	~~$ 420~~
Chapter 8 Distribution system	
per lot fee	$ 70
non-standard specifications	$ 420

transmission distribution (per lineal foot) .... $ 0.25
rural distribution system (per lineal foot) .... $ 0.03
sliplining existing mains (per lineal foot) .... $ 0.15
Chapter 9 Waste disposal .... $ 700
Appendix A
new systems .... $ 280
modifications .... $ 140

(b) through (c) and Schedule III remain the same.

(d) The fee schedule for designs requiring review for compliance with Department Circular DEQ-4 is set forth in Schedule IV, as follows:

SCHEDULE IV

Chapter 4 Pressure Dosing	$	280
Chapter ~~7~~ 5 Septic Tanks	$	280
Chapter~~s 8, 10, 11, 12, 13~~ 6 Soil Absorption ~~Trenches~~ Systems	$	280
~~Chapter 9 Dosing System~~	~~$~~	~~280~~
~~Chapter 14 Elevated Sand Mounds~~	~~$~~	~~280~~
Chapter 6, Subchapter 6.8 ETA and ET Systems	$	700
Chapter~~s 15, 16, 17~~, Subchapters 7.1, 7.2, and 7.3 Filters	$	280
~~Chapters 17, 18 ETA and ET Systems~~	~~$~~	~~700~~
Chapter ~~20~~ 7, Subchapter 7.4 Aerobic Treatment	$	700
Chapter ~~21~~ 7, Subchapter 7.5 Chemical Nutrient-Reduction Systems	$	700
Chapter 7, Subchapter 7.6 Alternate Advanced Treatment Systems	$	700
Chapter ~~24, 25, 26, 27~~ 8 Holding Tanks, Pit Privy, Seepage Pits, Waste Segregation, Experimental Systems	$	280
Appendix D	$	280
Non-degradation Review	$	420

(e) ~~The fee schedule for the review of plans and specifications not covered by a specific department design standard, but within one of the following categories,~~ The fee schedule for designs requiring review for compliance with Department Circular DEQ-10 is set forth in Schedule V as follows:

SCHEDULE V

Spring box and collection lateral .... $ 350

(f) The fee schedule for designs requiring review for compliance with Department Circular DEQ-16 is set forth in Schedule VI, as follows:

SCHEDULE VI

Cisterns .... $ 420

(3) through (7) remain the same.

AUTH: 75-6-108, MCA
IMP: 75-6-108, MCA

REASON: The proposed amendment to ARM 17.38.106(2) clarifies rule language. The proposed amendment is necessary to use correct language in the rule description. The proposed amendment is house-keeping in nature and has no direct effect on the regulation.

The proposed amendments to ARM 17.38.106(2)(a) modify the review fee categories under that schedule. The proposed amendments are necessary to correspond to the proposed 2014 edition of Department Circular DEQ-1 (DEQ-1). The proposed amendments do not modify any review fee. They merely correct the line item titles to reflect the new chapter numbering and naming. The cumulative amount for impacted persons is zero because there is no proposed increase, decrease, or new amount. No persons are affected fiscally by this rule amendment because the fees remain the same for every type of application.

The proposed amendments to ARM 17.38.106(2)(d) modify the review fee categories under that table. The proposed amendments are necessary to correspond to the 2013 edition of Department Circular DEQ-4 (DEQ-4). The Schedule IV table was not updated when DEQ-4 was updated in 2013; therefore, fee item headings described in the Schedule IV table are no longer accurate. The proposed amendments do not increase any fee. They correct the line item titles to reflect the new chapter numbering and naming. The cumulative amount for impacted persons is zero because there is no proposed increase, decrease, or new amount. No persons are affected fiscally by this rule amendment because the fees remain the same for every type of application.

The proposed amendment to ARM 17.38.106(2)(e) would modify the review fee language for Schedule V. The proposed amendment is necessary to incorporate new Department Circular DEQ-10 (DEQ-10) into the line item description. The review fee is not changed. Prior to adoption of DEQ-10, the department charged a review fee for review of plans and specifications not covered by a specific design standard, which covered spring boxes and collection laterals, of $350. The review fee for spring boxes and collection laterals under new DEQ-10 will remain at $350.

The proposed addition of ARM 17.38.106(2)(f) would create a new review fee Schedule VI. The proposed amendment is necessary to incorporate new Department Circular DEQ-16 (DEQ-16) into the fee schedule. The review fee is not changed. Prior to adoption of DEQ-16, the department charged a review fee of $420 for the review of cistern plans and specifications under Department Circular DEQ-1. The review fee for cisterns under new DEQ-10 will remain at $420.

4. Concerned persons may submit their data, views, or arguments, either orally or in writing, at the hearing. Written data, views, or arguments may also be submitted to Elois Johnson, Paralegal, Department of Environmental Quality, 1520 E. Sixth Avenue, P.O. Box 200901, Helena, Montana 59620-0901; faxed to (406) 444-4386; or e-mailed to ejohnson@mt.gov, no later than 5:00 p.m., May 22, 2014. To be guaranteed consideration, mailed comments must be postmarked on or before that date.

5. Katherine Orr, attorney for the board, or another attorney for the Agency Legal Services Bureau, has been designated to preside over and conduct the hearing.

6. The board and department maintain a list of interested persons who wish to receive notices of rulemaking actions proposed by this agency. Persons who wish to have their name added to the list shall make a written request that includes the name, e-mail, and mailing address of the person to receive notices and specifies that the person wishes to receive notices regarding: air quality; hazardous waste/waste oil; asbestos control; water/wastewater treatment plant operator certification; solid waste; junk vehicles; infectious waste; public water supplies; public sewage systems regulation; hard rock (metal) mine reclamation; major facility siting; opencut mine reclamation; strip mine reclamation; subdivisions; renewable energy grants/loans; wastewater treatment or safe drinking water revolving grants and loans; water quality; CECRA; underground/above ground storage tanks; MEPA; or general procedural rules other than MEPA. Notices will be sent by e-mail unless a mailing preference is noted in the request. Such written request may be mailed or delivered to Elois Johnson, Paralegal, Department of Environmental Quality, 1520 E. Sixth Ave., P.O. Box 200901, Helena, Montana 59620-0901, faxed to the office at (406) 444-4386, e-mailed to Elois Johnson at ejohnson@mt.gov; or may be made by completing a request form at any rules hearing held by the board or department.

7. The bill sponsor contact requirements of 2-4-302, MCA, do not apply.

8. With regard to the requirements of 2-4-111, MCA, the board and department have determined that the amendment of the above-referenced rules will significantly and directly impact small businesses.

Reviewed by: BOARD OF ENVIRONMENTAL REVIEW

/s/ John F. North
JOHN F. NORTH
Rule Reviewer

BY: */s/ Robin Shropshire*
ROBIN SHROPSHIRE
Chairman

DEPARTMENT OF ENVIRONMENTAL QUALITY

BY: */s/ Tracy Stone-Manning*
TRACY STONE-MANNING, Director

Certified to the Secretary of State, April 14, 2014.

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW AND THE DEPARTMENT OF ENVIRONMENTAL QUALITY OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.36.320, 17.36.321, 17.36.322, 17.36.323, 17.36.325, 17.36.912, 17.36.918, 17.38.101, AND 17.38.106 pertain to sewage systems, definitions, horizontal setbacks, floodplains, plans for public sewage systems, and fees	**PRESIDING OFFICER REPORT**

On May 19, 2014, the undersigned presided over and conducted the public hearing held in Room 111 of the Metcalf Building, 1520 East Sixth Avenue, Helena, Montana, to take public comment on the above-captioned proposed amendments to ARM 17.36.320, 17.36.321, 17.36,322, 17.36.323, 17.36.325, 17.36.912, 17.36.918, 17.38.101 and 17.38.106 pertaining to sewage systems, definitions, horizontal setbacks, floodplains, plans for public sewage system and fees.

1. The Notice of Public Hearing on Proposed Amendment (Subdivisions/On-site Subsurface Wastewater Treatment) (Public Water and Sewage Systems Requirements) was contained in MAR Notice No. 17-359, published on April 24, 2014. A copy of the Notice Of Public Hearing On Proposed Amendment is attached to this report. (Attachments are provided in the same order as they are referenced in this report.)

2. The hearing began at 1:30 p.m. The Department of Environmental Quality recorded the hearing. Ms. Rachael Clark, Mr. Paul Nicol and Ms. Barbara Kingery from the Department were present.

3. The undersigned announced that persons at the hearing would be given an opportunity to submit their data, views, or arguments concerning the proposed action, either orally or in writing. Details of where to submit written

views or arguments were provided. At the hearing, the undersigned identified the MAR notice and read the Notice of Function of Administrative Rule Review Committee as required by Mont. Code Ann. § 2-4-302(7)(a). The rulemaking interested persons list and the opportunity to have names placed on that list were addressed. The Presiding Officer explained the order of presentation.

SUMMARY OF HEARING

4. Ms. Barbara Kingery, the Section Supervisor for the Subdivision Section of the Department of Environmental Quality (Department) gave a brief statement that the Department was recommending that the amendments be adopted as proposed. She stated the amendments are necessary to establish procedures the Department and authorized reviewing authorities will use when reviewing applications for wastewater treatment systems. The amendments among other changes, address changes in the statutes and provide terms and minimum standards for septic systems.

5. At the hearing, Mr. Tim Read of Mineral County provided comments as to the steepness of the slope in ARM 17.36.322 and the fact that ARM 17.36.322(6) contains good additions. Mr. Read agreed that the changes regarding the flood plain are a good fix. Mr. Read provided written comments as well.

SUMMARY OF WRITTEN MATERIALS

6. After the hearing, written comments were received by the Department from Mr. Tim Read from Mineral County, Ms. Susan Brueggeman from Lake County, Ms. Shannon Therrioult from Missoula County, Mr. Ryan Casne from Casne and Associates, Ms. Barbara Woodbury from Park County and Ms. Denise Moldroski from Gallatin County. The written comments of these individuals are attached.

The Department also submitted a memorandum from Department staff attorney, Mr. Paul Nicol with HB 521 and HB 311 reviews of the proposed amendments and a Private Property Assessment Act Checklist. Mr. Nichol's memorandum is attached to this report.

7. HB 521 does not apply to the proposed amendments because there are no comparable federal rules comparable to these amendments. None of the amendments are more stringent than corresponding federal rules. Therefore, no further HB 521 analysis is required. No written findings are required under Mont. Code Ann. §§ 75-5-203, 75-56-309 or 75-6-116.

8. With respect to HB 311 (the Private Property Assessment Act, Mont. Code Ann. §§ 2-10-101 through 105), the State of Montana is required to assess the taking or damaging implications of a proposed rule affecting the use of private real property. This rulemaking affects the use of private real property. A Private Property Assessment Act Checklist was prepared, which shows that the proposed amendments do not have taking or damaging implications. Therefore, no further assessment is required.

9. The period to submit comments ended at 5 p.m. on May 22, 2014.

PRESIDING OFFICER COMMENTS

10. The Board has jurisdiction to adopt, amend, or repeal the amendment pursuant to Mont. Code Ann. §§ 76-4-104, 75-5-201 and 75-6-103.

11. House Bill 521 (1995) generally provides that the Board may not adopt a rule that is more stringent than comparable federal regulations or guidelines, unless the Board makes written findings after public hearing and comment. The proposed amendments are not more stringent than a comparable federal regulation or guideline. Therefore written findings are not necessary.

12. House Bill 311 (1995), the Private Property Assessment Act, codified as Mont. Code Ann. § 2-10-101 through -105, provides that a state agency must complete a review and impact assessment prior to taking an action with taking or

damaging implications. The proposed amendments affect real property. A Private Property Assessment Act Checklist was prepared in this matter. The proposed amendments do not have taking or damaging implications. Therefore, no further HB 311 assessment is necessary.

13. The procedures required by the Montana Administrative Procedure Act, including public notice, hearing, and comment upon belief have been followed.

14. The Board and Department may adopt the proposed rule amendment, reject it or adopt the rule amendment with revisions not exceeding the scope of the public notice.

15. Under Mont. Code Ann. § 2-4-305(7), for the rulemaking process to be valid, the Board must publish a notice of adoption within six months of the date the Board published the notice of proposed rulemaking in the Montana Administrative Register, or by October 24, 2014.

Dated this 27th day of July, 2014.

KATHERINE J. ORR
Presiding Officer

MEMORANDUM

To: Board of Environmental Review
From: Paul Nicol, DEQ Staff Attorney

Re: Stringency Analysis and Takings Checklist for Proposed Amendments to ARM 17.36.320, 17.36.321, 17.36.322, 17.36.323, 17.36.325, 17.36.912, 17.36.918, 17.38.101, and 17.38.106 pertaining to sewage systems, definitions, horizontal setbacks, floodplains, plans for public sewage system, and fees.

MAR Notice No. 17-359

Date: May 19, 2014

STRINGENCY REVIEW

Sections 75-5-203, MCA, requires the Board of Environmental Review (Board) to make certain written findings after a public hearing and public comment prior to adopting a rule that is more stringent than a comparable federal standard or guideline.

The amendments are proposed pursuant the Boards's authority under section 75-5-201, MCA. The proposed action will amend ARM 17.36.320, 17.36.321, 17.36.322, 17.36.323, 17.36.325, 17.36.912, 17.36.918, 17.38.101, and 17.38.106. Federal law does not contain similar provisions. Accordingly, there are no federal regulations or guidelines comparable to these Board rules. Therefore, none of the revisions to any Board rule or regulation are more stringent than corresponding federal draft or final regulations, guidelines, or criteria, and no written findings are required under section 75-5-203, 75-5-309, or 75-6-116, MCA.

TAKINGS REVIEW

The Private Property Assessment Act, codified as section 2-10-101, MCA, requires that, prior to adopting a proposed rule that has taking or damaging implications for private real property, an agency must prepare a taking or damaging impact statement. "Action with taking or damaging implications" means:

> [A] proposed state agency administrative rule, policy, or permit condition or denial pertaining to land or water management or to some other environmental matter that if adopted and enforced would constitute a deprivation of private property in violation of the United States or Montana Constitution.

§ 2-10-103, MCA.

Section 2-10-104, MCA, requires the Montana Attorney General to develop guidelines, including a checklist, to assist agencies in determining whether an agency action has taking or damaging implications. I have completed an Attorney General's "Private Property Assessment Act Checklist" pertaining to the Board's adoption of proposed revisions in MAR Notice No. 17-359, which is attached to this memo. Based upon completion of the checklist, the proposed revisions do not have taking or damaging implications. Therefore, no further HB 311 assessment is required.

PRIVATE PROPERTY ASSESSMENT ACT CHECKLIST

MAR Notice No. 17-359

DOES THE PROPOSED AGENCY ACTION HAVE TAKINGS IMPLICATIONS UNDER THE PRIVATE PROPERTY ASSESSMENT ACT?

YES	NO		
✓		1.	Does the action pertain to land or water management or environmental regulation affecting private real property or water rights?
	✓	2.	Does the action result in either a permanent or indefinite physical occupation of private property?
	✓	3.	Does the action deprive the owner of all economically viable uses of the property?
	✓	4.	Does the action deny a fundamental attribute of ownership?
✓		5.	Does the action require a property owner to dedicate a portion of property or to grant an easement? [If the answer is NO, skip questions 5a and 5b and continue with question 6.]
✓		5a.	Is there a reasonable, specific connection between the government requirement and legitimate state interests?
✓		5b.	Is the government requirement roughly proportional to the impact of the proposed use of the property?
	✓	6.	Does the action have a severe impact on the value of the property?
	✓	7.	Does the action damage the property by causing some physical disturbance with respect to the property in excess of that sustained by the public generally? [If the answer is NO, do not answer questions 7a through 7c.]
		7a.	Is the impact of government action direct, peculiar, and significant?

_____ _____ 7b. Has government action resulted in the property becoming practically inaccessible, waterlogged, or flooded?

_____ _____ 7c. Has government action diminished property values by more than 30% and necessitated the physical taking of adjacent property or property across a public way from the property in question?

Taking or damaging implication exist if YES is checked in response to question 1 and also to any one or more of the following questions: 2, 3, 4, 6, 7a, 7b, 7c; or if NO is checked in response to questions 5a or 5b.

[signature]
(Signature)

5-19-14
(Date)

Montana Department of
ENVIRONMENTAL QUALITY MEMO

SUBJECT: Proposed Rules ARM 17.36.320, 17.36.321, 17.36.322, 17.36.323, 17.36.325, 17.36.912, 17.36.918, 17.38.101, and 17.38.106 pertaining to sewage systems, definitions, horizontal setbacks, floodplains, plans for public sewage system, and fees.

DATE: May 19, 2014

SMALL BUSINESS IMPACT ANALYSIS

Section 75-5-201, MCA, of the Water Quality Act requires the Board of Environmental Review to adopt rules that provide for conserving, protecting, maintaining and improving water quality.

Section 2-4-111, MCA requires that an agency prepare a small business impact analysis if the agency determines that proposed rules may have a significant and direct impact on small businesses. Section 2-4-111, MCA further requires that the small business impact analysis; a) identify by class or group the small businesses probably affected by the proposed rule; b) include a statement of the probable significant and direct effects of the proposed rule on the small businesses identified; and c) include a description of any alternative methods that may be reasonably implemented to minimize or eliminate any potential adverse effects of adopting the proposed rule while still achieving the purpose of the proposed rule.

The proposed rules apply to all divisions of land that create one or more parcels containing less than 20 acres, exclusive of public roadways, in order that the title to or possession of the parcels may be sold, rented, leased, or otherwise conveyed. The proposed rules also apply to any resubdivision and any condominium or area, regardless of size, that provides permanent multiple space for recreational camping vehicles or mobile homes. The administration of these rules may significantly and directly impact small businesses due to changes in requirements associated with the review and subsequent approval documents.

The proposed rules adopt New Rule II from MAR Notice No. 17-358. New Rule II requires applicants to submit to DEQ, within 90 days after completion, certified "as built" plans, and provide the documents to show how the system was constructed. Plans that require a professional engineer must be constructed within 3 years or the owner must apply for a re-review.

The proposed subsurface sewage treatment area in an application now must include an area for 100% replacement of the drainfield system. This is an increase in the amount of land needed over the existing rule for those systems that currently receive a reduction in drainfield sizing due to additional treatment methods. Thus, the new rule likely requires additional land to be set aside for some drainfield replacement areas. Minimum setback distances must be maintained from water sources such as wells and from subsurface treatment systems and their mixing zones. Generally, a 100 foot setback applies between water wells/springs and drainfields/soil absorption systems and surface water features. Thus, this rule may require applicants to allocate more land to meet setback requirements for well isolation zones, sewage treatment systems, drainfield replacement areas, and water supply.

a) Small businesses potentially impacted by the proposed rules

Assuming that all subdivision applicants meet the definition of "small business" provided in section 2-4-102(13), MCA an estimated 500 subdivision applicants would be each year by the rules.

DEQ estimates that the rules would affect all of the subdivision applications. Using data from Barbara Kingery, Montana DEQ Subdivision Section Supervisor, over the past three years of data available (FY 11-13), the number of subdivision applications annually has been 475, 466, and 521. The average of these three numbers is roughly 500, all of may be affected.

Small business consultants, engineers and construction companies hired to do additional work will benefit from the extra work created for them from the rules. The total number is impossible to estimate.

b) Probable impacts of the proposed rule on small businesses

Constraining the building of a drainfield or septic system to a certain area may create more cost to a subdivision. More land may be needed. 100 foot setbacks between drainfield mixing zones and water sources may also increase costs for subdivisions. These costs would also result from the need for additional land in addition to the expense in building the sewage treatment system including.

The required replacement area for sewage treatment systems, mixing zones, and well location zones, may also result in additional costs. These costs could be significant and would likely be more significant for subdivisions with smaller lots. Larger subdivisions tend to have more resources and can spread additional land costs over more units. In areas like Bozeman, the price of land can be substantial and would create a larger cost effect than in other parts of Montana. In limited cases, needing to buy additional land could cause a development to move locations.

A professional engineer must certify that construction was completed in accordance with the approved design. The engineer must submit to the DEQ, within 90 days after completion, certified "as built" plans, and provide the documents to show how it was constructed. Plans that require a professional engineer must be constructed within 3 years or the owner must apply for a re-review. This would affect subdivisions in terms of the additional payment to the certified engineer creating the plan.

Quantifying the costs to just over 500 businesses per year is a near impossible task given the unique situation that each small business would face. It is very likely that additional costs from this rule would be felt on the front end of each development in terms of engineering, design and actual construction as well as additional land purchases, rather than stretching over many years. If each small business incurred an additional $1,000 cost on average in each the first two years from the rules, then additional costs would be just over $1 million dollars. If each small business incurred an additional $10,000 on average in each the first two years, then additional costs on small businesses from this rule would be just over $10 million. The impacts of this rule would be felt in the long term as the rule would continue to apply every year. It is unlikely that the additional costs would shut down most projects.

Subdivisions may save costs in the long run because of the reduced risk of contamination from their systems. Other small businesses in Montana such as consultants, engineers and landscapers would almost certainly benefit from additional business that these rules create. Some small businesses may benefit from better water quality.

c) Alternative methods

There do not appear to be any alternative methods that may be reasonably implemented to minimize or eliminate any potential adverse effects of adopting the proposed rules while still achieving the purpose of the proposed rules. However, some of the affected subdivisions and public entities may be able to consolidate systems or connect to a municipal system rather than follow these rules. Some public systems may choose to develop outside the state, although this is unlikely to occur in most cases.

17-379

Johnson, Elois

From: Kingery, Barbara
Sent: Wednesday, May 28, 2014 10:26 AM
To: Johnson, Elois
Subject: FW: Better late than never?

Another comment..

Barbara Kingery, PE Supervisor
Subdivision Plan Review Program
Permitting and Compliance Division
Department of Environmental Quality
e-mail - Bkingery@mt.gov
406-444-5368

From: Barbara Woodbury [mailto:bwoodbury@parkcounty.org]
Sent: Wednesday, May 28, 2014 10:10 AM
To: Kingery, Barbara
Subject: Better late than never?

Barb,
I know you asked me a month ago to provide you with my take on the floodplain definition. I hope I am not too late; you ask so little in return for all that you do for me!

The proposed definition in 17.36.912 is not that great. I do not like the reference to 100 years---that is such a misconception! May I suggest the definition found in the floodplain rules 36.15.101(11) "the area adjoining the water course or drainway which would be covered by the floodwater of a base flood except for sheet flooding areas that receive less than 1 foot of water per occurrence and are considered zone b areas by the federal emergency management agency. The floodplain consists of the floodplain and the flood fringe."
I would maybe take the underlined language out. Maybe include some language about the 1% chance of occurring in any year instead. I think that is such an important concept for people to understand. When we say "100 year floodplain" people have a mis-conception that flooding only occurs once every 100 years versus 1% chance in any given year.

So, I hope this is not too little too late. But here is my 2 cents.
I hope all is well.
Take care,
Barbara

Barbara Woodbury, R.S., MPH, CFM
Director of Environmental Health
414 E. Callender
Livingston, MT 59047
406-222-4145, ext. 1
406-223-1189 cell

17-359

Johnson, Elois

From: Kingery, Barbara
Sent: Wednesday, May 21, 2014 12:12 PM
To: 'Tim Read'
Cc: Johnson, Elois
Subject: RE: Rule Revision Comments
Attachments: TReadRuleRevisionCommentsOrr 5-21-14.pdf; TReadRuleRevisionCommentsNicolKingery 5-21-14.pdf

Thank you Tim!

Barbara Kingery, PE Supervisor
Subdivision Plan Review Program
Permitting and Compliance Division
Department of Environmental Quality
e-mail - Bkingery@mt.gov
406-444-5368

From: Tim Read [mailto:tread@co.mineral.mt.us]
Sent: Wednesday, May 21, 2014 11:45 AM
To: Kingery, Barbara; Nicol, Paul
Subject: Rule Revision Comments

Barb and Paul, attached are my comments in regard to the hearing. Hope this helps the process. Please forward Katherine's letter on to her as I am not sure of her email. Thanks

Tim Read, R.S.

MINERAL COUNTY
ENVIRONMENTAL HEALTH and PLANNING

Mineral County

P. O. Box 396
Superior, MT 59872

300 River Street
Phone (406) 822-3525

May 21, 2014

Katherine Orr
Department of Environmental Quality - Legal
PO Box 200901
Helena MT 59620-0901

RE: Subdivision/On-Site Subsurface Wastewater Treatment Rule Revision

Dear Katherine:

The following are my comments and concerns in regard to the rule changes as proposed at the May 19, 2014, Department hearing.

ARM 17.36.322 Sewage Systems: Siting (2) I do not agree that drainfields should be allowed on slopes greater than 25 percent. The waiver process states "after consultation with the local health department" - Does this mean that the local health department can object to the proposed installation on slopes greater than 25 percent and that the waiver will not be granted? There is no language that limits the amount of design flow as with installations on slopes from 15 to 25 percent.

ARM 17.36.322 Sewage Systems: Siting (6) On lots that have drainfields identified, the identification should also apply to water well locations. This will facilitate compliance with the standard that well isolation zones must be located on the lot.

ARM 17.36.918 Horizontal Setbacks Table 1 - Water mains have a setback distance, but water lines do not. Water lines should have an established minimum distance to sealed components and absorption systems.

Thank you for this opportunity to comment.

Sincerely,

Tim Read, R.S.

Tim Read, R.S.
Mineral County Environmental Health and Planning Department

cc: Barbara Kingery, DEQ Subdivision Program

Johnson, Elois

From: Moldroski, Denise <Denise.Moldroski@gallatin.mt.gov>
Sent: Thursday, May 22, 2014 4:07 PM
To: Johnson, Elois
Subject: Public Comment on MAR 17-359
Attachments: MAR 17-359 comments 052214.pdf

Hi, please let me know if you have difficulty opening this attachment!

Thanks ☺

Denise Moldroski, MS RS
Environmental Health Specialist
Gallatin City-Co Health Dept
215 W. Mendenhall, Rm 108
Bozeman, MT 59715
406-582-3120

HEALTHY

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Environmental Health Services
215 W. Mendenhall, Rm 108
Bozeman, MT 59715-3478
406-582-3120 • Fax: 406-582-3128

May 22, 2014

Eloise Johnson, Paralegal
MT Dept of Environmental Quality
PO Box 200901
Helena, MT 59620-0901

RE: Public Comment to MAR 17-359

Generally speaking, the Gallatin City-County Health Department believes all of the proposed changes are beneficial to both the public and the environment. Thank you for tackling them in a comprehensive manner. Specific comments follow:

ARM 17.36.321 (3)(g)(ii) comment: We are happy to see this addition to the rule which will allow us more flexibility in dealing with problem situations.

ARM 17.36.323 Table 2 Setback Distances comment: We like the changes made to Table 2, especially the addition of language that allows easements to be used to satisfy setbacks to property lines and the addition of Mixing Zones and Stormwater Ponds to the table.

ARM 17.36.323 Table 2, Footnote (12) comment: Thank you for clarification that the setback between drainfields or soil absorption system to irrigation ditches does not apply if the ditch is lined with a full culvert.

ARM 17.36.912 (14) comment: The increase to 240 minutes per inch in this definition is a good change.

ARM 17.36.918 Table 1 comments: We believe that the changes made to this table help clarify the regulations and are still protective of public health and the environment.

If you have any question on the above, call me at the Gallatin City-County Environmental Health Office at 582-3120.

Sincerely,

Denise Moldroski, MS RS
Environmental Health Specialist
Gallatin City-County Health Department

Johnson, Elois

From: Shannon Therriault <stherriault@co.missoula.mt.us>
Sent: Thursday, May 22, 2014 4:50 PM
To: Johnson, Elois
Subject: MAR 17-358 and MAR 17-359
Attachments: MAR 17-358 Comments 5-22-14.pdf; MAR 17-359 Comments 5-22-14.pdf

Hello. I have attached the Missoula City-County Health Department's comments for the above MAR notices. Please let me know if you have any questions or need more information. Shannon

Shannon Therriault, R.S.
Environmental Health Supervisor
Missoula City-County Health Dept
301 West Alder
Missoula, MT 59802
(406) 258-4988

Missoula Public Health
City-County Health Department

Missoula City-County Health Department
ENVIRONMENTAL HEALTH
301 West Alder Street | Missoula MT 59802-4123
www.co.missoula.mt.us/envhealth/
Phone | 406.258.4755
Fax | 406.258.4781

Memorandum

TO: Elois Johnson, Paralegal DEQ
FROM: Shannon Therriault, Environmental Health Supervisor
DATE: May 22, 2014
RE: Comments on MAR Notice No. 17-359

Thank you for the opportunity to comment on the proposed changes in MAR Notice No. 17-359. Overall, we support the proposed changes and appreciate Barbara Kingery's and DEQ's efforts to provide opportunities to discuss potential changes the formal rule process began. We offer the following suggestions for improving the rule further:

1. **17.36.323** (Table)

   Storm water is not defined in the rule or law. We recommend that "Storm water ponds and ditches" be further clarified so that the phrase includes only those structures that usually do not have water in them. You could do this with a footnote that reads, "Storm water ponds and ditches are those structures that temporarily hold or convey water as part of storm water management."

2. **17.36.323 (13)**

   This waiver opportunity is less stringent than the minimum standards in subchapter 9. Before this waiver could be granted by DEQ, the Health Board would have to approve a variance from the local wastewater regulations. Therefore, we recommend that the waiver include a requirement for local concurrence.

3. **17.36.912** – DEFINITIONS

   The proposed change to the wastewater definition deletes the provision that it is discharged from a building, ostensibly to include waste segregation systems like incinerating toilets in the definition. This broadens the definition to storm water running off roofs or down the street, carrying waste and detritus along with it. In addition, the term "water-carried" is still central in the definition. We recommend changing the definition as follows:

   (36) "Wastewater" means human excreta or water-carried waste that is discharged from a dwelling, building, or other facility, including:
   (a) household, commercial, or industrial wastes;
   (b) chemicals;
   ~~(c) human excreta;~~ or
   (d) animal and vegetable matter in suspension or solution.

With this definition, any human excreta, whether water-carried or not, will be considered wastewater, and therefore will have to be disposed of in an approved wastewater treatment and disposal system. (It would also be possible to leave "human excreta" in the list of water-carried waste, so that it's clear that both water and non-carried human excreta is considered wastewater.)

4. **17.36.918** – HORIZONTAL SETBACKS, FLOODPLAIN

   Footnote (3) defines "absorption systems" as only those systems in DEQ 4, Subchapter 6. That leaves out seepage pits, pit privies, cesspools, and experimental systems, all of which need to be located at least 100 feet from a well or surface water. We recommend that DEQ use the same language in Table 2 in ARM 17.36.323, "Drainfields/Soil Absorption Systems" for both the far right header and the 11th item in the lefthand list (now titled "Drainfields/sand mounds") and leave off the footnote. Alternatively, the footnote could be amended to read "Absorption systems include any part of a wastewater treatment and disposal system that discharges wastewater into the ground."

17-389

Johnson, Elois

From: Kingery, Barbara
Sent: Thursday, May 22, 2014 4:36 PM
To: 'Ryan Casne, PE'; Johnson, Elois
Cc: 'Randi Triem'
Subject: RE: Comments on ARM17.36.320(4)

Thank you Ryan.

Barbara Kingery, PE Supervisor
Subdivision Plan Review Program
Permitting and Compliance Division
Department of Environmental Quality
e-mail - Bkingery@mt.gov
406-444-5368

From: Ryan Casne, PE [mailto:ryan@casneinc.com]
Sent: Thursday, May 22, 2014 4:32 PM
To: Kingery, Barbara
Cc: 'Randi Triem'
Subject: Comments on ARM17.36.320(4)

Hi Barb,

Please accept the following comments on proposed ARM 17.36.320(4):

The proposed ARM requires that the replacement area for a system approved for a size reduction must have sufficient area without considering the size reduction for the primary. This rule doesn't appear to be based in good engineering logic. Level II systems are those that discharge BOD & TSS below 30 mg/L. High BOD & TSS levels are accepted as triggers for failure of a soil-based treatment system such as a drainfield. If a Level II system is indeed generating good sub 30/30 quality effluent, the soil interface within the drainfield should almost never fail. Even with a malfunctioning or failed Level II pre-treatment system, the failure does not typically occur in the drainfield. We have designed replacement Level II systems and in each case the Level II pre-treatment (RSF, fixed film media, etc) is where the failure occurs. We also have an associate that performs service and repair of drainfields downstream of conventional level II systems (RSF's) and in every case the drainfield is in almost new condition when uncovered to repair a pipe breakage, etc. In short, the risk for soil-based failure in a drainfield is effectively mitigated when Level II treatment or another advanced treatment (SBR, etc) is installed upstream. The largest downside of this rule change is the loss of developable land. In some cases the addition of 50% can add several acres to the replacement area. I urge the Department to look at the track history of drainfield failure when installed in conjunction with advanced treatment. It is my suspicion that this occurrence is very low and therefore a rule change is unnecessary.

Thank you for your consideration,

Ryan Casne, PE
Senior Engineer, Principal
CASNE & ASSOCIATES INC.
ryan@casneinc.com
(406) 443-1656

CASNE

&

ASSOCIATES INCORPORATED

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I-359
B.J.

Johnson, Elois

From: Susan Brueggeman <sbrueggeman@lakemt.gov>
Sent: Monday, May 12, 2014 2:51 PM
To: Johnson, Elois
Cc: Kingery, Barbara
Subject: Lake County Comments - ARM 17.36 SubChapters 3 and 9
Attachments: MX-5111N_20140512_133952.pdf

Ms. Johnson:

Please see the attached comments on the proposed changes to ARM 17.36 SubChapters 3 and 9.

Thank you,

Susan Brueggeman RS
Director
Lake County Environmental Health Department
106 4th Avenue East
Polson, Montana 59860

Ph 406-883-7236
Fx 406-883-7205

sbrueggeman@lakemt.gov

To complete a survey regarding the quality of customer service provided to you, please use the following link:
https://www.surveymonkey.com/s/lakecountycustomerservice
Thank you!

LAKE COUNTY ENVIRONMENTAL HEALTH
106 FOURTH AVENUE EAST
POLSON, MT 59860-2175

PH: 406-883-7236 FAX: 406-883-7205
Email:envhealth@lakemt.gov

RECEIVED
MAY 15 2014
DEQ DIRECTORS OFFICE

May 4, 2014

Elois Johnson, Paralegal
Montana Department of Environmental Quality
P.O. Box 200901
Helena, Montana 59620-0901

RE: Comments on Rule Amendment
ARM 17.36.320 and ARM 17.36 SubChapter 9

Ms. Johnson:

Please accept the following suggested changes and comments on the proposed amendments to ARM 17.36.320 and ARM 17.36 SubChapter 9

ARM 17.36 SUBCHAPTER 3

17.36.320 Sewage Systems

(3) It should be considered if drip irrigation systems should have the same slope limitations as other systems. Drip systems are installed very shallow and are micro-dosed. Their best uses include steep slopes, confined drainfield areas, and preserving plantings such as mature trees. The requirement for six feet of natural soil for these systems on steep slopes seems a contradiction in purpose and design. It is not clear if either a slope limitation or a greater depth of natural soils is necessary for this system design.

17.36.321 Sewage Systems: Siting

(2) This section should include absorption beds in order to remain consistent with SubChapter 9. However, it should be clarified that an ETA system is excluded from the definition of an absorption bed.

(3)(g) There should be a provision to allow for the use of holding tanks, by waiver, under other special situations. Lake County has found the need for the following special uses: automotive repair shop floor drains and beer/wine mash waste. There should be some provision under subdivision review for these very necessary uses.

(3) and (4) seem to be saying the same thing from a different perspective. Perhaps they would best be consolidated to simplify the rule.

17.36.322 Sewage Systems: Siting

(1) Again, I have concern that the slope limitations do not acknowledge that drip irrigation systems are designed for use on steep slopes and there should be some provision for this design regarding slope.

(3) This section should be reworded to assure that drainfields, in addition to the locations mentioned, may not be sited in an area that subjects the drainfifeld to any activity that would create compaction. It should be clear that there is no way to design a drainfield that would allow it to be subject to compaction by driving, parking, confined livestock, etc.

17.36.323 Sewage Systems: Horizontal Setbacks

It appears there is no separation distances required between water lines and sewer lines. Therefore, they could be laid side-by-side in the same trench. This department has supported the allowance of a waiver such that if one of the lines is fully sleeved they could be in the same trench. If this lack of separation is an omission, I would advocate for the ability to waive separation distance or allow for no separation if there is an approved sleeve design for lines less than 10' apart. Or, better yet, provide a sleeve design in DEQ4 such that a waiver is not required. This is something we would use quite often in our county as we often have very difficult site conditions such as bedrock outcroppings.

This department has been instructed that DEQ does not have authority to control water lines. Under subdivision review, DEQ has authority over the public health aspects of water, wastewater, stormwater, and solid waste. DEQ clearly has authority to determine the proper setbacks between water and sewer (transport) lines. One could more logically say DEQ has no authority over setback to property lines. Certainly, DEQ has authority to prescribe setbacks for water/sewer lines. Perhaps this is an error in mixing regulatory silos – ie, between public water/sewer regs and subdivision regs.

While Lake County could run with the idea of no setbacks and this would assist many of our projects on small lots, etc., from a public health perspective, this does not seem prudent. I am very uncomfortable with the statement that DEQ does not have authority for this – since that is what subdivision review is all about. Uniform plumbing codes may address plumbing in other matters; the responsibility of DEQ is to address public and environmental health under MCA 76-4.

This department is also concerned about the ability to waiver on "other" wells to drainfields. It has always been understood that a well is a conduit to an aquifer. The purpose for its use is not relevant. An irrigation well actually has a greater potential to be pumped excessively over a long period of time than a residential (drinking water) well. Therefore, the potential for the irrigation well to pull contaminants into the aquifer is greater and there should be equal protection. It is important to remember what we are protecting: the health of those using wells and the underlying aquifer. The aquifer does not care if it is a drinking water well or an irrigation well. Contamination is contamination.

ARM 17.36 SUBCHAPTER 9

17.36.912 Definitions

(1) Absorption Bed – Because absorption beds are described as being for replacement systems only in ARM 17.36.916(3), there should be some clarification that an ETA system is not an absorption bed for the purpose of these regulations.

(3) Bedrock - This department has had concern with this definition regarding the language "has insufficient fines to provide for the adequate treatment and disposal of wastewater." It is not clear what this means. In areas that are nearly pure gravel the conditions may meet the definition of "bedrock" in that the material has little fines and little treatment. Yet, in Circular DEQ4, Chapter 2.1.7, there is no minimum regarding a minimum percolation rate; the solution being only to add sand lining to the absorption trench. The two thoughts are incongruous.

(14) "Impervious Layer - It is not understood why 240mpi was chosen here. It is understood that there previously was conflict in the rules regarding the definition of a "limiting/impervious layer", but this department does not understand why 240mpi was chosen. This department has installed wastewater systems in areas of very tight soils – percolation rate is unknown. We are concerned that the 240mpi will render properties undevelopable without an understanding of why this number was chosen.

(24) "Replacement system" This department often permits replacement systems that are for expanded homes or systems that need relocation, etc. They are not failed, failing or contaminated. We have viewed these systems as "replacement systems". It should be clarified if such systems are or are not replacement systems as replacement systems are allowed certain benefits and flexibilities under the regulations.

(36) Wastewater Treatment System – It is critical that this proposed definition be corrected. As stated at the regional meeting in Kalispell, the rules require standards for wastewater treatment systems. If a person should install a wastewater treatment system that is not per Circular DEQ4, there is essentially no legal recourse, other than proving something under 17.36.913 which may be difficult. The installer can rightly say they did not install a wastewater treatment system described in Circular DEQ4; therefore, there is no violation. Our best local example is a wastewater system that was made out of a series of Rubbermaid garbage containers. That certainly was not a system per Circular DEQ4, and it needed to be addressed as a violation.

Our local board of health has had to wrestle with what is or is not a wastewater treatment system. The definition must be as follows: *a system that receives wastewater for purposes of treatment, storage, and/or disposal.* This will allow us to have legal authority over ANY system installed for this purpose, not just those prescribed in Circular DEQ4.

ARM 17.36.914 I Technical Requirements

(3) The language "under no circumstances" should be removed since it implies there is no ability to request a variance from this separation distance.

(5)(c) This could simply refer to the groundwater monitoring procedure described in Circular DEQ4.

(6) This language needs to be amended so that a replacement system that is not failed is also subject to the 200' connection requirement. Or, the definition of a "replacement system" needs to be reconsidered per previous comments on that definition.

17.36.916 Absorption beds, etc.

(1) and (5) There needs to be a provision to allow the use of holding tanks for special purposes such as auto repair shop floor drains and brewery/winery mash. This could be by variance, and our board has allowed for this. But, it is better to recognize in rule that there may be appropriate special uses for holding tanks beyond what is allowed currently.

This department and our local board of health also are concerned with the "seasonal use only" requirement for holding tanks. One is hard-pressed to find a logical reason for this limitation. If a holding tank is the right system, it is the right system regardless of the seasonality of use. We have also allowed, under variance, uses where installation of a discharging system simply does not make sense for economic or extremely limited use situations. There should be some flexibility, beyond a local variance, to allow for holding tank use.

(5) We recommend adding the requirement for a deed restriction that requires pumping and maintenance. It has been suggested that holding tanks should have limited permits, ie for a maximum of 5 years at which time they would be required to have a tightness test and inspection to certify soundness before another permit is issued. This would be a great addition to this rule.

ARM 17.36.918 Horizontal Setbacks, etc.

It should be discussed if drip irrigation systems must meet the slope setbacks and slope limitation of other systems. They are installed at a very shallow depth and mirco-dosed, so the potential for break-out is very low.

It appears there is no separation distances required between water lines and sewer lines. Therefore, they could be laid side-by-side in the same trench. This department has supported the allowance of a waiver such that if one of the lines is fully sleeved they could be in the same trench. If this lack of separation is an omission, I would advocate for the ability to waive separation distance or allow for no separation if there is an approved sleeve design for lines less than 10' apart. Or, better yet, provide a sleeve design in DEQ4 such that a waiver is not required. This is something we would use quite often in our county as we often have very difficult site conditions such as bedrock outcroppings.

This department has been instructed that DEQ does not have authority to control water lines. Under minimum standards for wastewater systems, DEQ has authority over the public health aspects of wastewater. DEQ clearly has authority to determine the proper setbacks between water and sewer (transport) lines. One could more logically say DEQ has no authority over setback to property lines. Certainly, DEQ has authority to prescribe setbacks for water/sewer lines. Perhaps this is an error in mixing regulatory silos – ie, between public water/sewer regs and these rules.

While Lake County could run with the idea of no setbacks and this would assist many of our projects on small lots, etc., from a public health perspective, this does not seem prudent. I am very uncomfortable with the statement that DEQ does not have authority for this – since that is what the minimum wastewater treatment system standards are all about. Uniform plumbing codes may address plumbing in other matters; the responsibility of DEQ is to address public and environmental health under environmental and public health law.

This department is also concerned about the ability to waiver on "other" wells to drainfields. It has always been understood that a well is a conduit to an aquifer. The purpose for its use is not relevant. An irrigation well actually has a greater potential to be pumped excessively over a long period of time than a residential (drinking water) well. Therefore, the potential for the irrigation well to pull contaminants into the aquifer is greater and there should be equal protection. It is important to remember what we are protecting: the health of those using wells and the underlying aquifer. The aquifer does not care if it is a drinking water well or an irrigation well. Contamination is contamination.

It is also unknown why a drainfield must be, not just out of the floodplain, but 100' out of the floodplain. Also, ARM 17.36.323(14) allows for drainfields to be 100' from surface water or 2' above the maximum flood elevation. The same language should be added here.

Thank you for your consideration of the above comments.

Sincerely,

Susan K. Brueggeman, R.S.
Director

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
AND THE DEPARTMENT OF ENVIRONMENTAL QUALITY
OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.36.320, 17.36.321, 17.36.322, 17.36.323, 17.36.325, 17.36.912, 17.36.918, 17.38.101, and 17.38.106 pertaining to sewage systems, definitions, horizontal setbacks, floodplains, plans for public sewage system, and fees	)	NOTICE OF AMENDMENT (SUBDIVISIONS/ON-SITE SUBSURFACE WASTEWATER TREATMENT) (PUBLIC WATER AND SEWAGE SYSTEM REQUIREMENTS)

TO: All Concerned Persons

1. On April 24, 2014, the Board of Environmental Review and the Department of Environmental Quality published MAR Notice No. 17-359 regarding a notice of public hearing on the proposed amendment of the above-stated rules at page 747, 2014 Montana Administrative Register, Issue Number 8.

2. The department has amended ARM 17.36.320 and 17.36.325 exactly as proposed and has amended ARM 17.36.321, 17.36.322, and 17.36.323 as proposed, but with the following changes, stricken matter interlined, new matter underlined. The board has amended ARM 17.38.101 and 17.38.106 exactly as proposed and has amended ARM 17.36.912 and 17.36.918 as proposed, but with the following changes, stricken matter interlined, new matter underlined:

17.36.321 SEWAGE SYSTEMS: ALLOWABLE NEW AND REPLACEMENT SYSTEMS (1) and (2) remain as proposed.

(3) The following sewage systems may not be used for new systems, but may be used as replacement systems subject to the limitations provided in Department Circular DEQ-4:

(a) through (c) remain as proposed.

(d) ~~cesspools~~ absorption beds;

(e) through (g)(ii) remain as proposed.

(4) Cesspools are prohibited as new or replacement systems. ~~The following systems may be used only as replacement systems, subject to the limitations provided in department Circular DEQ-4~~:

~~(a) cut systems;~~

~~(b) fill systems; and~~

~~(c) artificially drained systems.~~

(5) remains as proposed.

17.36.322 SEWAGE SYSTEMS: SITING (1) through (3) remain as proposed.

~~(3)~~ (4) No component of any sewage treatment system may be located under structures or driveways, parking areas or other areas subjected to vehicular traffic, or other areas subject to compaction, except for those components of the system designed to accommodate such conditions. Drainfields must not be located in swales or depressions where runoff may flow or accumulate.

(5) through (7) remain as proposed.

17.36.323 SETBACKS

(1) Minimum setback distances, in feet, shown in Table 2 of this rule must be maintained, except as provided in the table footnotes or as allowed through a deviation granted under ARM Title 17, chapter 38, subchapter 1. The setbacks in this rule are not applicable to gray water irrigation systems that meet the setbacks and other requirements of ARM 17.36.319.

TABLE 2
SETBACK DISTANCES
(in feet)

From	To Drinking Water Wells	To Sealed Components (1) and Other Components (2)	To Drainfields/Soil Absorption Systems (3)
Public or multiple-user drinking water wells/springs	-	100 ~~(3)~~ (4)	100
Individual and shared drinking water wells	-	50 ~~(3)~~ (4)	100
Other wells ~~(4)~~ (5)	-	50 ~~(3)~~ (4)	100 ~~(3)~~ (4)
Suction lines	-	50	100
Cisterns	-	25	50
Roadcuts, escarpment	-	10 ~~(5)~~ (6)	25
Slopes > 35 percent ~~(6)~~ (7)	-	10 ~~(5)~~ (6)	25
Property boundaries	10 ~~(7)~~ (8)	10 ~~(7)~~ (8)	10 ~~(7)~~ (8)
Subsurface drains	-	10	10
Water mains	-	10 ~~(8)~~ (9)	10
Drainfields/Soil absorption systems	100	10	-
Foundation walls	-	10	10

Surface water ~~(9)~~ (10), springs	100 ~~(3)~~ (4) ~~(10)~~ (11) ~~(11)~~ (12)	50 ~~(3)~~ (4) ~~(10)~~ (11)	100 ~~(3)~~ (4) ~~(10)~~ (11) ~~(12)~~ (13)
Floodplains	10 ~~(10)~~ (11)	- Sealed components - no setbacks (1) Other components - 100 ~~(3)~~ (4) ~~(10)~~ (11)	100 ~~(10)~~ (11) ~~(13)~~ (14)
Mixing zones	100 ~~(3)~~ (4)	-	-
Storm water ponds and ditches (15)	25 ~~(14)~~ (16)	10	25

Footnotes (1) and (2) remain as proposed.

(3) Absorption systems include the systems addressed in Department Circular DEQ-4, Chapters 6 and 8, subject to the limitations in ARM 17.36.321.

Footnotes (3) through (12) remain as proposed, but are renumbered (4) through (13).

~~(13)~~ (14) After consultation with the local health department ~~A~~a waiver may be granted by the department, pursuant to ARM 17.36.601, if the applicant demonstrates that the surface water or spring seasonally high water level is at least a 100-foot horizontal distance from the drainfield and the bottom of the drainfield will be at least two feet above the maximum 100-year flood elevation.

(15) Storm water ponds and ditches are those structures that temporarily hold or convey water as part of storm water management.

Footnote (14) remains as proposed, but is renumbered (16).

17.36.912 DEFINITIONS For purposes of this subchapter, the following definitions apply:

(1) and (2) remain as proposed.

(3) "Bedrock" means material that cannot be readily excavated by hand tools, or material that does not allow water to pass through or that has insufficient quantities of fines to provide for the adequate treatment and disposal of wastewater. The term does not include gravel and other rock fragments as defined in Department Circular DEQ-4, Appendix B.

(4) through (34) remain as proposed.

(35) "Wastewater" means water-carried wastes. For purposes of these rules, wastewater does not include storm water. The term ~~including~~ includes, but is not limited to, the following:

(a) through (36) remain as proposed.

17.36.918 HORIZONTAL SETBACKS, FLOODPLAINS (1) Minimum horizontal setback distances (in feet) are as follows:

Table 1 remains as proposed.

Footnotes (1) and (2) remain as proposed.

(3) Absorption systems include the systems addressed in Department Circular DEQ-4, Chapters 6 and 8 subject to the limitations in ARM 17.36.916.

Footnotes (4) through (8) remain as proposed.

(2) through (5) remain as proposed.

3. The following comments were received and appear with the board's and department's responses:

COMMENT NO. 1: Several comments were received in response to ARM 17.36.320, 17.36.322, and 17.36.918 stating that drip irrigation systems should not have the same slope limitations, natural soil depth, and setback requirements as other systems. The comments stated that drip systems are installed at a very shallow depth and mirco-dosed. The comments also stated that the best use of a drip system includes steep slopes, confined drainfield areas, and preserving plantings such as mature trees.

RESPONSE: Steep slopes are sensitive areas that are naturally affected by erosion and landslide potential. Application of effluent by subsurface drip systems increases the possibility of failure. Six feet of natural soil below the bottom of the trench ensures that hydraulic loading will occur only in areas where there will not be a limiting layer. The suggested change has not been made.

COMMENT NO. 2: ARM 17.36.320(4) requires that the replacement area for a system approved for a size reduction must have sufficient area without considering the size reduction for the primary system. This rule does not appear to be based in good engineering logic. With a malfunctioning or failed Level II pre-treatment system, the failure does not typically occur in the drainfield. The risk of soil-based failure in a drainfield is mitigated when Level II treatment, or another advanced treatment, is installed upstream. This rule change will result in the unnecessary loss of developable land.

RESPONSE: Although effluent from advanced treatment typically has very low BOD5 and TSS effluent characteristics, if the drainfield fails it will most likely be from factors other than the use of advanced treatment. Since the department cannot predict what those factors may be, the department requires a full-sized replacement area. If site constraints do not allow this configuration, the designer may request a waiver from the requirement. Moreover, a full-sized replacement area must be required for all systems, including elevated sand mounds and those with advanced treatment in order to maintain consistency with the standards of Department Circular DEQ-4. The suggested change has not been made.

COMMENT NO. 3: ARM 17.36.321(2) and 17.36.912(1) should be amended to clarify that an evapotranspiration absorption (ETA) system is not considered an absorption bed.

RESPONSE: ETA systems are not considered absorption beds. The distinctions between the two types of systems are clear in Department Circular DEQ-4 Section 6.8, Evapotranspiration Absorption and Evapotranspiration Systems, and Section 6.11, Absorption Beds.

COMMENT NO. 4: ARM 17.36.321(3)(g) should allow the use of holding tanks by waiver for systems with special uses, such as automotive repair shop floor drains and beer/wine mash waste.

RESPONSE: Holding tanks may be approved through waiver for facilities owned and operated by a local, state, or federal unit of government or in facilities licensed by the Department of Public Health and Human Services and inspected by the local health department. The department restricts the use of holding tanks more than other wastewater treatment systems because holding tanks require a higher level of maintenance and scheduled inspections. The suggested change has not been made.

COMMENT NO. 5: A county health department supports the addition of a waiver in ARM 17.36.321(3)(g) to allow holding tanks to replace a failed system when no other alternative is available. The rule will allow the county more flexibility in dealing with problem situations.

RESPONSE: The department acknowledges this comment.

COMMENT NO. 6: ARM 17.36.321(3) and (4), which describe systems allowed for new or replacement systems, seem to say the same thing from a different perspective. The department should simplify this rule. Also, this section should include absorption beds in order to remain consistent with subchapter 9.

RESPONSE: The department agrees that ARM 17.36.321(3) and (4) could be drafted more concisely. In response to this comment, the department has consolidated the lists in (3) and (4) and clarified that cesspools are prohibited.

COMMENT NO. 7: ARM 17.36.322(4) should be reworded to ensure that drainfields will not be located in any area that is subject to compaction.

RESPONSE: The department agrees and ARM 17.36.322(4) has been modified to include language that prohibits drainfields in all areas that would be subject to compaction, not just those listed in the proposed rule.

COMMENT NO. 8: ARM 17.36.322(2) should not allow drainfields on slopes greater than 25 percent. There is no language that limits the amount of design flow as with installations on slopes that are 15 to 25 percent. Also, does an objection from the local health department mean that the waiver will not be granted?

RESPONSE: ARM 17.36.322(2) amendments allow, through a department waiver, use of pressure-dosed systems on slopes greater than 25 percent and up to 35 percent, if a qualified person performs a soil evaluation. The department has found that in some situations pressure-dosed systems can be installed on these slopes without adverse consequences. The use of the waiver process will allow for consideration of the special circumstances in each case and will ensure the appropriate design flow.

While ARM 17.36.322(2) does require consultation with the local health department, this does not give the local health department the authority to determine if a waiver will be granted. The department retains the authority to grant or deny a waiver so long as it has consulted with the local health department.

COMMENT NO. 9: A county health department agrees with the changes to Table 2 in ARM 17.36.323, especially the addition of language that allows easements to be used to satisfy setbacks to property lines. The county also agrees with the addition of mixing zones and storm water ponds to the table.

RESPONSE: The department acknowledges this comment.

COMMENT NO. 10: Several comments were received in response to setback distances in Table 2 of ARM 17.36.323 and Table 1 of ARM 17.36.918. The comments asked for clarification as to why a setback was not required between water lines and sewer lines.

RESPONSE: The department does not require setbacks between water lines and sewer lines because the department does not believe that a setback between water and sewer lines is necessary to protect public health or the environment. Water and sewer lines run close together when entering and exiting a structure and, in many cases, water and sewer lines overlap once inside. This type of design is permissible because water lines are pressurized making the likelihood of contamination from a sewer line highly unlikely, even in an instance where both lines are broken. Additionally, requiring a setback between water and sewer lines creates consistency with the department's public water supply rules.

COMMENT NO. 11: Table 2 in ARM 17.36.323 and Table 1 in ARM 17.36.918 should not allow a waiver on "other" wells to drainfields. Any type of well is a conduit to an aquifer. The purpose for its use is not relevant. For example, irrigation wells have a greater potential to be pumped excessively over a long period of time than drinking water wells. Accordingly, the potential for the irrigation well to pull contaminants into the aquifer is greater and there should be equal protection.

RESPONSE: The amendments to ARM 17.36.323 provide for a waiver to the setback between "other wells" and components of wastewater treatment systems. The amendments to ARM 17.36.918 do not provide for a waiver of the setback, but local boards of health have authority to grant a variance from the setback. The department and the board acknowledge that some wells may have more potential to pull contaminants into an aquifer than others. The waiver and variance processes will allow the department and the counties to consider the use of the well and determine the appropriate restrictions. The suggested change has not been made.

COMMENT NO. 12: A county health department appreciates the clarification in proposed footnote (12) to Table 2 in ARM 17.36.323 that the setback between drainfields or soil absorption systems to irrigation ditches does not apply if the ditch is lined with a full culvert.

RESPONSE: The department acknowledges this comment.

COMMENT NO. 13: Proposed footnote (13) to Table 2 in ARM 17.36.323 provides for a waiver to the setback between drainfields and the floodplain. Proposed Footnote (13) authorizes a waiver if the applicant demonstrates that the seasonally high water level of the surface water or spring is at least 100 feet horizontally from the drainfield and the bottom of the drainfield is at least two feet above the maximum 100-year flood elevation. The waiver is less stringent than the minimum standards in subchapter 9. Before this waiver could be granted by the department, the local board of health would have to approve a variance from local wastewater regulations. We recommend that the waiver include a requirement for local concurrence.

RESPONSE: Proposed Footnote (13) (renumbered (14)) is not new, but is an existing provision that was moved from the rule to a footnote. Footnote (14) is not less stringent than the minimum standards in ARM Title 17, chapter 36, subchapter 9. Local boards of health have broad authority to allow variances under subchapter 9 and the variance process does not set forth specific criteria that must be met for this setback. The department's authority pursuant to ARM 17.36.323 is more limited because footnote (14) sets out specific conditions that must be met in order for a waiver to be granted. The department agrees that local input is important and has amended footnote (14) to require consultation with the local health department.

COMMENT NO. 14: In Table 2 of ARM 17.36.323, we recommend that "storm water ponds and ditches" be further clarified so that the phrase includes only those structures that usually do not have water in them. The department could do this with a footnote that reads, "storm water ponds and ditches are those structures that temporarily hold or convey water as part of storm water management."

RESPONSE: The department agrees and has made the suggested change.

COMMENT NO. 15: Why does footnote (14) to Table 2 in ARM 17.36.323 require that a drainfield be 100 feet from the floodplain?

RESPONSE: Footnote (14) requires a 100-foot setback to the floodplain because during a flood the floodplain is covered by surface water. The table requires the same setback for the floodplain as it does for surface water.

COMMENT NO. 16: Footnote (14) to Table 2 in ARM 17.36.323 allows for drainfields to be 100' from surface water if the drainfield is at least 100 feet horizontally from seasonally high water and at least two feet above the maximum 100-year flood elevation. The same language should be added to ARM 17.36.918.

RESPONSE: Before making the suggested change, comments should be obtained from other government entities and individuals who would be affected by the change. The board may consider including the suggested provision in a future rulemaking. The existing provisions of subchapter 9 would allow a local board of health to grant a variance that imposed the same conditions that are set out in footnote (14) of Table 2 in ARM 17.36.323.

COMMENT NO. 17: The definition of "bedrock" in ARM 17.36.912(3) is not consistent with some of the provisions in Department Circular DEQ-4. The definition states that bedrock includes material that "has insufficient quantities of fines to provide for the adequate treatment and disposal of wastewater." Gravel could meet this condition if it had few fines. However, gravel is not treated as bedrock in Department Circular DEQ-4, Section 2.1.7.

RESPONSE: The commenter correctly points out that Department Circular DEQ-4 does not treat gravel as bedrock. Four feet of vertical separation with natural soil is required between absorption trenches and bedrock. However, footnote (c) to

Table 2.1-1 in Department Circular DEQ-4 Section 2.1.7 allows absorption trenches to be installed within four feet of gravel if the system is pressure-dosed and the trenches are sand-lined. To be consistent with Department Circular DEQ-4, the definition of "bedrock" has been modified to clarify that the term does not include gravel and other rock fragments that are defined in Department Circular DEQ-4, Appendix B.

COMMENT NO. 18: A county health department supports the increase to 240 minutes per inch in the definition of "impervious layer" in ARM 17.36.912(14).

RESPONSE: The board acknowledges this comment.

COMMENT NO. 19: The definition of "floodplain" in ARM 17.36.912(10) should be amended to remove reference to a 100-year flood. Instead, the rule should refer to this flood as one that has a one percent chance of occurring during any year.

RESPONSE: In pertinent part, the language in ARM 17.36.912(10), as it appears in the rule notice, defines a floodplain as "the area adjoining the watercourse or drainway that would be covered by a flood that is expected to recur on the average of once every 100 years or by a flood that has a one percent chance of occurring in any given year." The reference to the 100-year flood remains in the rule because this description is commonly used to refer to a flood of this intensity. The board believes that this definition is consistent with the commenter's suggestion.

COMMENT NO. 20: In the definition of "impervious layer" in ARM 17.36.912(14), the limitation of 240 minutes per inch is unnecessary. This county has successfully installed ETA systems in soils that are tighter than 240 minutes per inch. Our concern is that the 240 minutes per inch limit will unnecessarily result in declaring properties undevelopable.

RESPONSE: This definition is the same as the definition in the recently revised Department Circular DEQ-4. Soils with percolation rates slower than 240 minutes per inch have very little capacity for wastewater infiltration, requiring that other treatment options be assessed.

COMMENT NO. 21: The board should consider expanding the definition of "replacement system" in ARM 17.36.912(24) to eliminate the restriction to systems that replace a "failed, failing, or contaminating" system. This is necessary to accommodate the need for new systems on parcels that serve expanded homes or systems that need relocation. It should be clarified whether such systems are or are not replacement systems, since replacement systems are allowed certain benefits and flexibilities under the rules.

RESPONSE: New systems that serve an expanded home, or installed to relocate an existing system, are not replacement systems unless the system is replacing a "failed, failing, or contaminating system." The commenter is correct that replacement systems do not always have to meet the same requirements as new systems. Systems should be considered new systems unless they are replacing a "failed, failing, or contaminating system."

COMMENT NO. 22: The proposed amendment to the definition of "wastewater" in ARM 17.36.912(35) deletes the provision that refers to discharge from a building, in order to include waste segregation systems like incinerating toilets. However, the amendment broadens the definition so that it now could include storm water running off roofs or down the street carrying waste and detritus with it. The definition should also be amended to clarify that it applies to human excreta, whether water-carried or not.

RESPONSE: Storm water is not treated as wastewater in these rules and applicable department circulars. The definition of "wastewater" has been modified to clarify that it does not include wastes carried in storm water. A corresponding change to the definition of "wastewater" in Department Circular DEQ-4 will be proposed at a later date. The wastes listed in (a) through (d) are water-carried wastes by definition, regardless of whether they are, in fact, carried in water.

COMMENT NO. 23: In proposed ARM 17.36.912(36), the board should consider redefining "wastewater treatment system" as "a system that receives wastewater for purposes of treatment, storage, and/or disposal." This will allow the reviewing authority to have legal authority over any system installed for this purpose, not just those prescribed in Department Circular DEQ-4.

RESPONSE: In pertinent part, the definition of "wastewater treatment system," as it appears in the rule notice, already defines "wastewater treatment system" as "a system that receives wastewater for purposes of treatment, storage, or disposal." The term includes, but is not limited to, all disposal methods described in Department Circular DEQ-4. The amended definition addresses the commenter's concern.

COMMENT NO. 24: ARM 17.36.914(3) states that "under no circumstances" may the vertical separation distance between a drainfield and a limiting layer be less than four feet of natural soil. The language "under no circumstances" should be removed, since it implies there is no ability to request a variance from this separation distance.

RESPONSE: This comment is outside the scope of the current rulemaking, since no amendments were proposed to ARM 17.36.914.

COMMENT NO. 25: The ground water monitoring procedures set out in ARM 17.36.914(5)(c) could simply refer to the ground water monitoring procedure described in Department Circular DEQ-4.

RESPONSE: This comment is outside the scope of the current rulemaking, since no amendments were proposed to ARM 17.36.914.

COMMENT NO. 26: ARM 17.36.914(6) should be amended so that a replacement system that is not failed is also subject to the 200-foot connection requirement. Or, the definition of a "replacement system" needs to be reconsidered per previous comments on that definition.

RESPONSE: This comment is outside the scope of the current rulemaking, since no amendments were proposed to ARM 17.36.914. In response to the comment on ARM 17.36.912(24), the board stated that a system is considered a new system unless it is replacing a "failed, failing, or contaminating system."

COMMENT NO. 27: The board should consider amending ARM 17.36.916(1) and (5) to allow the use of holding tanks for special purposes such as auto repair shop floor drains and brewery/winery mash.

RESPONSE: This comment is outside the scope of the current rulemaking, since no amendments were proposed to ARM 17.36.916.

COMMENT NO. 28: In ARM 17.36.916, the board should consider changing the seasonal use requirement for holding tanks. If a holding tank is the right system, it is the right system regardless of the seasonality of use. There should be some flexibility, beyond a local variance, to allow for holding tank use.

RESPONSE: This comment is outside the scope of the current rulemaking, since no amendments were proposed to ARM 17.36.916.

COMMENT NO. 29: The board should consider amending ARM 17.36.916(5) to require deed restrictions for pumping and maintenance of holding tanks. Holding tanks should not be permitted for more than five years. Holding tanks should be required to have a tightness test and inspection to certify soundness before another permit is issued.

RESPONSE: This comment is outside the scope of the current rulemaking, since no amendments were proposed to ARM 17.36.916.

COMMENT NO. 30: A county health department states that the changes made to ARM 17.36.918 Table 1 help clarify the regulations and are still protective of public health and the environment

RESPONSE: The board acknowledges this comment.

COMMENT NO. 31: In the amendments to the setbacks in ARM 17.36.918, water mains have a setback distance, but water lines do not. Water lines should have an established minimum distance to sealed components and absorption systems.

RESPONSE: A horizontal setback between water service lines and drainfield components is not necessary to protect public health or the environment, because water lines are pressurized making the likelihood of contamination from a sewer line highly unlikely, even in an instance where the lines are simultaneously broken.

COMMENT NO. 32: Footnote (3) to Table 1 in ARM 17.36.918 defines "absorption systems" as only those systems in Department Circular DEQ-4 Subchapter 6. That leaves out seepage pits, pit privies, cesspools, and experimental systems, all of which need to be located at least 100 feet from a well or surface water.

RESPONSE: The commenter is correct. In order to include a reference to seepage pits, pit privies, cesspools, and experimental systems, ARM 17.36.918 has been modified to include a reference to Department Circular DEQ-4 Subchapters 6 and 8. In response to this comment, the department will make a similar modification to the setback table in the Sanitation Act rules. See modifications to Table 2 in ARM 17.36.323, above.

4. No other comments or testimony were received.

Reviewed by: BOARD OF ENVIRONMENTAL REVIEW

_____________________________ By: ___________________________________

JOHN F. NORTH
Rule Reviewer

ROBIN SHROPSHIRE
Chairman

Certified to the Secretary of State, _____________, 2014.

BOARD OF ENVIRONMENTAL REVIEW
AGENDA ITEM
EXECUTIVE SUMMARY FOR RULE ADOPTION

AGENDA # III.B.2.

AGENDA ITEM SUMMARY - The department requests the board to adopt the proposed amendments to the public water supply rules to:

1. Amend existing public water supply engineering rules to adopt updated Department Circular DEQ-1, 2014 edition, which sets forth the requirements for the design and preparation of plans and specifications for public water supply systems;
2. Amend existing public water supply engineering rules to adopt updated Department Circular DEQ-3, 2014 edition, which sets forth minimum design standards for small water systems;
3. Adopt new Department Circular DEQ-10, 2014 edition, which sets forth the standards for development of springs to serve public water supply systems;
4. Adopt new Department Circular DEQ-16, 2014 edition, which sets forth standards for cisterns to serve non-community public water supply systems;
5. Amend existing Checklists to incorporate proposed changes in DEQ-1 and DEQ-3 and previous changes to Department Circular DEQ-4, 2013 edition;
6. Clarification of existing rules related to when a professional engineer is required to submit plans and specifications;
7. Amend, for clarification, existing rules related to submission of required documents by a professional engineer;
8. Amend existing rules for clarification related to submission of plans and specifications for systems that have never submitted plans and specifications and for those systems that fail to complete construction within the 3 year window;
9. Amend the engineering review fee tables to incorporate new numbering and line item descriptions; and
10. Amend Subdivision rules that adopt DEQ-1 and DEQ-3 to reference the 2014 editions.

LIST OF AFFECTED RULES - ARM 17.38.101 and 17.36.345

AFFECTED PARTIES SUMMARY – All owners and operators of public water systems, consulting engineers, and well drillers proposing to construct or modify public water supply systems.

SCOPE OF PROPOSED PROCEEDING - The board is considering final action on adoption of amendments to the above-referenced rules as proposed in the Montana Administrative Register.

BACKGROUND – The legislature requires the Board of Environmental Review to adopt rules related to the sitting, construction, operation, and modification of a public water supply system. 75-6-103, MCA. The board has adopted various Circulars to describe design requirements. The board currently adopts Circulars DEQ-1 and DEQ-3, 2006 editions, related to the construction of public water supply systems. Montana does not adopt by reference the 10 State Standards; however, DEQ-1 and DEQ-3 include language similar to a significant portion of those standards. The 10 State Standards are intended to standardize design requirements across the country. This leads to reduced costs in design, operation, and litigation and increases consumer confidence and health. In 2012, the 10 State Standards were updated and we now propose to update DEQ-1 and DEQ-3 to incorporate those changes.

Proposed changes to DEQ-1 and DEQ-3 include references to proposed new DEQ-10 and DEQ-16.

The new Department Circular DEQ-10, 2014 edition, sets forth the standards for development of springs to serve public water supply systems. Springs are used as a source of water for some public systems; however, the board has not adopted design standards for their use. The lack of design standards does not eliminate the use of springs as a public water supply source, but may lead to the requirement that the water be treated as a surface water source as opposed to groundwater. The cost and complexity for treatment for surface water sources is many times more than what is required for groundwater sources. Design standards will reduce the costs associated with the design, construction, and operation of systems using springs and will increase public health protection.

The new Department Circular DEQ-16, 2014 edition, sets forth standards for cisterns to serve non-community public water supply systems. The use of cisterns for community systems is not allowed, but cistern's use in non-community systems may be appropriate under the right circumstances. Cisterns have increased risks associated with water quality due to their design. Design standards will reduce the costs associated with the design, construction, and operation of non-public systems proposing to use a cistern and will increase public health protection.

The board has adopted a number of Checklists that expedites the review process which reduces the costs associated with design review and approval. The proposed amendments to these checklists are intended to incorporate the proposed changes in design standards into the checklists.

The department proposes clarification of the rules related to the use of a professional engineer for the design of various facilities and for the submission of various documents. The clarifications will describe when a professional engineer is required to submit plans and specifications. Although the criterion the department uses to make that determination is not changing, the proposed amendments will clarify those criteria for the regulated public. The department also proposes to clarify the use of a professional engineer to submit required documentation. The current language infers that only the design engineer can submit those required documents. The department wishes to clarify that when a professional engineer is required to design a system, any qualified professional engineer may submit the required documents. The department has had to make exceptions to existing rule language where the design engineer has passed away, moved out of state, or has refused to submit the documents. Any engineer submitting documents is bound by their license and code of ethics to ensure the appropriate use of their stamp.

The department proposes to clarify its implementation of the statutory requirement for all systems to have department review and approval of the system's plans and specifications. There exists a significant portion of the regulated community that was in existence prior to the requirement for plan and specification review. The statute does not contain a "grandfather" clause. In its enforcement discretion, the department does not require those systems that are in continuous operation to submit plans and specifications. The department relies on the systems routine monitoring to indicate whether the system is capable of producing a safe supply of water. In the event that a system's routine monitoring indicates that the water may not be safe or where the system has not been in continuous operation, defined as being out of operation for three or more years, the department will require the system to submit plans and specifications. In either instance, the plans and specifications will be reviewed against the most current design standards and may require the system to correct design deficiencies.

The proposed amendments to ARM, 17.36.345 are intended to update the adoption by reference of DEQ-1 and DEQ-3 in the Subdivisions/On-site Subsurface Wastewater Treatment rules, to the 2014 edition.

HEARING INFORMATION – Katherine Orr conducted a public hearing on March 7, 2014, on the proposed amendments. The Presiding Officer's Report and the draft Notice of Amendment are attached to this executive summary. Draft responses to comments received are incorporated into the proposed notice.

BOARD OPTIONS - The board may:

1. Adopt the proposed amendments as set forth in the attached Notice of Public Hearing on Proposed Amendment;
2. Adopt the proposed amendments with revisions that the board finds are appropriate and that are consistent with the scope of the Notice of Public Hearing on Proposed Amendment and the record in this proceeding; or
3. Decide not to adopt the amendments.

DEQ RECOMMENDATION - The department recommends adoption of the proposed amendments as set forth in the attached Notice of Public Hearing on Proposed Amendment.

ENCLOSURES -

1. Notice of Public Hearing on Proposed Amendment
2. Presiding Officer's Report
3. HB521 and 311 Analysis
4. Public Comments
5. Draft Notice of Amendment

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
AND THE DEPARTMENT OF ENVIRONMENTAL QUALITY
OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.36.345 and 17.38.101 pertaining to adoption by reference and plans for public water supply or public sewage system	)	NOTICE OF PUBLIC HEARING ON PROPOSED AMENDMENT (WATER QUALITY) (SUBDIVISIONS/ON-SITE SUBSURFACE WASTEWATER TREATMENT) (PUBLIC WATER AND SEWAGE SYSTEMS REQUIREMENTS)

TO: All Concerned Persons

1. On March 7, 2014, at 1:30 p.m., the Board of Environmental Review and the Department of Environmental Quality will hold a public hearing in Room 111, Metcalf Building, 1520 East Sixth Avenue, Helena, Montana, to consider the proposed amendment of the above-stated rules.

2. The board and department will make reasonable accommodations for persons with disabilities who wish to participate in this public hearing or need an alternative accessible format of this notice. If you require an accommodation, contact Elois Johnson, Paralegal, no later than 5:00 p.m., February 27, 2014, to advise us of the nature of the accommodation that you need. Please contact Elois Johnson at Department of Environmental Quality, P.O. Box 200901, Helena, Montana 59620-0901; phone (406) 444-2630; fax (406) 444-4386; or e-mail ejohnson@mt.gov.

3. The rules proposed to be amended provide as follows, stricken matter interlined, new matter underlined:

17.36.345 ADOPTION BY REFERENCE (1) For purposes of this chapter, the department adopts and incorporates by reference the following documents. All references to these documents in this chapter refer to the edition set out below:

(a) Department Circular DEQ-1, "Standards for Water Works," ~~2006~~ 2014 edition;

(b) remains the same.

(c) Department Circular DEQ-3, "Standards for Small Water Systems," ~~2006~~ 2014 edition;

(d) through (f) remain the same.

(g) Department Circular DEQ-10, "Standards for the Development of Springs for Public Water Systems," 2014 edition;

(g) remains the same, but is renumbered (h).

(i) Department Circular DEQ-16, "Standards for Hauled Water Cisterns for Noncommunity Public Systems," 2014 edition;

(h) through (k) remain the same, but are renumbered (j) through (m).
(2) remains the same.

AUTH: 76-4-104, MCA
IMP: 76-4-104, MCA

REASON: The proposed amendment to (1)(a) updates the adoption by reference of Department Circular DEQ-1, "Standards for Water Works," and Department Circular DEQ-3, "Standards for Small Water Systems," to the 2014 editions. The proposed amendments to these circulars are necessary to make the design standards consistent with current industry standards contained in the Recommended Standards for Water Works, which are commonly referred to as the 10 States Standards. The 10 States Standards have been developed by the states and provinces in the Great Lakes and upper Mississippi Rivers regions to ensure the safety of drinking water. They are used by those states and provinces to apply consistent engineering standards across those regions. They are also used by a number of other states across the country, including Montana. The 10 States Standards are periodically updated to incorporate changes in technology and drinking water protection practices. The board and department do not adopt the 10 States Standards by reference. However, they do adopt, via Department Circulars DEQ-1 and DEQ-3, language similar to a significant portion of language found in the 10 States Standards. Use of these standards, or language similar to these standards, protects potable water, reduces the costs associated with the preparation of plans and specifications, and increases consumer confidence in the safety of the system. DEQ-1 and DEQ-3 are currently based on the 2003 edition of the 10 States Standards. The 10 States Standards were updated in 2007 and 2012. The board and department are proposing to modify DEQ-1 and DEQ-3 to incorporate the 2007 and 2012 changes, to make the circular consistent with recent changes to water well requirements adopted by the Board of Water Well Contractors, to make the circular consistent with recent changes to Water Use Act rules adopted by the Department of Natural Resources and Conservation, to remove requirements that are beyond the the board and department's authority, and to make style and grammar changes for readability. A more detailed summary of the major changes to these circulars is available as indicated in section 4 of this notice.

The proposed new Department Circular DEQ-10, "Standards for the Development of Springs for Public Water Systems," would set standards for the development of springs to serve public water supply systems. The proposed standards are necessary to ensure that a spring that is developed to supply water for a public system is capable of producing a safe supply of water. In addition to jeopardizing public health, incorrectly developed spring sources can be very expensive to fix. The board and department have not adopted standards for the use of springs as a public water supply source. However, they have adopted Department Circular DEQ-11, Montana Standards for Development of Springs for Individual and Shared Non-Public Systems. Proposed DEQ-10 adapts the DEQ-10 standards to public systems.

The proposed new Department Circular DEQ-16, "Standards for Hauled Water Cisterns for Noncommunity Public Systems," would set standards for the construction and maintenance of cisterns in public water supply systems. Incorrectly installed or maintained cisterns have a significant potential to create public health and regulatory issues. The proposed standards are necessary to ensure that a noncommunity public water supply system using cisterns has an adequate and safe supply of water. The board and department have not adopted standards for the use of cisterns within a noncommunity public water supply system. However, they have adopted Department Circular DEQ-17, Montana Standards for Cisterns (Water Storage Tanks) for Individual Non-Public Systems. Cisterns used for noncommunity public systems are similar to cisterns used for private systems, and proposed DEQ-16 generally adapts the DEQ-17 standards to noncommunity public systems.

17.38.101 PLANS FOR PUBLIC WATER SUPPLY OR PUBLIC SEWAGE SYSTEM (1) through (3)(n)(ii) remain the same.

(4) A person may not commence or continue the construction, alteration, extension, or operation of a public water supply system or public sewage system until the applicant has submitted a design report along with the necessary plans and specifications for the system to the department or a delegated division of local government for its review and has received written approval. Three sets of plans and specifications are needed for final approval. Approval by the department or a delegated division of local government is contingent upon construction and operation of the public water supply or public sewage system consistent with the approved design report, plans, and specifications. Failure to construct or operate the system according to the approved plans and specifications or the department's conditions of approval is an alteration for purposes of this rule. Design reports, plans, and specifications must meet the following criteria:

(a) remains the same.

(b) the design report, plans, and specifications for noncommunity water systems must be prepared in accordance with the format and criteria set forth in ~~d~~Department Circular DEQ-3, "~~Montana Department of Environmental Quality~~ Standards for Small Water Systems."

(i) The department or a delegated division of local government may require the plans and specifications for such a system to be prepared by a professional engineer when the complexity of the proposed system warrants such engineering (e.g., systems using gravity storage, pressure booster/reduction stations).

(ii) Except as provided in (iii), ~~The~~ the department or a delegated division of local government will require the plans and specifications for such a system to be prepared by a professional engineer when:

(A) ~~treatment processes and equipment,~~ system components subject to review under ~~d~~Department Circular DEQ-1, "~~Montana Department of Environmental Quality~~ Standards for Water Works," are proposed~~.~~;

(B) chlorination subject to review under Department Circular DEQ-3, "Standards for Small Water Systems," is proposed; or

(C) springs subject to review under Department Circular DEQ-10, "Standards for the Development of Springs to Serve Public Water Supply Systems" are proposed.

(iii) The department or a delegated division of local government may allow standard plans and specifications previously approved by the department to be used for such a system in place of those prepared by a professional engineer on a case-by-case basis;

(c) through (5) remain the same.

(6) Plans and specifications for a project that would violate the approval of a public water supply system, public wastewater system, or that would cause a significant deficiency, as defined in ARM 17.38.104(1), ~~will~~ may not be approved by the reviewing authority.

(7) through (8)(c) remain the same.

(9) Except as provided in ~~(10)~~ (11)(b), unless the applicant has completed the construction, alteration, or extension of a public water supply or public sewage system within three years after the department or a delegated unit of local government has issued its written approval, the approval is void and a design report, plans, and specifications must be resubmitted as required by (4) with the appropriate fees specified in this subchapter. ~~The department may grant a completion deadline extension if the applicant requests an extension in writing and demonstrates adequate justification to the department.~~

(a) If the relevant design standards and administrative rules have not changed since the original approval was issued, the department may, at its discretion, reapprove the project using the following abbreviated process:

(i) The original design report, plans, and specifications must be resubmitted as required by (4).

(ii) The engineer or firm that originally submitted the project must, in writing, grant permission for the department to re-review the plan set, and state that the conditions surrounding the original submission have not changed.

(iii) The review fee will be established by the hourly rate designated in ARM 17.38.106(3) multiplied by the time required to review the plans and specifications.

(10) Continuously active public water supply systems that have never submitted plans and specifications for department review are not required to submit plans and specifications unless specifically required by the department. All public water supply systems that are inactive for three or more years must submit a design report, plans, and specifications, as required by (4) with the appropriate fees specified in this subchapter, for approval prior to reactivation. Previously approved systems that have been inactive for three or more years may, at the department's discretion, use the abbreviated review process described in (9)(a).

~~(10)~~ (11) As provided in 75-6-131, MCA, the following requirements apply to regional public water supply systems for which a final engineering report has been approved by the United States Bureau of Reclamation. These requirements are in addition to the other requirements in this chapter, except where a rule specifically provides otherwise:

(a) and (b) remain the same.

(c) Except as provided in (4) and ~~(10)~~ (11)(b), the approval of a regional water system's standard construction contract documents and provisions for amendments to those documents remains in effect for the construction period of the project as contained in the final engineering report approved by the United States Bureau of Reclamation.

(11) remains the same, but is renumbered (12).

~~(12)~~ (13) A person may not commence or continue the operation of a public water supply or public sewage system, or any portion of such system, prior to certifying by letter to the department or a delegated division of local government that the system, or portion of the system constructed, altered, or extended to that date, was completed in substantial accordance with plans and specifications approved by the department and there are no deviations from the design standards of the applicable circulars other than those previously approved by the department pursuant to ARM 17.38.101. For a system or any portion of a system designed by a professional engineer, ~~the~~ an engineer shall sign and submit the certification letter to the department or a delegated division of local government.

~~(13)~~ (14) Within 90 days after the completion of construction, alteration, or extension of a public water supply or public sewage system, or any portion of such system, a complete set of certified "as-built" drawings must be signed and submitted to the department or a delegated division of local government. The department may require that the "as-built" submittal be accompanied by an operation and maintenance manual. For a system or any portion of a system designed by a professional engineer, ~~the~~ an engineer shall sign and submit the certified "as-built" drawings to the department or a delegated division of local government.

(14) through (18)(b) remain the same, but are renumbered (15) through (19)(b).

~~(19)~~ (20) For purposes of this chapter, the board adopts and incorporates by reference the following documents. All references to these documents in this chapter refer to the edition set out below:

(a) Department ~~of Environmental Quality~~ Circular DEQ-1, ~~2006~~ 2014 edition, which sets forth the requirements for the design and preparation of plans and specifications for public water supply systems;

(b) remains the same.

(c) Department ~~of Environmental Quality~~ Circular DEQ-3, ~~2006~~ 2014 edition, which sets forth minimum design standards for small water systems;

(d) remains the same.

(e) Department ~~of Environmental Quality~~ Water Main Certified Checklist, ~~2007~~ 2014 edition, which sets forth minimum criteria and design standards for water main extensions and replacements;

(f) Department ~~of Environmental Quality~~ Sewer Main Certified Checklist, ~~2007~~ 2014 edition, which sets forth minimum criteria and design standards for sewer main extensions and replacements;

(g) Department ~~of Environmental Quality~~ Community Water Supply Well Expedited Review Checklist, ~~2007~~ 2014 edition, which sets forth minimum criteria and design standards for new community water supply wells;

(h) Department ~~of Environmental Quality~~ Noncommunity Water Supply Well Expedited Review Checklist, ~~2007~~ 2014 edition, which sets forth minimum criteria and design standards for new noncommunity water supply wells; ~~and~~

(i) 40 CFR 141.5, which sets forth siting requirements for public water supply components~~.~~;

(j) Department Circular DEQ-10, 2014 edition, which sets forth the standards for development of springs to serve public water supply systems; and

(k) Department Circular DEQ-16, 2014 edition, which sets forth standards for cisterns to serve noncommunity public water supply systems.

~~(20)~~ (21) A copy of any of the documents adopted under ~~(19)~~ (20) may be ~~obtained from~~ viewed at the Department of Environmental Quality, P.O. Box 200901, Helena, MT 59620-0901.

AUTH: 75-6-103, MCA
IMP: 75-6-103, 75-6-112, 75-6-121, MCA

REASON: ARM 17.38.101(4)(b) is being amended to correct the titles of Department Circular DEQ-1 and Department Circular DEQ-3. This amendment has no significant impact and is housekeeping in nature only.

The other proposed amendments modify a requirement to employ a professional engineer to prepare plans and specifications for department review of a noncommunity system. The proposed amendments are necessary because all systems must submit plans and specifications for review against department design standards. The minimum standards for drinking water are described in two separate documents, Department Circular DEQ-1 and Department Circular DEQ-3. The minimum design standards are based on risks associated with exposure and the complexity of treatment. Community systems must submit under Department Circular DEQ-1 and must use a professional engineer. Noncommunity systems may submit under Department Circular DEQ-3 and are required to use a professional engineer only when directed by the department. Because of the complexities involved, the department currently requires an engineer for systems using chlorination or springs. Inserting a requirement to use a professional engineer upfront would avoid the return and resubmittal of plans and specifications that were originally submitted without the use of a professional engineer.

The proposed amendment to (6) provides clarification. The proposed amendment is necessary to clarify that the department may not approve plans and specifications that would create a violation of a previously issued approval, whether for a water system or a wastewater system, or that would create a significant deficiency.

The proposed amendments to (9) are intended to clarify the rule and to make the rule consistent with the statute. The proposed amendments are necessary because the current rule, which allows for an extension for non-completed facilities, is in conflict with the law. The law states that a system that has not completed construction within three years "must" resubmit those unconstructed portions of the facility for re-review. No authority exists for a department extension.

The proposed addition of (9)(a) is intended to create a potentially abbreviated review process for those facilities that did not complete construction within the three-year window. The proposed addition is necessary to ensure that newly constructed facilities meet the current design standards, but will also allow for a reduced cost approach when the standards used in the original review have not been significantly modified.

The proposed addition of (10) would set out the department's approval process for existing systems that have not previously been required to undergo department review and approval. The proposed addition provides that existing systems that have never received department review and approval may be subject to that requirement. In its enforcement discretion, the department does not routinely require systems that were in existence prior to the requirement for submittal of plans and specifications to submit those documents for department review and approval. Satisfactory routine monitoring reports submitted by these operating systems are used in place of the review to determine if the system is capable of producing a safe supply of water. Those systems with unsatisfactory results, or those systems that have discontinued operations for more than three years, are required to submit plans and specifications for department review. Proposed (10) lays out that process.

The proposed changes in (12) and (13) clarify that "an" engineer must submit required documents as opposed to "the" design engineer exclusively. The proposed change is necessary to resolve issues in which the design engineer is unable or unwilling to submit the required documents. The non-design engineer will be bound by the engineering code of ethics and licensure requirements to ensure the appropriate use of their stamp on a project that they may have acquired after the project was initiated.

The additional proposed change in (13) clarifies that deviations from the approved plans and specifications during construction may not violate a design standard. The proposed change is necessary to allow engineers the ability to resolve construction issues encountered during construction, but makes it clear that those changes may not violate a design standard unless the department has approved the deviation.

The proposed amendments to (20)(a) and (c) simply incorporate by reference into the public water supply rules the new versions of DEQ-1 and DEQ-3. The proposed amendments to (20)(e), (g), and (h) simply incorporate into the public water supply updated checklists that reflect the amendments made to DEQ-1 and DEQ-3. The proposed amendments to (20)(f) simply incorporate by reference into the public water supply rules an updated checklist that reflects changes made to DEQ-2 in 2012. The proposed new (20)(i) and (k) simply incorporate by reference into the public water supply rules the new DEQ-10 and DEQ-16.

4. The proposed new and amended circulars and checklists may be viewed at and copied from the department's web site at http://deq.mt.gov/wqinfo/pws/PlanReviewEngineer.mcpx. Also, copies may be obtained by contacting Leata English at Department of Environmental Quality, P.O. Box 200901, Helena, MT 59620-0901; by phone at (406) 444-4224; or by e-mail at LEnglish@mt.gov.

5. Concerned persons may submit their data, views, or arguments, either orally or in writing, at the hearing. Written data, views, or arguments may also be submitted to Elois Johnson, Paralegal, Department of Environmental Quality, 1520 E. Sixth Avenue, P.O. Box 200901, Helena, Montana 59620-0901; faxed to (406) 444-4386; or e-mailed to ejohnson@mt.gov, no later than 5:00 p.m., March 13, 2014. To be guaranteed consideration, mailed comments must be postmarked on or before that date.

6. Katherine Orr, attorney for the board, or another attorney for the Agency Legal Services Bureau, has been designated to preside over and conduct the hearing.

7. The board and department maintain a list of interested persons who wish to receive notices of rulemaking actions proposed by this agency. Persons who wish to have their name added to the list shall make a written request that includes the name, e-mail, and mailing address of the person to receive notices and specifies that the person wishes to receive notices regarding: air quality; hazardous waste/waste oil; asbestos control; water/wastewater treatment plant operator certification; solid waste; junk vehicles; infectious waste; public water supplies; public sewage systems regulation; hard rock (metal) mine reclamation; major facility siting; opencut mine reclamation; strip mine reclamation; subdivisions; renewable energy grants/loans; wastewater treatment or safe drinking water revolving grants and loans; water quality; CECRA; underground/above ground storage tanks; MEPA; or general procedural rules other than MEPA. Notices will be sent by e-mail unless a mailing preference is noted in the request. Such written request may be mailed or delivered to Elois Johnson, Paralegal, Department of Environmental Quality, 1520 E. Sixth Ave., P.O. Box 200901, Helena, Montana 59620-0901, faxed to the office at (406) 444-4386, e-mailed to Elois Johnson at ejohnson@mt.gov; or may be made by completing a request form at any rules hearing held by the board or department.

8. The bill sponsor contact requirements of 2-4-302, MCA, do not apply.

9. With regard to the requirements of 2-4-111, MCA, the department has determined that the amendment of the above-referenced rules will not significantly and directly impact small businesses.

Reviewed by: BOARD OF ENVIRONMENTAL REVIEW

/s/ John F. North
JOHN F. NORTH
Rule Reviewer

BY: */s/ Robin Shropshire*
ROBIN SHROPSHIRE
Chairman

DEPARTMENT OF ENVIRONMENTAL QUALITY

BY: */s/ Tracy Stone-Manning*
TRACY STONE-MANNING, Director

Certified to the Secretary of State, February 3, 2014.

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW AND THE DEPARTMENT OF ENVIRONMENTAL QUALITY OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.36.345 AND 17.38.101 pertaining to adoption by reference and plans for public water supply or public sewage system	PRESIDING OFFICER REPORT

On March 7, 2014, the undersigned presided over and conducted the public hearing held in Room 111 of the Metcalf Building, 1520 East Sixth Avenue, Helena, Montana, to take public comment on the above-captioned proposed amendments to ARM 17.36.345 and 17.38.101. The amendments pertain to the amendment of existing rules for clarification, the amendment of existing design circulars, and amendment of subdivision rules that adopt Department of Environmental Quality (Department) design circulars by reference.

1. The Notice of Public Hearing on Proposed Amendment (Water Quality) (Subdivisions/On-site Subsurface Wastewater Treatment) (Public Water and Sewage Systems Requirements) was contained in MAR Notice No. 17-354, published on February 13, 2014. A copy of the Notice Of Public Hearing On Proposed Amendment is attached to this report. (Attachments are provided in the same order as they are referenced in this report.)

2. The hearing began at 1:30 p.m. The Department of Environmental Quality recorded the hearing. Mr. Eugene Pizzini and Ms. Rachael Clark were present from the Department.

3. The undersigned announced that persons at the hearing would be given an opportunity to submit their data, views, or arguments concerning the proposed action, either orally or in writing. Details of where to submit written views or arguments were provided. At the hearing, the undersigned identified the MAR notice and read the Notice of Function of Administrative Rule Review

Committee as required by Mont. Code Ann. § 2-4-302(7)(a). The rulemaking interested persons list and the opportunity to have names placed on that list were addressed. The Presiding Officer explained the order of presentation.

SUMMARY OF HEARING

4. Mr. Eugene Pizzini, the Rules Expert for the Public Water Supply Section of the Department of Environmental Quality (Department) gave a brief statement that the Department was recommending that the amended rules and proposed new rules be adopted as proposed. Mr. Pizzini's testimony is attached.

5. At the hearing, Mr. Arthur Robinson of the Montana Board of Water Well Contractors provided comments as to the continuous feed method, the additional cost method and design standards consistent with Ten States Standards.

SUMMARY OF WRITTEN MATERIALS

6. After the hearing, written comments were received by the Department from the Board of Water Well Contractors, Mr. Arthur Robinson on behalf of the Water Well Contractors, Mr. Travis Ross of the Missoula County City-County Health Department and Ms. Ronda Wiggers, on behalf of the Montana Water Well Driller's Association. The written comments of these individuals are attached.

The Department also submitted a memorandum from Department staff attorney, Ms. Carol Schmidt with HB 521 and HB 311 reviews of the proposed amendments and a Private Property Assessment Act Checklist. Ms. Schmidt's memorandum is attached to this report.

7. HB 521 does not apply to the propose amendments because there are no comparable federal rules establishing design standards for public water supply and public sewage systems. Therefore, no further HB 521 analysis is required.

8. With respect to HB 311 (the Private Property Assessment Act, Mont. Code Ann. §§ 2-10-101 through 105), the State of Montana is required to assess the taking or damaging implications of a proposed rule affecting the use of private real

property. This rulemaking affects the use of private real property. A Private Property Assessment Act Checklist was prepared, which shows that the proposed amendments do not have taking or damaging implications. Therefore, no further assessment is required. A small business impact analysis was prepared for rules adopting revisions to DEQ-1 and DEQ-3. The analysis is attached

9. The period to submit comments ended at 5 p.m. on March 13, 2014.

PRESIDING OFFICER COMMENTS

10. The Board and the Department have jurisdiction to adopt, amend, or repeal the amendment pursuant to Mont. Code Ann. §§ 76-4-104 and 75-6-103.

11. House Bill 521 (1995) generally provides that the Board may not adopt a rule that is more stringent than comparable federal regulations or guidelines, unless the Board makes written findings after public hearing and comment. The proposed amendments are not more stringent than a comparable federal regulation or guideline. Therefore written findings are not necessary.

12. House Bill 311 (1995), the Private Property Assessment Act, codified as Mont. Code Ann. § 2-10-101 through -105, provides that a state agency must complete a review and impact assessment prior to taking an action with taking or damaging implications. The proposed amendments affect real property. A Private Property Assessment Act Checklist was prepared in this matter. The proposed amendments do not have taking or damaging implications. Therefore, no further HB 311 assessment is necessary.

13. The procedures required by the Montana Administrative Procedure Act, including public notice, hearing, and comment, have been followed.

14. The Board and Department may adopt the proposed rule amendment, reject it or adopt the rule amendment with revisions not exceeding the scope of the public notice.

15. Under Mont. Code Ann. § 2-4-305(7), for the rulemaking process to be valid, the Board and Department must publish a notice of adoption within six months of the date the Board published the notice of proposed rulemaking in the Montana Administrative Register, or by August 13, 2014.

Dated this 22nd day of July, 2014.

KATHERINE J. ORR
Presiding Officer

MEMORANDUM

To: Board of Environmental Review

From: Carol Schmidt
DEQ Staff Attorney

Re: HB 521 Analysis and Takings Checklist

MAR Notice No. 17-354

In the matter of the amendment of ARM 17.36.345 and 17.38.101 pertaining to adoption by reference and plans for public water supply or public sewage system.

Date: March 13, 2014

<u>HB 521 Analysis</u>

Pursuant to § 75-6-116, MCA ("HB 521"), the Board may not adopt a rule that is more stringent than comparable federal regulations or guidelines that address the same circumstances, unless the Board makes certain written findings as set forth in the statute.

The amendments to ARM 17.36.345 and 17.38.101 addresses design standards for public water supply and public sewage systems. No HB 521 findings are necessary for these amendments because there are no comparable federal rules establishing design standards for public water supply and public sewage systems.

<u>Private Property Assessment Act</u>

Section 2-10-101, MCA, requires that, prior to adopting a proposed rule that has taking or damaging implications for private real property, an agency must prepare a taking or damaging impact statement. "Action with taking or damaging implications" means:

> [A] proposed state agency administrative rule, policy, or permit condition or denial pertaining to land or water management or to some other environmental matter that if adopted and enforced would constitute a deprivation of private property in violation of the United States or Montana Constitution.

Section 2-10-103, MCA.

Section 2-10-104, MCA, requires the Montana Attorney General to develop guidelines, including a checklist, to assist agencies in determining whether an agency action has taking or damaging implications. A completed Attorney General checklist for the proposed rules is attached. Based on the guidelines provided by the Attorney General, the proposed rule amendments do not constitute an "action with taking or damaging implications" in violation of the United States or Montana Constitutions.

Attachment: Attorney General HB 311 Checklist

Board of Environmental Review
MAR Notice No. 17-354

In the matter of the amendment of ARM 17.36.354 and 17.38.101 pertaining to adoption by reference and plans for public water supply or public sewage system.

PRIVATE PROPERTY ASSESSMENT ACT CHECKLIST

DOES THE PROPOSED AGENCY ACTION HAVE TAKINGS OR DAMAGINGS IMPLICATIONS UNDER THE PRIVATE PROPERTY ASSESSMENT ACT?

YES	NO	
X		1. Does the action pertain to land or water management or environmental regulation affecting private real property or water rights?
	X	2. Does the action result in either a permanent or indefinite physical occupation of private property?
	X	3. Does the action deny a fundamental attribute of ownership? (ex.: right to exclude others, disposal of property)
	X	4. Does the action deprive the owner of all economically viable uses of the property?
	X	5. Does the action require a property owner to dedicate a portion of property or to grant an easement? [If no, go to (6)].
		5a. Is there a reasonable, specific connection between the government requirement and legitimate state interests?
		5b. Is the government requirement roughly proportional to the impact of the proposed use of the property?
	X	6. Does the action have a severe impact on the value of the property? (consider economic impact, investment-backed expectations, character of government action)
	X	7. Does the action damage the property by causing some physical disturbance with respect to the property in excess of that sustained by the public generally?
	X	7a. Is the impact of government action direct, peculiar, and significant?
	X	7b. Has government action resulted in the property becoming practically inaccessible, waterlogged or flooded?
	X	7c. Has government action lowered property values by more than 30% and necessitated the physical taking of adjacent property or property across a public way from the property in question?
	X	Takings or damaging implications? (Taking or damaging implications exist if YES is checked in response to question 1 and also to any one or more of the following questions: 2, 3, 4, 6, 7a, 7b, 7c; or if NO is checked in response to questions 5a or 5b; the shaded areas)

If taking or damaging implications exist, the agency must comply with §5 of the Private Property Assessment Act, to include the preparation of a taking or damaging impact assessment. Normally, preparation of an impact assessment will require consultation with agency legal staff.

[signature]
Signature of Reviewer

March 13, 2014
Date

Missoula Public Health
City-County Health Department

Missoula City-County Health Department
WATER QUALITY DISTRICT
301 West Alder Street | Missoula MT 59802-4123
www.co.missoula.mt.us/wq
Phone | 406.258.4890
Fax | 406.258.4781

March 3, 2014

Elois Johnson, Paralegal
Department of Environmental Quality
1520 E. Sixth Avenue, P.O. Box 200901
Helena, Montana 59620-0901

Dear Ms. Johnson

Regarding proposed changes to DEQ 1, Standards for Water Works, The Missoula Water Quality District feels the addition of improved grouting requirements in chapter 3 is considerably more protective of groundwater quality and public health than the previous requirement of continuous feed grouting. The Missoula Water Quality District fully supports this change to Circular DEQ 1. Please contact me if you have any questions.

Thank you for the opportunity to comment.

Sincerely,

Travis Ross, RS, QEP
Environmental Health Specialist

MONTANA WATER WELL DRILLERS

Association

"Join the Rest-Make MONTANA the Best."

March 12, 2014

Elois Johnson

Department of Environmental Quality

P.O. Box 200901

Helena, MT 59620-0901

RE: amendment of ARM 17.36.345 and 17.38.101

Ms. Johnson,

The Montana Water Well Driller's Association is pleased with the opportunity to offer comment on the DEQ proposed rule changes. Two of our members serve on the Board of Water Well Contractors. We have worked closely with them and Art Robinson, on reviewing the proposal. We are in complete agreement with the BWWC comments and will not repeat the information provided by Mr. Robinson.

However, we would like to reiterate one concern and add one other.

We understand the Department's desire to provide safe drinking water for commercial users. However, there is no evidence that the "drill and drive/continuous feed" method, currently allowed in rule, does not provide for this safety. In fact, test cases cited by the BWWC actually document that this is an effective and safe method to seal the casing. It is often the most cost effective for small rural business, costing as much as 30% less than the grouting proposed to be required under the new rule.

Recognizing that this is not always the appropriate method, we would respectfully ask that the Department change the proposed rule to allow "drill and drive" to be used in appropriate circumstances.

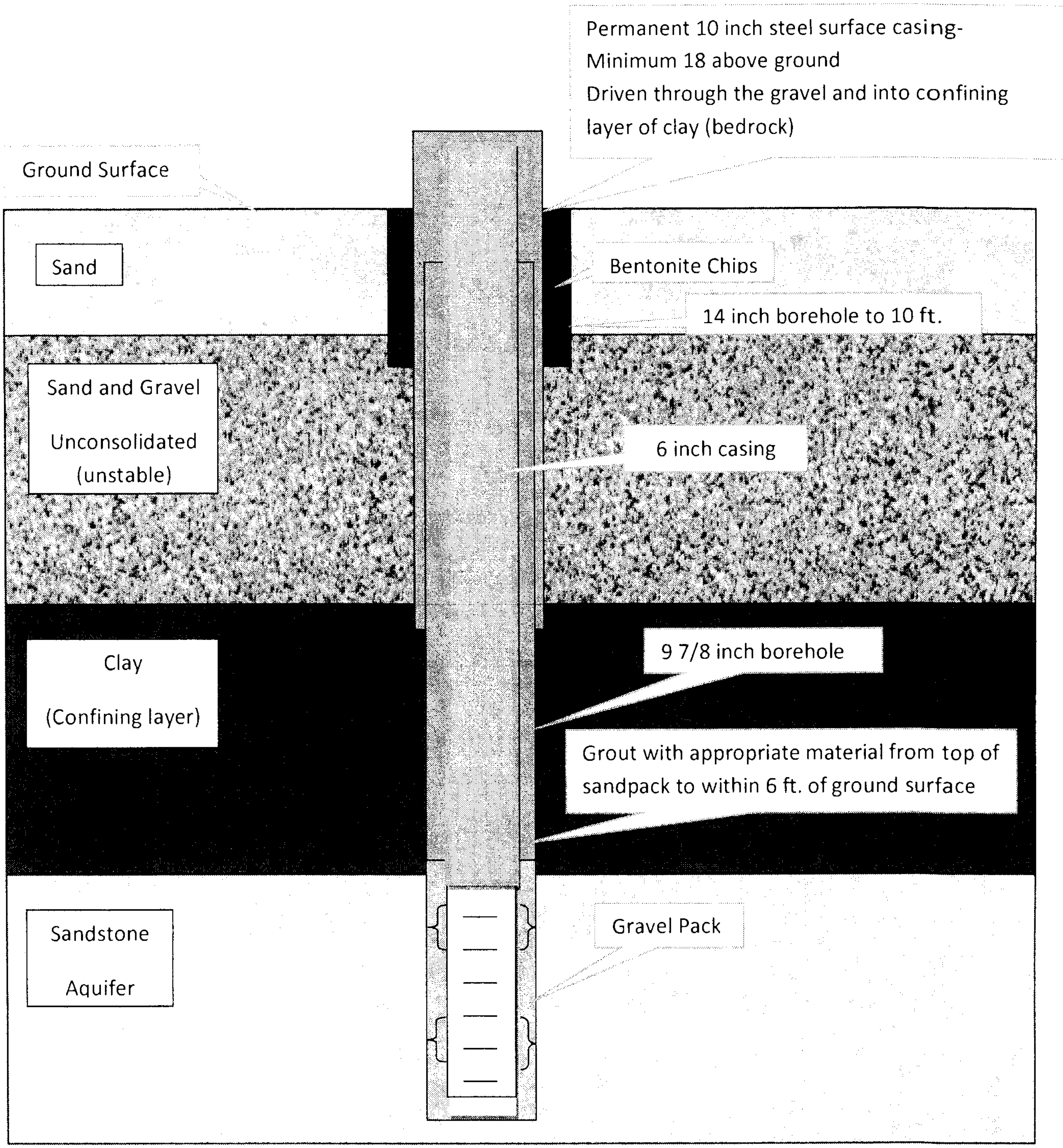
We appreciate the work that has gone into these proposed changes and the opportunity to offer industry input.

Sincerely,

Ronda Wiggers

Ronda Wiggers

Ronda Wiggers
Lobbyist
Montana Water Well Driller's Association
3208 2nd Avenue South
Great Falls, MT 59405
Office: 406-727-5659
Cell: 406-899-5659
rondawiggers@bresnan.net



I'm wondering how the DEQ would view this design.

Even though it doesn't have 25 feet of annular seal around the 10-inch surface casing, it does have an adequate annular seal from the top of the Gravel Pack, extending inside the 10-inch casing to within 6 feet of the surface.

Additionally, once the sand and gravel is encountered, the 1 ½ inch seal is pointless. Because, if surface contamination reaches the top of the unconsolidated sand and gravel, it will just migrate through this permeable layer alongside the sealing material, still contaminating the unconsolidated sand and gravel.

The Montana Board of Water Well Contractors has approved suggested changes to the DEQ's rule changes. The Board feels that clarity is needed on some rules and specific wording that is found in the Board's construction Rules. I ~~will~~ Have submit a written response that has this information in it.

The Board is opposing the elimination of the "drill and drive" method of well casing surface sealing. The dill and drive method is also referred to as the "continuous feed" method. The continuous feed method of obtaining a surface seal on the casing is approved by the Board for private domestic wells. There are soil and geological conditions in which this method is the best way for a surface seal.

$2 - 10 K Montana has ~~is diverse~~ a Big Range of Soil Cond. are

The drill and drive is more economical for the smaller water systems. Rural small water systems will be affected by the increase in drilling cost due to the additional work required to over drill and grout the casing. Some of these are churches, bars and restaurants, and mobile trailer courts. The additional cost to the well owner of having to over-drill and grout vs. continuous feed, maybe the difference of staying open and in business or closing. The difference in cost is not equal to the benefit.

$ 2.10K No Known Contaminated Sources from drill-drive Method - done follow all state MT Const. Standards

In 1994 geologist from University of South Florida conducted a study on the sealing of the drill and drive method. The study concluded that this method of a surface seal provided a seal that did not allow surface water penetration to the aquifer.

The Nebraska Grout Study referred to in the summary of the proposed rule change did not study the continuous feed method of surface sealing. The Nebraska Grout Study studied only different grout, cement and sand mixtures as to what provided the best grout seal. That Study concluded that bentonite alone provided the best seal and this is what the continuous feed method provides.

A summary of the drill and drive study is attached to the written comments that I have submitted. A full copy of the study may be obtained from Rachel Clark, DEQ PWS enforcement or from me, Art Robinson, Program Manager for the Montana Board of Water Well Contractors.

TESTIMONY INFORMATION

TO: All Concerned Persons

If you wish to make an oral statement at this hearing, please fill in this form and give it to the Presiding Officer. Your name will be called during the hearing.

If you have a written statement, data, views, or arguments, you may write on this form or attach your written material to this form.

Please Print:

NAME: Arthur Robinson

I am representing: ________ Myself

________ The following organization:

Montana Board of Water Well Contractors

Address: PO Box 201601

Helena, MT 59620

Telephone: 444-6643

Fax: 444-0533

E-mail: arobinson@mt.gov

Please check:

________ I am a proponent of the proposed rulemaking.

___X___ I am an opponent of the proposed rulemaking.

________ I am neutral but wish to offer information pertinent to the proposed action.

________ Other

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
AND THE DEPARTMENT OF ENVIRONMENTAL QUALITY
OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.36.345 and 17.38.101 pertaining to adoption by reference and plans for public water supply or public sewage system	NOTICE OF AMENDMENT (WATER QUALITY) (SUBDIVISIONS/ON-SITE SUBSURFACE WASTEWATER TREATMENT) (PUBLIC WATER AND SEWAGE SYSTEMS REQUIREMENTS)

TO: All Concerned Persons

1. On February 13, 2014, the Board of Environmental Review and the Department of Environmental Quality published MAR Notice No. 17-354 regarding a notice of proposed amendment of the above-stated rules at page 267, 2014 Montana Administrative Register, Issue Number 3.

2. The department has amended ARM 17.36.345 exactly as proposed and the board has amended ARM 17.38.101 exactly as proposed. The only changes made in response to these comments are made to the Circulars and not the rules.

3. The following comments were received and appear with the board's responses:

COMMENT NO. 1: In DEQ-1 Standard 3.2.1.3(a)(2), if the "well exclusion zone" and "well continued protection zone" mean the same thing, only one term should be used.

RESPONSE: The board and department concur with this recommendation and, to ensure clarity and consistency between public water and subdivision rules, have replaced both terms with "well isolation zone" and have added the statutory definition of "well isolation zone" found in 76-4-102, MCA, to the Glossary.

COMMENT NO. 2: Sources of viral contamination should be identified in DEQ-1 Standard 3.2.5.2(d) and DEQ-3 Standard 3.2.5.1(d).

RESPONSE: The board and department concur with this recommendation and have changed the language to state "sources of viral or bacterial contamination from human or animal waste."

COMMENT NO. 3: In DEQ-1 Standard 3.2.5.7(b)(3) and DEQ-3 Standard 3.2.5.5(b)(2)(c), the period of time that work is to be discontinued is specified by the Board of Water Well Contractors in ARM 36.21.654(1)(d) and should be referenced in this standard.

<u>RESPONSE:</u> The board and department concur with this recommendation and have added the language "in accordance with ARM 36.21.654(1)(d)" to these standards.

<u>COMMENT NO. 4:</u> In DEQ-1 Standard 3.2.5.7(b)(4) and DEQ-3 Standard 3.2.5.5(b)(2)(d), for sealing materials, the proper density and percent should be as specified by the Board of Water Well Contractors in ARM 36.21.634(34).

<u>RESPONSE:</u> The board and department concur with this recommendation and have added the language "and must be applied in accordance with the definitions in ARM 36.21.634" to these sections.

<u>COMMENT NO. 5:</u> Two commenters stated that, in DEQ-1 Standard 3.2.5.7(b)(6) and DEQ-3 Standard 3.2.5.5(b)(2)(f), there are some soil conditions where the drill and drive method provides the best surface seal. There is no proof that the drill and drive method does not provide a good surface seal. The method of drill and drive placement of bentonite is allowed by the Board of Water Well Contractors for private wells. The drill and drive method saves money and time.

<u>RESPONSE:</u> The board and department concur that there may be situations where the drill and drive method provides an appropriate surface seal, depending on the specific lithology at a proposed well site. The board and department prefer to address those situations through the deviation process to ensure site-specific information can be analyzed. In order to clarify that the board and department may consider a deviation to allow drill and drive, in lieu of the required 1.5 inches of exterior grout, DEQ-1 Standard 3.2.5.7(b)(6) and DEQ-3 Standard 3.2.5.5(b)(2)(f) have been deleted.

<u>COMMENT NO. 6:</u> Regarding DEQ-1 Standard 3.2.5.7 (b)(6), the Missoula Water Quality District feels the addition of improved grouting requirements in Chapter 3 is considerably more protective of ground water quality and public health than the previous requirement of continuous feed grouting. The Missoula Water Quality District fully supports this change to Circular DEQ-1.

<u>RESPONSE:</u> The board and department concur that the proposed grouting standard is more protective of ground water quality in most circumstances, but has deleted DEQ-1 Standard 3.2.5.7(b)(6) and DEQ-3 Standard 3.2.5.5(b)(2)(f) to clarify that deviations from this standard may be considered under appropriate circumstances. See Response to Comment No. 5, above.

<u>COMMENT NO. 7:</u> With regard to DEQ-1 Standard 3.2.6.5(c), gravel refill pipes are not typically used and this option should be eliminated.

<u>RESPONSE:</u> The board and department concur with this recommendation and have deleted DEQ-1 Standards 3.2.6.5(c) and (d).

<u>COMMENT NO. 8:</u> In DEQ-1 Standard 3.2.7.4, the description of the pitless adapter and pitless unit are combined in description and use. This section should be rewritten to match industry standards and definitions.

<u>RESPONSE:</u> The department interprets this comment to mean that, although the title in DEQ-1 Standard 3.2.7.4 is "pitless well units," it includes both pitless well units and pitless adapters. This section should be rewritten to clarify that the

standards in (a), (b), and (c) only apply to pitless well units and that pitless adapters can be used in lieu of pitless well units. Definitions for pitless well units and pitless adapters also should be added.

The board and department concur with this recommendation and have amended the title of this section to state "Pitless Well Units and Adapters," added definitions for both "pitless adapter" and "pitless unit" to the glossary, and added a new (d) that states "pitless adapters may be used in lieu of pitless units."

COMMENT NO. 9: In DEQ-1 Standard 3.2.7.8, clarify what types of liners are covered under this standard.

RESPONSE: The board and department concur with this recommendation and have changed the tile of this section to "Well Liners" to clarify the intent of the standard.

COMMENT NO. 10: In DEQ-3 Standard 3.2.5.2(b), this section should match DEQ-1 Standard 3.2.5.3(b) and require a drive shoe when driven.

RESPONSE: The board and department concur with this recommendation and have changed this standard to match the language in DEQ-1 to ensure consistent standards.

4. No other comments or testimony were received.

Reviewed by:	BOARD OF ENVIRONMENTAL REVIEW
______________________	By: ______________________
JOHN F. NORTH Rule Reviewer	ROBIN SHROPSHIRE Chairman

Certified to the Secretary of State, ______________, 2014.

BOARD OF ENVIRONMENTAL REVIEW
AGENDA ITEM
EXECUTIVE SUMMARY FOR RULE ADOPTION

Agenda Item # III.B.3.

Agenda Item Summary – The department requests that the board adopt the rules for numeric criteria for total nitrogen and total phosphorus for the protection of surface water beneficial uses. These include a new Department Circular, DEQ-12A, which contains the standards. The surface waters affected are: virtually all wadeable streams and small rivers statewide; and one large river segment (lower Yellowstone River).

List of Affected Board Rules – ARM 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.619, 17.30.622, 17.30.623, 17.30.624, 17.30.625, 17.30.626, 17.30.627, 17.30.628, 17.30.629, 17.30.635, 17.30.702, and 17.30.715.

Affected Parties Summary –The primary parties affected by the standards are current and future MPDES permit holders. This includes both municipalities and private companies.

Scope of Proposed Proceeding – The department requests that the board adopt Department Circular DEQ-12A and the rules that were published in MAR notice No. 17-356 with the amendments which have been provided in the board mailing package.

Background – The department has documented that various forms of nitrogen and phosphorus rank among the top ten most common types of pollution in Montana's flowing waters. In fact, excess nitrogen and phosphorus levels account for 17 percent of all stream miles impaired by all forms of water pollution in Montana. The intent of the proposed nutrient standards is to control the undesirable effects of eutrophication. Eutrophication is the enrichment of a waterbody (e.g., a stream or lake) by nitrogen and phosphorus, which leads to increased plant and algae growth and decay and all the consequential changes to the water quality that occur as a result. At present, Montana does not have numeric water quality standards for controlling eutrophication, except on the Clark Fork River. The proposed criteria will protect surface water beneficial uses from eutrophication impacts linked to nitrogen and phosphorus.

The department began developing statewide numeric nutrient criteria for flowing waters in 2001 and completed the most up-to-date criteria recommendations in May of 2013. This work has been extensively peer reviewed by external academic reviewers. The scientifically-derived criteria concentrations are low in relation to commonly-used wastewater treatment technologies of today. As a result, starting in late 2008, the department began hosting informal meetings with affected stakeholders (municipalities, industries, forestry, agriculture, environmental interests, etc.). Later, senate bills in the 2009 and 2011 legislative sessions gave the department authority to provide MPDES permit holders temporary variances from the criteria when economic impacts or limits of technology preclude a discharger from meeting the criteria. Temporary variances allow time for technologies to advance and alternative effluent and nonpoint source management methods to be considered and implemented. These laws are now codified at 75-5-313, MCA.

In addition to allowances for variances, the 2009 legislative actions created a new advisory group to the department, the nutrient work group. The department has met with the Nutrient Work Group 24 times since its formation and many complex implementation issues associated with the criteria and the variances have been addressed and resolved.

Rulemaking associated with the variances discussed above are department rules, and were not directly considered by the board. The department rules have followed a parallel public process to the board rules, with an identical public comment period and a public hearing which was held at the same place on the same day as the board rules. The department has received a comparable number of comments on both rulemakings.

Hearing Information – A public hearing for these rules was noticed in MAR notice No. 17-256 and was held on March 24th, 2014 in room 111 of the Lee Metcalf Building, Helena. The board attorney received both oral and written comments. The public comment period ended on April 1, 2014, by which time additional written comments were received.

Board Options – The board may:

1. Adopt the rules as proposed;
2. Adopt the rules with modifications; or
3. Not adopt the rules.

DEQ Recommendation – The department recommends the board adopt the rules with the modifications contained in the enclosed draft notice of amendment

Enclosures –

1. Notice of Public Hearing on Proposed Amendment
2. Presiding Officer Report
3. Amended Department Circular DEQ-12A
4. Public Comments
5. HB 311 and 521 Analyses
6. Draft Notice of Amendment with comments/responses and amendments

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.619, 17.30.622, 17.30.623, 17.30.624, 17.30.625, 17.30.626, 17.30.627, 17.30.628, 17.30.629, 17.30.635, 17.30.702, and 17.30.715 pertaining to permit application, degradation authorization, and annual permit fees, specific restrictions for surface water mixing zones, standard mixing zones for surface water, definitions, incorporations by reference, A-1 classification standards, B-1 classification standards, B-2 classification standards, B-3 classification standards, C-1 classification standards, C-2 classification standards, I classification standards, C-3 classification standards, general treatment standards, definitions, and criteria for determining nonsignificant changes in water quality	NOTICE OF PUBLIC HEARING ON PROPOSED AMENDMENT (WATER QUALITY)

TO: All Concerned Persons

1. On March 24, 2014, at 2:00 p.m., the Board of Environmental Review will hold a public hearing in Room 111, Metcalf Building, 1520 East Sixth Avenue, Helena, Montana, to consider the proposed amendment of the above-stated rules. At 9:00 a.m., immediately preceding the hearing for MAR Notice No. 17-355 (which is scheduled for 10:00 a.m.), at the same location, the Department of Environmental Quality will hold an informal question and answer session regarding this rulemaking and MAR Notice No. 17-355, which is the Department's proposed adoption of numeric nutrient standards variances rules.

2. The board will make reasonable accommodations for persons with disabilities who wish to participate in this public hearing or need an alternative accessible format of this notice. If you require an accommodation, contact Elois Johnson, Paralegal, no later than 5:00 p.m., March 10, 2014, to advise us of the nature of the accommodation that you need. Please contact Elois Johnson at Department of Environmental Quality, P.O. Box 200901, Helena, Montana 59620-0901; phone (406) 444-2630; fax (406) 444-4386; or e-mail ejohnson@mt.gov.

3. The board is proposing to adopt new Department Circular DEQ-12A (DEQ-12A), which contains base numeric nutrient standards for total nitrogen and

total phosphorus, and to incorporate new DEQ-12A into the surface water quality classifications and the nondegradation rules. The board is also proposing rule amendments pertaining to definitions and a low flow for base numeric nutrient standards appropriate for the design of disposal systems.

The department has documented that various forms of nitrogen and phosphorus rank as the 4th, 8th, 10th, and 12th most common types of pollution in Montana's flowing waters. In fact, excess nitrogen and phosphorus levels account for 17 percent of all stream miles impaired by all forms of water pollution in Montana. The intent of the proposed nutrient standards is to control the undesirable effects of eutrophication. Eutrophication is the enrichment of a waterbody (e.g., a stream or lake) by nitrogen and phosphorus, which leads to increased plant and algae growth and decay and all the consequential changes to the water quality that occur as a result. At present Montana does not have numeric water quality standards for controlling eutrophication, except on the Clark Fork River. Therefore, in most cases, permit limits, including waste load allocations determined in Total Maximum Daily Loads (i.e., TMDLs) are based upon the narrative water quality standard. The narrative standard prohibits substances in water that "create conditions which produce undesirable aquatic life" (ARM 17.30.637(1)(e)). Translating the narrative standard into enforceable permit limits on a case-by-case basis is time-consuming, dependent upon judgment which invites controversy, and may result in inconsistent or differing permit limits due to various interpretations among permit or TMDL writers. Numeric nutrient criteria will resolve this.

The effects of excess nitrogen and phosphorus in streams and rivers go well beyond the undesirable aquatic life referred to in the narrative standard. Excess nitrogen and phosphorus affect other water quality parameters for which Montana already has standards (dissolved oxygen, pH). The state of the science is such that linkages can clearly be made between nitrogen and phosphorus concentrations and these other, already-adopted standards. Thus, the numeric nutrient criteria will also ensure protection and attainment of Montana's dissolved oxygen and pH standards which are, in and of themselves, critical to the protection of fish and aquatic life.

State law requires that waterbodies support multiple beneficial uses (e.g., agriculture, fish and associated aquatic life, recreation). In turn, a water quality criterion for a given pollutant is established at a concentration that protects the most sensitive of the beneficial uses from the impacts caused by the pollutant. Numeric criteria for nitrogen and phosphorus concentrations are contained in DEQ-12A and vary geographically across the state. For streams and small rivers of western Montana, the numeric nutrient criteria have generally been established at concentrations that will prevent nuisance levels of bottom-attached algae and ensure that dissolved oxygen levels are maintained at standards already established by the state. The nuisance threshold for attached algae was determined via scientific polling of Montana citizens and river and stream users, and is, therefore, associated with the recreation use. Dissolved oxygen standards, in contrast, are associated with the fish and aquatic life beneficial use. In western Montana, the fish and aquatic life use and the recreation use have broadly similar sensitivities to nitrogen and phosphorus pollution.

In eastern Montana, the criteria are established at levels that will protect the indigenous fish populations and will generally ensure that dissolved oxygen levels do not decline below state standards. The attached algae threshold was not used to derive nutrient criteria for eastern Montana streams and small rivers because (a) the department's scientific poll did not address the types of streams typical of eastern Montana, and (b) attached algae levels higher than the nuisance threshold have been periodically observed in reference streams of the region. Nitrogen and phosphorus criteria concentrations are substantially higher in eastern Montana and this is due, in part, to the higher natural turbidity of those streams. Nutrient criteria for large rivers are mostly still under development. However, they have been completed for a large river segment (the lower Yellowstone), which is included in DEQ-12A. In the lower Yellowstone River, the nutrient criteria are set at concentrations that will prevent nuisance bottom-attached algae and extreme variations in pH (the latter of which impacts fish). The scientific bases for the criteria are laid out in more detail in the following documents: Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers (2008) and Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers: Update 1 (2013). These documents may be viewed on the department's web site at http://www.deq.mt.gov/wqinfo/standards/NumericNutrientCriteria.mcpx. They may also be obtained from the department at the address or phone number listed in paragraph 5 of this notice.

The nutrient criteria concentrations being proposed for adoption as standards are generally low, particularly in the western region of Montana. In many cases, the concentrations are below the limits of current wastewater treatment technology, particularly for nitrogen. Therefore, when little or no stream dilution is available, dischargers will find it difficult or impossible to meet the standards. Senate Bill 95 (2009 Legislature) and Senate Bill 367 (2011 Legislature), now codified at 75-5-313, MCA, addressed the high cost and technological difficulties associated with meeting the nutrient standards in the short term. Section 75-5-313, MCA, allows dischargers to be granted variances from numeric nutrient standards, once the criteria have been adopted as standards, in those cases where meeting the standards today would be an unreasonable economic burden or technologically infeasible. Variances from the standards may be granted for up to twenty years. Thus, 75-5-313, MCA, allows for the nutrient standards to be met in a staged manner, over time, as alternative effluent management methods are considered, nutrient removal technologies become more cost-effective and efficient, and nonpoint sources of nutrients are addressed. Rules implementing 75-5-313, MCA, are within the rulemaking authority of the Department of Environmental Quality, not the Board of Environmental Review. Concurrent with the board's rulemaking process initiated by this notice, the department has proposed rulemaking to implement the variance process. See MAR Notice No. 17-355. The department will hold a separate hearing on those rules. Comments regarding the variance process must be submitted to the department as indicated in MAR Notice No. 17-355.

4. The rules proposed to be amended provide as follows, stricken matter interlined, new matter underlined:

17.30.201 PERMIT APPLICATION, DEGRADATION AUTHORIZATION, AND ANNUAL PERMIT FEES (1) through (5) remain the same.

(6) The fee schedules for new or renewal applications for, or modifications of, a Montana pollutant discharge elimination system permit under ARM Title 17, chapter 30, subchapter 11 or 13, a Montana ground water pollution control system permit under ARM Title 17, chapter 30, subchapter 10, or any other authorization under 75-5-201, 75-5-301, or 75-5-401, MCA, or rules promulgated under these authorities, are set forth below as Schedules I.A, I.B, I.C, and I.D. Fees must be paid in full at the time of submission of the application. For new applications under Schedule I.A, the annual fee from Schedule III.A for the first year must also be paid at the time of application. For new applications under Schedule I.B and I.C, the annual fee is included in the new permit amount and covers the annual fee for the calendar year in which the permit coverage becomes effective.

(a) through (e) remain the same.

(f) Applications for new permits or permit renewals for sources that constitute a new or increased source, as defined in ARM 17.30.702~~(18)~~ (17), must pay a significance determination fee for each outfall in addition to the application fee.

(g) through (11)(b) remain the same.

AUTH: 75-5-516, MCA
IMP: 75-5-516, MCA

REASON: The amendment to ARM 17.30.201(6)(f) modifies a cross-reference to ARM 17.30.702 because the numbering in that rule is proposed to be changed in this notice.

17.30.507 SPECIFIC RESTRICTIONS FOR SURFACE WATER MIXING ZONES (1) Mixing zones for surface waters are ~~to comply with~~ subject to the following water quality standards:

(a) narrative water quality standards, standards for harmful substances, numeric acute and chronic standards for aquatic life~~,~~; standards in Department Circular DEQ-12A; and standards based on human health must not be exceeded beyond the boundaries of the surface water mixing zone;

(b) through (3) remain the same.

AUTH: 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

REASON: The amendment to this rule is necessary to ensure that mixing zones are available for nutrient standards and to ensure that the nutrient standards must be met beyond the mixing zone. A mixing zone is a nationally recognized and useful tool to implement standards in permits, and there is no reason that this tool should not be available for nutrient standards.

17.30.516 STANDARD MIXING ZONES FOR SURFACE WATER (1) and (2) remain the same.

(3) Facilities that meet the terms and conditions in (a) through ~~(d)~~ (e) qualify for a standard mixing zone as follows:

(a) through (d) remain the same.

(e) Facilities that discharge the parameters found in Department Circular DEQ-12A to surface water. Discharge limitations must be based on dilution with the entire seasonal 14-day, five-year (seasonal 14Q5) low flow of the receiving water without the discharge.

(4) The length of a standard mixing zone for flowing surface water, other than a nearly instantaneous mixing zone, must not extend downstream more than the one-half mixing width distance or extend downstream more than ~~10~~ ten times the stream width, whichever is more restrictive. For purposes of making this determination, the stream width as well as the discharge limitations are considered at the 7Q10 or seasonal 14Q5 low flow. The seasonal 14Q5 low flow may be used only in conjunction with base numeric nutrient standards in Department Circular DEQ-12A. The recommended calculation to be used to determine the one-half mixing width distance downstream from a stream bank discharge is described below.

(a) $A_{1/2} = [0.4(W/2)^2V]/L$, where:

(i) remains the same.

(ii) W = width in feet at the 7Q10 or seasonal 14Q5;

(iii) V = velocity of the stream at the 7Q10 or seasonal 14Q5 downstream of the discharge (in ft/second);

(iv) L = lateral dispersion coefficient for the 7Q10 or seasonal 14Q5 downstream of the discharge (in ft^2/second), where:

(b) L = CDU, where:

(i) through (i)(E) remain the same.

(ii) D = average water depth at the 7Q10 or seasonal 14Q5 downstream of the discharge (in feet);

(iii) remains the same.

(c) $U = (32.2DS)^{1/2}$, where:

(i) remains the same.

(ii) D = average water depth at the 7Q10 or seasonal 14Q5 downstream of the discharge (in feet); and

(iii) through (6) remain the same.

AUTH: 75-5-301, MCA
IMP: 75-5-301, MCA

REASON: The manner in which nutrients affect and impact beneficial uses in streams and rivers is different from toxic and harmful compounds found in Department Circular DEQ-7 (DEQ-7), and it is necessary to develop an appropriate low flow design flow (the seasonal 14Q5) specifically for permitting nutrient discharges. Derivation of the seasonal 14Q5 is discussed in the proposed changes to ARM 17.30.635. Here, the rule amendments incorporate the seasonal 14Q5 flow into the calculations used to determine the length of a standard mixing zone. ARM 17.30.516 is proposed to be amended to provide that the full volume of a seasonal 14Q5, as opposed to some fraction of it, is to be used for dilution calculations for nutrients in DEQ-12A. This allowance reflects the non-toxic nature of nutrients at the concentrations found in DEQ-12A.

17.30.602 DEFINITIONS In this subchapter the following terms have the meanings indicated below and are supplemental to the definitions given in 75-5-103, MCA:

(1) through (32) remain the same.

(33) "Total nitrogen" means the ~~total nitrogen concentration (as N) of unfiltered water. This may be determined by direct methods, or derived as the sum of the soluble (as N) and non-soluble (as N) nitrogen fractions. The filter used to separate the soluble and non-soluble fractions must be 0.45 µm~~ sum of all nitrate, nitrite, ammonia, and organic nitrogen, as N, in an unfiltered water sample. Total nitrogen in a sample may also be determined by the persulfate digestion or as the sum of total kjeldahl nitrogen plus nitrate plus nitrite.

(34) "Total phosphorus" means the ~~total phosphorus concentration (as P) of unfiltered water~~ sum of orthophosphates, polyphosphates, and organically bound phosphates, as P, in an unfiltered water sample. Total phosphorus may also be determined directly by persulfate digestion.

(35) through (38) remain the same.

(39) "DEQ-7" means the department circular that is adopted and incorporated by reference in ARM 17.30.619 and is entitled "Montana Numeric Water Quality Standards." This circular establishes water quality standards for toxic, carcinogenic, ~~bioconcentration~~ bioconcentrating, ~~nutrient,~~ radioactive, and harmful parameters, and also establishes human health-based water quality standards for the following specific nutrients with toxic effects:

(a) nitrate;

(b) nitrate + nitrite; and

(c) nitrite.

(40) "DEQ-12A" means the department circular that is adopted and incorporated by reference in ARM 17.30.619 and is entitled "Montana Base Numeric Nutrient Standards." This circular contains numeric water quality standards for total nitrogen and total phosphorus in surface waters.

(41) "DEQ-12B" means the department circular that is adopted and that is entitled "Montana Base Numeric Nutrient Standards Variances." This circular describes procedures for receiving a variance from the standards and will document recipients of individual variances.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

REASON: The proposed amendments to ARM 17.30.602 provide modification of existing definitions and a new definition in order to implement the nutrient standards. The modified definition of "total nitrogen," at (33), provides a more technically accurate description compared to the old definition. The same is true for "total phosphorus," at (34). In the definition for "DEQ-7," at (39), "nutrient" has been removed because base numeric nutrient standards will now be housed in a new department circular, DEQ-12A. Some nitrogen compounds (nitrate, nitrate + nitrite, and nitrite) have toxic effects at relatively high concentrations and standards for them already exist and are intended to protect human health. By definition at 75-

5-103(2)(b), MCA, these compounds are not considered part of the base numeric nutrients standards. Therefore, they will remain in DEQ-7 and are now listed under the DEQ-7 definition for better clarity. The new definition at (40), "DEQ-12A," defines the new department circular where base numeric nutrient standards are found. In addition to the criteria concentrations, the circular includes instructions on how to develop permits for base numeric nutrient standards. In MAR Notice No. 17-355, the department is proposing to adopt new Department Circular DEQ-12B. It contains the procedures for receiving a variance from the standards and will document recipients of individual variances. The board anticipates that DEQ-12B will be adopted before or at the same time DEQ-12A is adopted.

17.30.619 INCORPORATIONS BY REFERENCE (1) The board adopts and incorporates by reference the following state and federal requirements and procedures as part of Montana's surface water quality standards:

(a) Department Circular DEQ-7, entitled "Montana Numeric Water Quality Standards" (October 2012 edition), which establishes water quality standards for toxic, carcinogenic, bioconcentrating, ~~nutrient,~~ radioactive, and harmful parameters and also establishes human health-based water quality standards for the following specific nutrients with toxic effects:

(i) nitrate;

(ii) nitrate + nitrite; and

(iii) nitrite;

(b) remains the same.

(c) 40 CFR Part 136 (July 1, 2011), which establishes guidelines and procedures for the analysis of pollutants; ~~and~~

(d) 40 CFR 131.10(g), (h) and (j) (2000), which establishes criteria and guidelines for conducting a use attainability analysis~~.~~; and

(e) Department Circular DEQ-12A, entitled "Montana Base Numeric Nutrient Standards" (December 2013 edition), which establishes numeric water quality standards for total nitrogen and total phosphorus in surface waters.

(2) If a court of competent jurisdiction declares 75-5-313, MCA, or any portion of that statute invalid, or if the United States Environmental Protection Agency disapproves 75-5-313, MCA, or any portion of that statute, under 30 CFR 131.21, or if rules adopted pursuant to 75-5-313(6) or (7), MCA, expire and general variances are not available, then (1)(e) and all references to DEQ-12A, base numeric nutrient standards and nutrient standards variances in ARM 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.622 through 17.30.629, 17.30.635, 17.30.702, and 17.30.715 are void, and the narrative water quality standards contained in ARM 17.30.637 are the standards for total nitrogen and total phosphorus in surface water, except for the Clark Fork River, for which the standards are the numeric standards in ARM 17.30.631.

(2) remains the same, but is renumbered (3).

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

REASON: The amendments to the definitions for DEQ-7, in (1)(a), correspond to those already discussed above for definitions (ARM 17.30.602). Proposed new (2) is a non-severability clause. Essentially, if the statute that defines the nutrient standards variance process is rendered invalid, or if general variance rules expire and general variances are not available, then the base numeric nutrient standards would no longer be contained in the rules. The Legislature intended that variances be available to permittees once base numeric nutrient standards were adopted and both pieces (base numeric standards and variances) must remain together as a package.

17.30.622 A-1 CLASSIFICATION STANDARDS (1) and (2) remain the same.

(3) No person may violate the following specific water quality standards for waters classified A-1:

(a) through (g) remain the same.

(h) Concentrations of carcinogenic, bioconcentrating, toxic, radioactive, nutrient or harmful parameters may not exceed the applicable standards set forth in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A.

(i) Dischargers issued permits under ARM Title 17, chapter 30, subchapter 13, shall conform with ARM Title 17, chapter 30, subchapter 7, the nondegradation rules, and may not cause receiving water concentrations to exceed the applicable standards contained in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A when stream flows equal or exceed the design flows specified in ARM 17.30.635~~(4)~~ (2).

(j) and (k) remain the same.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

17.30.623 B-1 CLASSIFICATION STANDARDS (1) remains the same.

(2) No person may violate the following specific water quality standards for waters classified B-1:

(a) through (g) remain the same.

(h) Concentrations of carcinogenic, bioconcentrating, toxic, radioactive, nutrient, or harmful parameters may not exceed the applicable standards set forth in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A.

(i) Dischargers issued permits under ARM Title 17, chapter 30, subchapter 13, shall conform with ARM Title 17, chapter 30, subchapter 7, the nondegradation rules, and may not cause receiving water concentrations to exceed the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A when stream flows equal or exceed the design flows specified in ARM 17.30.635~~(4)~~ (2).

(j) and (k) remain the same.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

17.30.624 B-2 CLASSIFICATION STANDARDS (1) remains the same.

(2) No person may violate the following specific water quality standards for waters classified B-2:

(a) through (g) remain the same.

(h) Concentrations of carcinogenic, bioconcentrating, toxic, radioactive, nutrient, or harmful parameters may not exceed the applicable standards set forth in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A.

(i) Dischargers issued permits under ARM Title 17, chapter 30, subchapter 13, shall conform with ARM Title 17, chapter 30, subchapter 7, the nondegradation rules, and may not cause receiving water concentrations to exceed the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A when stream flows equal or exceed the design flows specified in ARM 17.30.635~~(4)~~ (2).

(j) and (k) remain the same.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

17.30.625 B-3 CLASSIFICATION STANDARDS (1) remains the same.

(2) No person may violate the following specific water quality standards for waters classified B-3:

(a) through (g) remain the same.

(h) Concentrations of carcinogenic, bioconcentrating, toxic, radioactive, nutrient, or harmful parameters may not exceed the applicable standards set forth in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A.

(i) Dischargers issued permits under ARM Title 17, chapter 30, subchapter 13, shall conform with ARM Title 17, chapter 30, subchapter 7, the nondegradation rules, and may not cause receiving water concentrations to exceed the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A when stream flows equal or exceed the design flows specified in ARM 17.30.635~~(4)~~ (2).

(j) and (k) remain the same.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

17.30.626 C-1 CLASSIFICATION STANDARDS (1) remains the same.

(2) No person may violate the following specific water quality standards for waters classified C-1:

(a) through (g) remain the same.

(h) Concentrations of carcinogenic, bioconcentrating, toxic, radioactive, nutrient, or harmful parameters may not exceed the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A.

(i) Dischargers issued permits under ARM Title 17, chapter 30, subchapter 13, shall conform with ARM Title 17, chapter 30, subchapter 7, the nondegradation rules, and may not cause receiving water concentrations to exceed the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A when stream flows equal or exceed the design flows specified in ARM 17.30.635~~(4)~~ (2).

(j) and (k) remain the same.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

17.30.627 C-2 CLASSIFICATION STANDARDS (1) remains the same.

(2) No person may violate the following specific water quality standards for waters classified C-2:

(a) through (g) remain the same.

(h) Concentrations of carcinogenic, bioconcentrating, toxic, radioactive, nutrient, or harmful parameters may not exceed the applicable standards specified in ~~d~~Department Circular ~~WQB~~ DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A.

(i) Dischargers issued permits under ARM Title 17, chapter 30, subchapter 13, shall conform with ARM Title 17, chapter 30, subchapter 7, the nondegradation rules, and may not cause receiving water concentrations to exceed the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A when stream flows equal or exceed the design flows specified in ARM 17.30.635~~(4)~~ (2).

(j) and (k) remain the same.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

REASON: The proposed amendments to ARM 17.30.622 through 17.30.627 are necessary to incorporate DEQ-12A standards and nutrient standards variance limits into the surface water classes.

17.30.628 I CLASSIFICATION STANDARDS (1) remains the same.

(2) No person may violate the following specific water quality standards for waters classified I:

(a) through (i) remain the same.

(j) Beneficial uses are considered supported when the concentrations of toxic, carcinogenic, nutrient, or harmful parameters in these waters do not exceed the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A when stream flows equal or exceed the flows specified in ARM 17.30.635~~(4)~~ (2) or, alternatively, for aquatic life when site-specific criteria are adopted using the procedures given in 75-5-310, MCA. The limits shall be used as water quality standards for the affected waters and as the basis for permit limits instead of the applicable standards in ~~d~~Department Circular DEQ-7.

(k) Limits for toxic, carcinogenic, or harmful parameters in new discharge permits issued pursuant to the MPDES rules (ARM Title 17, chapter 30, subchapter 13) are the larger of ~~either~~ the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A, site-specific standards, or one-half of the mean in-stream concentrations immediately upstream of the discharge point.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

REASON: The proposed amendment to ARM 17.30.628 is necessary to incorporate DEQ-12A and the nutrient standards variance limits into the I surface water class. I Class waterbodies are those which had severe human-caused pollution problems at the time the surface water class system was adopted in the 1970s, and it is the board's intent that these waterbodies will eventually support beneficial uses typical for ecologically similar, unimpacted waterbodies.

17.30.629 C-3 CLASSIFICATION STANDARDS (1) remains the same.

(2) No person may violate the following specific water quality standards for waters classified C-3:

(a) through (g) remain the same.

(h) Concentrations of carcinogenic, bioconcentrating, toxic, radioactive, nutrient, or harmful parameters may not exceed the applicable standards set forth in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A.

(i) Dischargers issued permits under ARM Title 17, chapter 30, subchapter 13, shall conform with ARM Title 17, chapter 30, subchapter 7, the nondegradation rules, and may not cause receiving water concentrations to exceed the applicable standards specified in ~~d~~Department Circular DEQ-7 and, unless a nutrient standards variance has been granted, Department Circular DEQ-12A when stream flows equal or exceed the design flows specified in ARM 17.30.635~~(4)~~ (2).

(j) and (k) remain the same.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, 75-5-313, MCA

REASON: The proposed amendments to ARM 17.30.629 are necessary to incorporate DEQ-12A standards and nutrient variance limits into the C-3 surface water class.

17.30.635 GENERAL TREATMENT STANDARDS (1) through (1)(e) remain the same.

(2) For design of disposal systems, stream flow dilution requirements must be based on the minimum consecutive seven-day average flow which may be expected to occur on the average of once in ten years. When dilution flows are less than the above design flow at a point discharge, the discharge is to be governed by the permit conditions developed for the discharge through the waste discharge

permit program. If the flow records on an affected surface water are insufficient to calculate a ten-year seven-day low flow, the department shall determine an acceptable stream flow for disposal system design. ~~The department shall determine the acceptable stream flow for disposal system design for controlling nitrogen and phosphorus concentrations.~~ For total nitrogen and total phosphorus, the stream flow dilution requirements must be based on the seasonal 14Q5, which is the lowest average 14 consecutive day low flow, occurring from July through October, with an average recurrence frequency of once in five years.

(3) remains the same.

AUTH: 75-5-201, 75-5-301, MCA
IMP: 75-5-301, MCA

REASON: The proposed amendments to ARM 17.30.635 will provide a low flow for the design of disposal systems specific to eutrophication-based nutrient standards. Work by the department and others shows that nuisance benthic algae can develop in about 15-20 days once nutrient concentrations exceed the proposed standards. In many streams, these algae levels can ultimately lead to dissolved oxygen impacts. The use of the seasonal 14Q5 flow for the design of disposal systems is appropriate because this flow should not allow excess algae levels to develop more often than about once in five summers, on average. This frequency of exceedence is within the acceptable recommendations of the U.S. Environmental Protection Agency for the protection of aquatic life. Unlike the 7Q10 flow, which will continue to be used for parameters in DEQ-7 and which was derived from year-round flow data, the seasonal 14Q5 flow is derived from July through October data and is, therefore, in alignment with the proposed nutrient standards' periods of application. The seasonal 14Q5 is routinely calculated and reported by the U.S. Geological Survey.

17.30.702 DEFINITIONS The following definitions, in addition to those in 75-5-103, MCA, apply throughout this subchapter (Note: 75-5-103, MCA, includes definitions for "base numeric nutrient standards," "degradation," "existing uses," "high quality waters," "mixing zone," and "parameter"):

(1) through (16) remain the same.

~~(17) "Nutrients" means total inorganic phosphorus and total inorganic nitrogen.~~

(18) through (21) remain the same, but are renumbered (17) through (20).

~~(22)~~ (21) "Reporting values (RRV)" means the detection level that must be achieved in reporting surface water or ground water monitoring or compliance data to the department unless otherwise specified in a permit, approval, or authorization issued by the department. The RRV is the ~~department's~~ board's best determination of a level of analysis that can be achieved by the majority of commercial, university, or governmental laboratories using EPA approved methods or methods approved by the department. The RRV is listed in Department Circular DEQ-7, Department Circular DEQ-12A, and in the definition of "total inorganic phosphorus."

(23) remains the same, but is renumbered (22).

(23) "Total nitrogen" means the sum of all nitrate, nitrite, ammonia, and

organic nitrogen, as N, in an unfiltered water sample. Total nitrogen in a sample may also be determined by persulfate digestion, or as the sum of total kjeldahl nitrogen plus nitrate plus nitrite.

(24) "Total phosphorus" means the sum of orthophosphates, polyphosphates, and organically bound phosphates, as P, in an unfiltered water sample. Total phosphorus may also be determined directly by persulfate digestion.

(24) and (25) remain as proposed, but are renumbered (25) and (26).

~~(26)~~ (27) The board adopts and incorporates by reference:

(a) Department Circular DEQ-7, entitled "Montana Numeric Water Quality Standards" (October 2012 edition), which establishes water quality standards for toxic, carcinogenic, bioconcentrating, ~~nutrient,~~ radioactive, and harmful parameters and also establishes human health-based water quality standards for the following specific nutrients with toxic effects:

(i) nitrate;

(ii) nitrate + nitrite; and

(iii) nitrite;

(b) Department Circular DEQ-12A, entitled "Montana Base Numeric Nutrient Standards" (December 2013 edition), which establishes numeric water quality standards for total nitrogen and total phosphorus in surface waters;

(b) through (d) remain the same, but are renumbered (c) through (e).

AUTH: 75-5-301, 75-5-303, MCA
IMP: 75-5-303, MCA

REASON: The proposed amendments to ARM 17.30.702 will modify current definitions in the nondegradation rules and will add new definitions necessary for the implementation of base numeric nutrient standards. "Base numeric nutrients standards" have been added to the list of definitions from 75-5-103, MCA, that are incorporated by reference. The current definition of "nutrients," at (17), is being repealed, because it is not consistent with the use of the term in DEQ-12A, which contains standards for total nutrients. Further, the definition of "nutrients" added no clear value to the nondegradation rules, because, where needed, specific nutrient compounds or forms (e.g., TKN, nitrate as N) are named or referenced in the nondegradation rules. The proposed definitions of "total nitrogen," at (24), and "total phosphorus," at (25), correspond to those discussed above for amendments to ARM 17.30.602. The definition of "DEQ-7," in (28)(b), has been amended for the same reasons described above for ARM 17.30.602.

17.30.715 CRITERIA FOR DETERMINING NONSIGNIFICANT CHANGES IN WATER QUALITY (1) The following criteria will be used to determine whether certain activities or classes of activities will result in nonsignificant changes in existing water quality due to their low potential to affect human health or the environment. These criteria consider the quantity and strength of the pollutant, the length of time the changes will occur, and the character of the pollutant. Except as provided in (2), changes in existing surface or ground water quality resulting from the activities that meet all the criteria listed below are nonsignificant, and are not required to undergo review under 75-5-303, MCA:

(a) and (b) remain the same.

(c) discharges containing toxic parameters, inorganic nitrogen, or inorganic phosphorus ~~or nutrients~~, except as specified in (1)(d) and (e), which will not cause changes that equal or exceed the trigger values in ~~d~~Department Circular DEQ-7. Whenever the change exceeds the trigger value, the change is not significant if the resulting concentration outside of a mixing zone designated by the department does not exceed 15% of the lowest applicable standard;

(d) through (e) remain the same.

(f) changes in the quality of water for any harmful parameter, including parameters listed in Department Circular DEQ-12A, for which water quality standards have been adopted other than ~~nitrogen, phosphorous, and~~ carcinogenic, bioconcentrating, or toxic parameters, in either surface or ground water, if the changes outside of a mixing zone designated by the department are less than 10% of the applicable standard and the existing water quality level is less than 40% of the standard;

(g) through (3) remain the same.

(4) If a court of competent jurisdiction declares 75-5-313, MCA, or any portion of that statute invalid or if the United States Environmental Protection Agency disapproves 75-5-313, MCA, or any portion of that statute under 30 CFR 131.21, then the significance criteria contained in (1)(g) are the significance criteria for total nitrogen and total phosphorus in surface water.

AUTH: 75-5-301, 75-5-303, MCA
IMP: 75-5-303, MCA

REASON: The proposed amendments to ARM 17.30.715 will allow the department to calculate nonsignificant changes in water quality for the base numeric nutrient standards in DEQ-12A. If adopted by the board, base numeric nutrient standards will preclude the need to use the narrative standards at ARM 17.30.637(1)(e) to interpret eutrophication-based water quality impacts from nutrients. Base numeric nutrient standards are intended to control eutrophication and, at the concentrations found in DEQ-12A, the board considers base numeric nutrient standards to be harmful parameters. Therefore, DEQ-12A is incorporated into (1)(f), the section of the nondegradation rules addressing nonsignificance specific to harmful parameters. Nitrogen compounds at concentrations that are toxic, e.g., nitrate at ten mg/L, will remain in DEQ-7, as discussed earlier, and toxics-based nonsignificance criteria applicable to such compounds will continue to be applied to them. The proposed deletion of "or nutrients," in (1)(c), corresponds with the retaining of toxic-level nitrogen compounds in DEQ-7 and the relocation of eutrophication-based nitrogen and phosphorus standards to DEQ-12A. In addition, the term "or nutrients" in (1)(c) has been replaced with "or total inorganic phosphorus or total inorganic nitrogen," for the specific purpose of providing a nonsignificance threshold for nondegradation review of new dischargers, which are commonly subdivisions. This change allows the department to continue to carry out these reviews in the same manner as currently practiced, because DEQ-7 provides a trigger value for both of these inorganic compounds. ARM 17.30.715(1)(c) also provides: "Whenever the change exceeds the trigger value, the change is not significant if the resulting concentration outside of a mixing zone designated by the department does not exceed 15% of the lowest applicable standard." When these provisions become applicable, the "lowest applicable standard" would be the narrative standard contained in ARM 17.30.637(1)(e). Significance would then be determined under ARM 17.30.715(1)(g). Proposed new (4) is a non-severability clause. If the statute that defines the nutrient standards variance process is rendered invalid, then the numeric nutrient standards in DEQ-12A are void and the narrative standard for nutrients at ARM 17.30.637(1)(e) applies. As a result, the part of the nondegradation rules at ARM 17.30.715(1)(g) that relate to the narrative standards would apply. The Legislature intended that both major pieces of the numeric nutrient standards rules (base numeric nutrient standards and nutrient standards variances) remain together as a package.

5. The proposed new circular may be viewed at and copied from the department's web site at http://deq.mt.gov/wqinfo/Standards/default.mcpx. Also, copies may be obtained by contacting Carrie Greeley at Department of Environmental Quality, P.O. Box 200901, Helena, MT 59620-0901; by phone at (406) 444-6749; or by e-mail at CGreeley@mt.gov.

6. Concerned persons may submit their data, views, or arguments, either orally or in writing, at the hearing. Written data, views, or arguments may also be submitted to Elois Johnson, Paralegal, Department of Environmental Quality, 1520 E. Sixth Avenue, P.O. Box 200901, Helena, Montana 59620-0901; faxed to (406) 444-4386; or e-mailed to ejohnson@mt.gov, no later than 5:00 p.m., April 1, 2014. To be guaranteed consideration, mailed comments must be postmarked on or before that date.

7. Katherine Orr, attorney for the board, or another attorney for the Agency Legal Services Bureau, has been designated to preside over and conduct the hearing.

8. The board maintains a list of interested persons who wish to receive notices of rulemaking actions proposed by this agency. Persons who wish to have their name added to the list shall make a written request that includes the name, e-mail, and mailing address of the person to receive notices and specifies that the person wishes to receive notices regarding: air quality; hazardous waste/waste oil; asbestos control; water/wastewater treatment plant operator certification; solid waste; junk vehicles; infectious waste; public water supply; public sewage systems regulation; hard rock (metal) mine reclamation; major facility siting; opencut mine reclamation; strip mine reclamation; subdivisions; renewable energy grants/loans; wastewater treatment or safe drinking water revolving grants and loans; water quality; CECRA; underground/above ground storage tanks; MEPA; or general procedural rules other than MEPA. Notices will be sent by e-mail unless a mailing preference is noted in the request. Such written request may be mailed or delivered to Elois Johnson, Paralegal, Department of Environmental Quality, 1520 E. Sixth Ave., P.O. Box 200901, Helena, Montana 59620-0901, faxed to the office at (406) 444-4386, e-mailed to Elois Johnson at ejohnson@mt.gov, or may be made by

completing a request form at any rules hearing held by the board.

9. The bill sponsor contact requirements of 2-4-302, MCA, do not apply.

10. With regard to the requirements of 2-4-111, MCA, the department has determined that the amendment of the above-referenced rules will significantly and directly impact small businesses.

Reviewed by: BOARD OF ENVIRONMENTAL REVIEW

/s/ John F. North BY: */s/ Robin Shropshire*
JOHN F. NORTH ROBIN SHROPSHIRE
Rule Reviewer Chairman

Certified to the Secretary of State, February 3, 2014.

**BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
OF THE STATE OF MONTANA**

IN THE MATTER OF THE AMENDMENT OF ARM 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.619, 17.30.622, 17.30.623, 17.30.624, 17.30.625, 17.30.626, 17.30.627, 17.30.628, 17.30.629, 17.30.635, 17.30.702, AND 71.30.715 pertaining to permit application, degradation authorization, and annual permit fees, specific restrictions for surface water mixing zones, standard mixing zones for surface water, definitions, incorporations by reference, A-1 classification standards, B-1 classification standards, B-2 classification standards, B-3 classification standards, C-1 classification standards, C-2 classification standards, C-2 classification standards, I classification standards, C-3 classification standards, general treatment standards, definitions, and criteria for determining nonsignificant changes in water quality	PRESIDING OFFICER REPORT

On March 24, 2014, the undersigned presided over and conducted the public hearing held in Room 111 of the Metcalf Building, 1520 East Sixth Avenue, Helena, Montana, to take public comment on the above-captioned proposed amendments pertaining to adoption of numeric nutrient standards for Montana water bodies such as wadeable streams and large rivers for nitrogen and phosphorus.

1. The Notice of Public Hearing on Proposed Amendment (Water Quality) MAR Notice No. 17-356 was published on February 13, 2014. A copy of the Notice Of Public Hearing On Proposed Amendment is attached to this report. (Attachments are provided in the same order as they are referenced in this report.)

2. The hearing began at 2 p.m. Ms. Cheryl Romsa of Cheryl Romsa Court Reporting was the Court Reporter for the hearing.

3. The undersigned announced that persons at the hearing would be given an opportunity to submit their data, views, or arguments concerning the proposed action, either orally or in writing. Details of where to submit written views or arguments were provided. At the hearing, the undersigned identified the MAR notice and read the Notice of Function of Administrative Rule Review Committee as required by Mont. Code Ann. § 2-4-302(7)(a). The rulemaking interested persons list and the opportunity to have names placed on that list were addressed. The Presiding Officer explained the order of presentation.

SUMMARY OF HEARING

4. Dr. Michael Suplee, a limnologist in the water Quality Standards Unit of the Department of Environmental Quality (Department) gave a statement describing the rule amendments that adopt numeric nutrient standards including standards in circulars and stating that the Department was recommending that the amendments be adopted.

5. At the hearing, there were ten persons who provided oral comments. The list of these persons is as follows: Mr. Dave Mumford of the League of Cities and Towns, Mr. Dave Galt of the Montana Petroleum Association, Ms. Tammy Johnson of the Montana Mining Association, Ms. Victoria Marquis of Arch Coal, Mr. James Schell Mayor of East Helena, Mr. Randall Camp, the Public Works Director of the City of Helena, Mr. Chris Brick of the Clark Fork Coalition, Mr. Brian Sugden of Plum Creek Timber and Mr. John Wilson of the City of Whitefish.

The commenters were generally supportive of the effort. Proponents were Mr. Mumford and Ms. Brick. Opponents at least in part were Mr. Galt expressing concerns about various subjects including economic implications of compliance, the inadequacy of the severability clause in the case that a variance is nullified, the scope of protection of downstream uses and of responsibility of dischargers beyond the first location of loading, contributions from point source dischargers being the only discharges addressed through the rules and circulars. He stated the definitions in DEQ-12A and 12B are unclear as well as monthly and annual averages.

Ms. Johnson stated it isn't tenable to allow the nutrient standards to remain without an effective variance process available to all dischargers and there is a lack of clarity as to the interplay of the non-degradation statutes which apply to a new or increased source and the numeric standards and variance rulemaking.

Ms. Marquis stated the rule amendments will have adverse and costly impacts on their permitting process for the Otter Creek Project. It is not clear how the rules will impact storm water permits or how the process will work with the TMDL process. Ms. Marquis stated the technology necessary to meet the numeric standards is expensive and in some cases there may not be a cost-effective treatment available at all.

In summary form, Mr. Sugden stated that the numeric standards proposed for wadeable streams in Montana appear reasonable and are supported by good science and sound rationale. He stated there should be a functional non-severability clause in the rules that ensures the integrity of the overall program. There should be adaptability in the rules with future modifications possibly necessary. He said the assessment method that the Department uses to determine compliance with the standards should work together with the circulars that are proposed and the variance process. Mr. Sugden supports postponing the adoption of standards for Flathead Lake pending a more thorough technical review. Mr. Wilson commented on the extreme cost of the regulations and the need for outreach and education regarding the content of the rules.

SUMMARY OF WRITTEN MATERIALS

6. After the hearing, written comments were received by the Department from Michael Suplee, MT DEQ; Dave Galt, MT Petroleum Association; Victoria Marquis, Arch Coal; John North, DEQ legal, MT DEQ; Carla Parks, Mayor, City of

Thompson Falls; Jon Cuthbertson, Director, MT Environmental Laboratory, LLC; Mike Roland, Otter Creek Coal; Mark Vander Meer, Watershed, Consulting LLC; Bruce Kania, CEO, Floating Islands Int.; Anna Kania, BioHaven, Inc.; Robin Steinkraus, Ex. Dir., Flathead Lakers; Tammy Johnson, Ex. Dir., MT Mining Association; James Thomas, Tight Squeeze Farm; Tina Laidlaw, U.S. EPA; Chris Brick, Clark Fork Coalition; Guy Alsentzer, Esq., Upper Missouri Water Keeper; Mark Reinsel, PhD, PE, Apex Engineering PLLC; Randy Weimer, Stillwater Mining Company; Jon Harvala, Missoula Valley WQ District; Frank Stewart, Stewart Engineering; Robin Steinkraus, Ex. Dir., Flathead Lakers; Brian Sugden, Plum Creek Timber. The written comments of these individuals are attached.

7. The Department also submitted a memorandum from Department staff attorney, Mr. John North with HB 521 and HB 311 reviews of the proposed amendments and a Private Property Assessment Act Checklist. Mr. North's memorandum is attached to this report. The criteria and procedures proposed for MAR Notice No. 17-356 are not more stringent than EPA guidlines. Therefore, no further HB 521 analysis is required.

8. With respect to HB 311 (the Private Property Assessment Act, Mont. Code Ann. §§ 2-10-101 through 105), the State of Montana is required to assess the taking or damaging implications of a proposed rule affecting the use of private real property. This rulemaking affects the use of private real property. A Private Property Assessment Act Checklist was prepared, which shows that the proposed amendment and new rules do not have taking or damaging implications. Therefore, no further assessment is required.

9. The period to submit comments ended at 5 p.m. on April 1, 2014.

PRESIDING OFFICER COMMENTS

10. The Board has jurisdiction to adopt, amend, or repeal the amendment pursuant to Mont. Code Ann. §§ 75-5-516, 75-5-301 and 75-5-303.

11. House Bill 521 (1995) generally provides that the Board may not adopt a rule that is more stringent than comparable federal regulations or guidelines, unless the Board makes written findings after public hearing and comment. The proposed amendments are not more stringent than a comparable federal regulation or guideline.

12. House Bill 311 (1995), the Private Property Assessment Act, codified as Mont. Code Ann. § 2-10-101 through -105, provides that a state agency must complete a review and impact assessment prior to taking an action with taking or damaging implications. The proposed amendments affect real property. A Private Property Assessment Act Checklist was prepared in this matter. The proposed amendments do not have taking or damaging implications. Therefore, no further HB 311 assessment is necessary.

13. The procedures required by the Montana Administrative Procedure Act, including public notice, hearing, and comment upon belief have been followed.

14. The Board and Department may adopt the proposed rule amendment, reject it or adopt the rule amendment with revisions not exceeding the scope of the public notice.

15. Under Mont. Code Ann. § 2-4-305(7), for the rulemaking process to be valid, the Board must publish a notice of adoption within six months of the date the Board published the notice of proposed rulemaking in the Montana Administrative Register, or by August 13, 2014.

Dated this __27__ day of July, 2014.

KATHERINE J. ORR
Presiding Officer

~~Version 6.8~~

Montana Department of
ENVIRONMENTAL QUALITY

DEPARTMENT CIRCULAR

DEQ-12A

Montana Base Numeric Nutrient Standards

GENERAL INTRODUCTION

This circular (DEQ-12A) contains information pertaining to the base numeric nutrients standards (§75-5-103(2), MCA) and their implementation. This information includes the standards' concentration limits, where the standards apply, and their period of application. DEQ-12A is adopted by the Board of Environmental Review under its rulemaking authority in §75-5-301(2), MCA.

Circular DEQ-12B contains information about variances from the base numeric nutrient standards and is a separate document available from the Department. DEQ-12B addresses effluent treatment requirements associated with general nutrient standards variances, as well as effluent treatment requirements for individual nutrient standards variances and to whom these apply. Unlike DEQ-12A, DEQ-12B is not adopted by the Board of Environmental Review; DEQ-12B is adopted by the Department following its formal rulemaking process, pursuant to §75-5-313, MCA.

The Department has reviewed a considerable amount of scientific literature and has carried out scientific research on its own in order to derive the base numeric nutrient standards (see **References** in this circular). Because many of the base numeric nutrient standards are stringent and may be difficult for MPDES permit holders to meet in the short term, Montana's Legislature adopted laws (e.g., §75-5-313, MCA) allowing for the achievement of the standards over time via the variance procedures in Circular DEQ-12B. This approach should allow time for nitrogen and phosphorus removal technologies to improve and become less costly and to allow time for nonpoint sources of nitrogen and phosphorus pollution to be better addressed.

Circular DEQ-12A

~~DECEMBER~~ JULY 2013~~3~~4 EDITION

1.0 Introduction

Elements comprising Circular DEQ-12A are found below. These elements are adopted by the Montana Board of Environmental Review. The nitrogen and phosphorus concentrations provided here have been set at levels that will protect beneficial uses and prevent exceedences of other surface water quality standards which are commonly linked to nitrogen and phosphorus concentrations (e.g., pH and dissolved oxygen; see Circular DEQ-7 for those standards). The nitrogen and phosphorus concentrations provided here also reflect the intent of the narrative standard at ARM 17.30.637(1)(e) and will preclude the need for case-by-case interpretations of that standard in most cases.

1.1 Definitions

1. **Ecoregion** means mapped regions of relative homogeneity in ecological systems derived from perceived patterns of a combination of causal and integrative factors including land use, land surface form, potential natural vegetation, soils, and geology. See also Endnote 1.

2. **Large river** means a perennial waterbody which has, during summer and fall baseflow (August 1 to October 31 each year), a wadeability index (product of river depth [in feet] and mean velocity [in ft/sec]) of 7.24 ft^2/sec or greater, a depth of 3.15 ft or greater, or a baseflow annual discharge of 1,500 ft^3/sec or greater. See also, Endnote 6.

3. **Total nitrogen** means the sum of all nitrate, nitrite, ammonia, and organic nitrogen, as N, in an unfiltered water sample. Total nitrogen in a sample may also be determined via persulfate digestion or as the sum of total kjeldahl nitrogen plus nitrate plus nitrite.

4. **Total phosphorus** means the sum of orthophosphates, polyphosphates, and organically bound phosphates, as P, in an unfiltered water sample. Total phosphorus may also be determined directly by persulfate digestion.

5. **Wadeable stream** means a perennial or intermittent stream in which most of the wetted channel is safely wadeable by a person during baseflow conditions.

2.0 Base Numeric Nutrient Standards

Table 12A-1 contains the base numeric nutrient standards for Montana's flowing waters. In **Table 12A-1** nutrient standards for wadeable streams are grouped by ecoregion, either at level III (coarse scale) or level IV (fine scale). Following the ecoregional standards is a list of wadeable streams with reach-specific standards. These waterbodies have characteristics disimilar from those of the ecoregions in which they reside and have therefore been provided reach-specific values. **For wadeable streams, the standards should be applied in this order: named stream reach first (if applicable) then level IV ecoregion (if applicable) then level III ecoregion.** **Table 12A-1** also contains a list of large river segments for which base numeric nutrient standards have been developed. Note that the ecoregional values in **Table 12A-1** do not apply to large rivers within those ecoregions. See Endnote 6 for a list of all large Montana rivers. If a particular large river reach is not listed in **Table 12A-1**, standards for it have not yet been developed.

Table 12A-2 ~~contains~~ is a placeholder table for future base numeric nutrient standards for Montana's lakes and reservoirs. The Department has not yet developed regional lake criteria, but it is expected that when they are developed they will be grouped by ecoregion. As such, placeholders for future ecoregionally-based criteria are provided in the table. The table also provides for lake-specific standards. The Department anticipates that reservoir standards will generally be developed case-by-case and, therefore, will be individually listed, as provided for in the table.

Table 12A-1. Base Numeric Nutrient Standards for Wadeable Streams in Different Montana Ecoregions. If standards have been developed for level IV ecoregions (subcomponents of the level III ecoregions) they are shown in italics below the applicable level III ecoregion. Individual reaches are in the continuation of this table.

			Numeric Nutrient Standard[4]	
Ecoregion[1,2] (level III or IV) and Number	**Ecoregion Level**	**Period When Criteria Apply[3]**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**
Northern Rockies (15)	III	July 1 to September 30	25	275
Canadian Rockies (41)	III	July 1 to September 30	25	325
Idaho Batholith (16)	III	July 1 to September 30	25	275
Middle Rockies (17)	III	July 1 to September 30	30	300
Absaroka-Gallatin Volcanic Mountains (17i)	IV	July 1 to September 30	105	250
Northwestern Glaciated Plains (42)	III	June 16 to September 30	110	1300
Sweetgrass Upland (42l), Milk River Pothole Upland (42n), Rocky Mountain Front Foothill Potholes (42q), and Foothill Grassland (42r)	IV	July 1 to September 30	80	560
Northwestern Great Plains (43) and Wyoming Basin (18)	III	July 1 to September 30	150	1300
River Breaks (43c)	IV	See Endnote 5	See Endnote 5	See Endnote 5
*Non-calcareous Foothill Grassland (43s), Shields-Smith Valleys (43t), Limy Foothill Grassland (43u), Pryor-Bighorn Foothills (43v), and Unglaciated Montana High Plains (43o)**	IV	July 1 to September 30	33	440

*For the Unglaciated High Plains ecoregion (43o), criteria only apply to the polygon located just south of Great Falls, MT.

[1] See Endnote 1

[2] See Endnote 2

[3] See Endnote 3

[4] See Endnote 4

Table 12A-1, Continued. Base Numeric Nutrient Standards for Individual Wadeable Streams (and Wadeable-stream Reaches), and Large-river Reaches.

		Numeric Nutrient Standard[4]	
Individual Stream or Reach Description[2]	**Period When Criteria Apply[3]**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**
Wadeable Streams: Clark Fork River basin			
Flint Creek, from Georgetown Lake outlet to the ecoregion 17ak boundary (46.4002, -113.3055)	July 1 to September 30	72	500
Wadeable Streams: Gallatin River basin			
Bozeman Creek, from headwaters to Forest Service Boundary (45.5833, -111.0184)	July 1 to September 30	105	250
Bozeman Creek, from Forest Service Boundary (45.5833, -111.0184) to mouth at East Gallatin River	July 1 to September 30	76	270
Hyalite Creek, from headwaters to Forest Service Boundary (45.5833,-111.0835)	July 1 to September 30	105	250
Hyalalite Creek, from Forest Service Boundary (45.5833,-111.0835) to mouth at East Gallatin River	July 1 to September 30	90	260
East Gallatin River between Bozeman Creek and Bridger Creek confluences	July 1 to September 30	50	290
East Gallatin River between Bridger Creek and Hyalite Creek confluences	July 1 to September 30	40	300
East Gallatin River between Hyalite Creek and Smith Creek confluences	July 1 to September 30	60	290
East Gallatin River from Smith Creek confluence mouth (Gallatin River)	July 1 to September 30	40	300
Large Rivers[6]:			
Yellowstone River (Bighorn River confluence to Powder River confluence)	August 1 -October 31	55	655
Yellowstone River (Powder River confluence to stateline)	August 1 -October 31	95	815

[2] See Endnote 2

[3] See Endnote 3

[4] See Endnote 4

[6] See Endnote 6

Table 12A-2. Base Numeric Nutrient Standards and Other Standards for Lakes and Reservoirs.

Ecoregion[1] (level III) and Number, or Individual Lake or Reservoir Description	Period of Application	Numeric Nutrient Standard[7] Total Phosphorus (µg/L)	Total Nitrogen (µg/L)	Other Standards[8]
LAKES/RESERVOIRS by ecoregion:				
Middle Rockies (17)	Year-round	[]	[]	
Northern Rockies (15)	Year-round	[]	[]	
Canadian Rockies (41)	Year-round	[]	[]	
Idaho Batholith (16)	Year-round	[]	[]	
LAKE SPECIFIC CRITERIA:				
~~Flathead Lake~~[9]	Year-round	~~5.0~~ []	~~95~~ []	~~Secchi depth ≥ m during non-turbidity plume conditions. Phytoplankton chlorophyll *a* 1.0 µg/L, as an annual average, not to be exceeded more than once in any three year period, on average.~~
RESERVOIR SPECIFIC CRITERIA:				
	Year-round	[]	[]	

[1] See Endnote 1

[7] See Endnote 7

[8] See Endnote 8

[9] ~~See Endnote 9~~

2.1 Required Reporting Values for Base Numeric Nutrient Standards

Table 12A-3 presents the required reporting values (RRVs) for total phosphorus and total nitrogen, as well as the RRVs for nitrogen fractions that can be used to compute total nitrogen.

Table 12A-3. Required reporting values[a,b] for total nitrogen and phosphorus measurements.

Nutrient		Method of Measurement	Required Reporting Value
Total phosphorus		Persulfate digestion	3 µg/L
Total nitrogen		Persulfate digestion	70 µg/L
Total nitrogen	Sum of:	(a) total kjeldahl nitrogen	~~150~~ 225 µg/L
		(b) nitrate + nitrite	See RRVs below
Nitrate- as N			20 µg/L
Nitrite- as N			10 µg/L
Nitrate + Nitrite-as N			20 µg/L

[a] See definition for required reporting values found in footnote 19 of Department Circular DEQ-7.

[b] Concentrations in Table 12A-3 must be achieved unless otherwise specified in a permit, approval, or authorization issued by the Department (DEQ-7; ARM 17.30.702).

2.2 Developing Permit Limits for Base Numeric Nutrient Standards

For total nitrogen and total phosphorus, the critical low-flow for the design of disposal systems shall be based on the seasonal 14Q5 of the receiving water (ARM 17.30.635(2)). When developing permit limits for base numeric nutrient standards, the Department will use an average monthly limit (AML) only, based on a calendar month, using methods appropriate for criterion continuous concentrations (i.e., chronic concentrations). Permit limits will be established using a value corresponding to the 95th percentile probability distribution of the effluent. Nitrogen and phosphorus concentrations of the receiving waterbody upstream of the discharge may be characterized using other frequency distribution percentiles. The Department shall use methods that are appropriate for criterion continuous concentrations which are found in the document "*Technical Support Document for Water Quality-based Toxics Control*," Document No. EPA/505/2-90-001, United States Environmental Protection Agency, 1991.

3.0 Endnotes

(1) Ecoregions are based on the 2009 version (version 2) of the U.S. Environmental Protection Agency maps. These can be found at: http://www.epa.gov/wed/pages/ecoregions/mt_eco.htm . For Geographic Information System (GIS) use within the Department, the GIS layers may be found at: L:\DEQ\Layers\Reference\Ecoregions.lyr

(2) Within and among the geographic regions or watersheds listed, base numeric nutrient standards of the downstream reaches or other downstream waterbodies must continue to be maintained. Where possible, modeling methods will be utilized to determine the limitations required which provide for the attainment and maintenance of water quality standards of downstream waterbodies.

(3) For the purposes of ambient surface water monitoring and assessment only, a ten-day window (plus/minus) on the beginning and ending dates of the period when the criteria apply is allowed in order to accommodate year-specific conditions (an early-ending spring runoff, for example).

(4) The ~~30 day~~ average concentration during a period when the standards apply ~~of these parameters~~ may not ~~be~~ exceed~~ed~~ the standards more than once in any five-year period, on average.

(5) In this level IV ecoregion, the narrative standard for nuisance aquatic life (ARM 17.30.637(1)(e)) applies in lieu of specific base numeric nutrient standards.

(6) **Table E-1** below shows the beginning and ending locations for large rivers in Montana.

Table E-1. Large river segments within the state of Montana.

River Name	Segment Description
Big Horn River	Yellowtail Dam to mouth
Clark Fork River	Bitterroot River to state-line
Flathead River	Origin to mouth
Kootenai River	Libby Dam to state-line
Madison River	Ennis Lake to mouth
Missouri River	Origin to state-line
South Fork Flathead River	Hungry Horse Dam to mouth
Yellowstone River	State-line to state-line

(7) No lake or reservoir in **Table12A-2** shall have a total nutrient concentration that exceeds the values shown, as an annual average, more than once in any three year period, on average. The Department will determine on a case-by-case basis whether or not a permitted discharge to a stream or river is likely to be affecting any downstream lake or reservoir. If so, the permittee would be required to meet its average monthly nutrient limit year-round.

(8) Parameters listed under this column are standards specific to lakes and reservoirs.

~~(9) Standards and related assessment information (excluding Secchi depth) are to be determined from 0-30 m depth-integrated samples. Samples and Secchi depth measurements are to be collected at the Midlake Deep site which is located approximately 1 mile west of Yellow Bay Point in a pelagic area of the lake (approximately at latitude 47.861, longitude -114.067).~~

4.0 References

The following are citations for key scientific and technical literature used to derive the base numeric nutrient standards. This is not a complete list; rather, it contains the most pertinent citations. Many other articles and reports were reviewed during the development of the standards.

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Elser, J.J., M.E.S. Bracken, E.E. Cleland, D.S. Gruner, W.S. Harpole, H. Hillebrand, J.T. Ngai, E.W. Seabloom, J.B. Shurin, and J.E. Smith, 2007. Global Analysis of Nitrogen and Phosphorus Limitation of Primary Producers in Freshwater, Marine and Terrestrial Ecosystems. Ecology Letters 10: 1135-1142.

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From: Anne Kania <floatingisland@me.com>
Sent: Tuesday, April 01, 2014 12:47 PM
To: DEQ WQP Admin
Subject: Comments on Circular DEQ-12B
Attachments: DEQ Response-BioHaven.docx; Dodkins report swansea.pdf; Summaries sorted by type and date 4-01-14.pdf

Dear Carrie,

Please find attached a word document containing comments on the Variances Circular, and two supporting pdf documents.

Thank you,
Anne Kania

Anne Kania

BioHaven, Inc.
10052 Floating Island Way,
Shepherd, MT 59079

1-800 450 1088
PH: +1 406 373 5200
Cell: +1 406 208 5520
Skype name: LAMLOW

Ms. Carrie Greeley
Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901
deqwqpadmin@mt.gov

April 1, 2014

To Whom It May Concern:

Re: Circular DEQ-12A Nutrient Standards and DEQ-12B Variances

Thank you for the opportunity to comment on the proposed new Nutrient Standards and Variances. We appreciate the work DEQ is doing to protect the environment for future generations, especially our aquatic environment.

I am making these comments on behalf of BioHaven, Inc., a licensee of Floating Island International (FII). We are based in Montana and handle the business of sales and marketing in the Rocky Mountain States and Western Canada, in partnership with FII's other four licensed manufacturers and our network of distribution partners in Montana and the surrounding states.

We are the sole distributors of BioHaven floating islands, a type of constructed wetland that in our opinion can be considered a viable alternative solution to assist Montanans in meeting the proposed nutrient standards.

I wish to comment specifically on the Variances circular.

We note that, "Montana's Legislature adopted laws (e.g., §75-5-313, MCA) allowing for the achievement of the standards over time via the variance procedures found here in Circular DEQ-12B. *This approach should allow time for nitrogen and phosphorus removal technologies to improve and become less costly, and to allow time for nonpoint sources of nitrogen and phosphorus pollution to be better addressed."*

We believe that such a nitrogen and phosphorus removal technology is available now. I would like to take this opportunity to present some background information that will enable it be placed on the list of approved alternative technologies to be consulted whenever a variance is applied for.

BioHaven floating treatment wetlands (FTWs) offer an *in situ* solution that can directly augment the nutrient removal process in lagoons; or alternatively, can be applied "further up" in the watershed, to mitigate the total load within the watershed, and reduce the downstream affects on beneficial uses. The possibilities of applying this technology in Big Timber, for example, to limit the nutrient load coming downstream to Billings, spreads the economic impact between multiple communities, and provides a template for credits to be traded within and beyond the watershed.

A summary report on FTWs, attached to this email, is the most comprehensive literature review of FTWs to date, published jointly by Swansea University (Wales) and Seacams in March 2014. BioHaven Inc.'s Montana-based technology is clearly identified as the industry leader. There is no question that FTWs are an effective and affordable means to treat point source and nonpoint source waste waters.

The website www.floatingislandinternational.com contains numerous case studies and research papers testifying to the efficacy of BioHaven floating islands. A complete list of publications is attached for your reference.

Thank you again for the opportunity to comment. I look forward to being able to serve Montanans, and the life and waters that depend on our stewardship, through deployment of cost-effective and sustainable technology.

Anne Kania
BioHaven Inc.

PO Box 252
Shepherd, MT 59079

(406) 373-9522 – tel.
(406) 208-5520 - mobile
floatingisland@icloud.com

SEACAMS Swansea University

Enterprise Assist:

Floating Treatment Wetlands (FTWs) in Wastewater Treatment: Treatment efficiency and potential benefits of activated carbon.

I Dodkins & AF Mendzil

March 2014

Prepared for: FROG Environmental Ltd, Ban y Berllan,
Llansadwrn, Llanwrda, SA19 8NA.

Ewrop & Chymru: Buddsoddi yn eich Dyfodol
Cronfa Datblygu Rhanbarthol Ewrop
Europe & Wales: Investing in your Future
European Regional Development Fund

Sustainable Expansion of the Applied Coastal
And Marine Sectors (SEACAMS)
Prifysgol Abertawe/Swansea University
Abertawe/Swansea
Cymru/Wales
SA2 8PP
www.seacams.ac.uk
Ebost/Email: I.R.Dodkins@swansea.ac.uk

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Literature Review:

Floating Treatment Wetlands (FTWs) in Wastewater Treatment: Treatment efficiency and potential benefits of activated carbon.

Dr. Ian Dodkins; Anouska Mendzil; Leela O'Dea

Executive Summary

Floating Treatment Wetlands (FTWs) have many benefits over Free Water Surface (FWS) wetlands:

1. Plant roots assisting in filtering and settling processes for sediment bound P and metals
2. Plant roots acting as a large surface area for micro-organism activity in: decomposition, nitrification, and denitrification (removal of BOD and N).
3. Mild acidification of water due to release of humic acids; and a C input from senescent vegetation, assisting denitrification.
4. They can adjust to varying water levels
5. A higher retention time is possible as they can be made deeper without submerging the vegetation

Percentage removal of nutrients and metals from effluent is around 20-40% higher in FTWs than in conventional FWS ponds. Removal efficiency, particularly of nitrogen, can be further increased with tighter control on the water chemistry (aeration; adding $CaCO_3$; adding a carbon source). 20% coverage of islands is optimal for aerobic basins. 100% cover is optimal for anaerobic basins or aerobic basins where there is artificial aeration. The design the FTW and the control of basin water chemistry is essential for optimising treatment efficiencies. The passive use of activated carbon within layers of floating islands is unlikely to be cost effective.

Introduction

Definition

Floating Treatment Wetlands (FTWs) comprise of wetland basins or cells, on which there are artificial mats containing emergent plants (Figure 1). This is not to be confused with treatment using floating leaved plants such as *Eichhornia crassipes* (Water Hyacinth), *Pistia stratiotes* (water lettuce), *Lemna* spp. (duck weed) or *Azolla* spp. (water fern) e.g. Reddy & Smith (1987); Kivaisi (2001), or where natural floating islands have established. Floating Treatment Wetlands are also referred to as Constructed Floating Wetlands (CFWs) or Floating Mat Constructed Wetlands, but we will use FTW throughout the review. Floating Islands (FIs) will be used to refer only to the islands within the treatment system. 'Effluent' refers to the water being treated at any stage within the wetland and 'inflow' refers to effluent entering the wetland, and 'outflow' as effluent leaving the wetland. Comparison will regularly be made between FTWs and other wetlands. Where 'conventional wetlands' is referred to, this means other treatment wetlands in general. Basins where there is open water but no islands, are known as Free Water Surface (FWS) wetlands.

The core of this review assesses process, performance and design of FTWs and includes a section on the potential for incorporating activated carbon into FTWs.

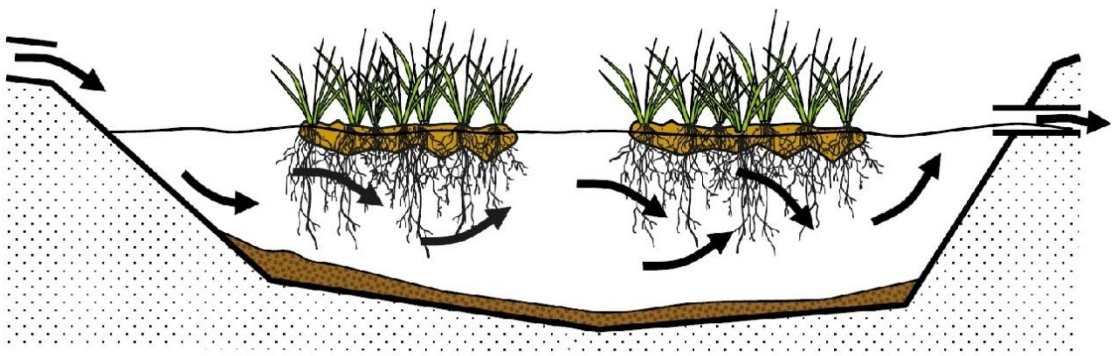

Figure 1. A Floating Treatment Wetland (FTW). Emergent plants are grow within a floating artificially constructed material. The roots are directly in contact with the effluent and can intercept suspended particles. The roots also provide a high surface area for microbiological activity. Image: Headley and Tanner (2006).

1.1 History

Floating islands are a natural occurrence, and can be found where emergent aquatic plants have broken from the land, sometimes developing in highly nutrient rich or sulphurous pools (Duzer, 2004). Floating leaved plants for treatment date back to the 11th Century, when the floating *Azolla* fern was used by Chinese and Vietnamese farmers to extract dissolved nutrients from wetlands and rice paddies, after which it was dried and applied as soil fertiliser (Whitton & Potts, 2002). The use of Water Hyacinth (*Eichhornia crassipes*) to remove nutrients also developed in South East Asia, and both have been used for centuries for water treatment within this region (Whitton & Potts, 2002). *E. crassipes* was suggested for use in the early 20th Century in both Auckland (Australia) and Yorkshire (UK) (Dymond, no date), and then in 1975 NASA used it to treat a sewage lagoon in the USA (Wolverton & Mcdonald, 1978).

Constructed floating islands were first developed in Japan in the 1990s, with *Cana generalis* being grown in floating beds to absorb nutrients from fish ponds and treatment basins (Wu et al., 2000). Twenty percent coverage of soilless artificial floating islands, again using *C. generalis*, was later recommended to improve water quality in China (Bing & Chen, 2001).

Floating mats have also developed unintentionally in many open water treatment systems. Sometimes detrimental effects were observed, such as in Florida, where mats which had grown to 50% coverage moved with the wind across a shallow basin; scraping the bottom and disturbing sediments, resulting in increased outflow turbidity and phosphorus release (Kadlec & Wallace, 2009).

1.2 Range of applications

FTWs can be specifically designed or they can be installed in currently operating open water wetlands i.e. retrofitting. Potentially FTWs can be used with the same waste streams as conventional wetland systems. Some examples from the literature are:

Domestic wastewater treatment

Conventional vegetated wetlands have often been advocated for wastewater polishing, rather than heavy nutrient loads, since they can become clogged and plants are very good at removing low concentrations of nutrients. However, given that there is primary sedimentation, FTWs can potentially deal with larger nutrient loadings since they have higher P and N removal capacity compared to conventional wetland treatment systems. The exposed roots aid sediment deposition, thus reducing turbidity, and there is greater surface area available for microbiologically mediated nitrification/denitrification reactions. Treatment with floating islands has been done on domestic waste in a highly controlled environment i.e. as a hydroponic system (Vaillant et al., 2003).

Metals treatment

Wet detention ponds are used as a Best Management Practice for stormwater run-off in the USA (Chang et al., 2013). FTWs have thus become a popular choice as a retrofit for stormwater run-off treatment in these ponds (Chimney et al., 2006; Headley & Tanner, 2006; Tanner & Headley, 2008, 2011; Hwang & Lepage, 2011; Chang et al., 2012; Borne et al., 2013; White & Cousins, 2013; Winston et al., 2013). FIs are beneficial since they can treat water effectively even with the large fluctuations in water depth that occur during storms.

Strosnider & Nairn (2010) stated that FTWs are ideal for acid mine drainage, particularly if anaerobic conditions are maintained using high island cover. The resulting anaerobic conditions and the decomposing plant material aids denitrification, making the water more alkaline.

Agricultural waste

The enhanced nitrate removal rate of FTWs makes them appealing in reducing pollution from agricultural run-off (Stewart et al., 2008; Yang et al., 2008) as well as for more concentrated wastewaters, such as from swine effluent (Hubbard et al., 2004).

Habitats

FIs have sometimes been constructed specifically to create habitats e.g. to protect birds from land-based predators (Hancock, 2000), including a huge floating island of 3 700 m^2 in Sheepy lake (California) as a habitat for nesting Caspian Terns (Patterson, 2012). Only islands designed for effluent treatment will be covered in this review, however FIs do provide habitats as a secondary function. Emergent grasses can attract waterfowl and terrestrial birds because of the seeds, nesting material, nesting cover and available water. Fish have been introduced into some open water wetlands, however those that feed or nest on the bottom have been found to disturb sediments, increasing suspended solids (Kadlec & Wallace, 2009, p.779).

Although usually not problematic, there have been incidences where large bird communities have contaminated open water treatment wetlands with faeces (Orosz-Coghlan et al., 2006), or have disturbed sediments, increasing turbidity (Knowlton et al., 2002). Geese herbivory can devastate the establishment of wetland plants, especially if planted during the spring or autumn migratory period (Kadlec & Wallace, 2009). However, a benefit of floating islands is that other herbivores e.g. rabbits, cannot usually access the islands. Mosquitoes may also be a problem with an open water system, particularly where monotypic vegetation such as cattail, bulrush and common reed restrict predator access (Knight et al., 2004). However, removing leaf litter, and ensuring that water depth is greater than 40cm (Sinclair et al., 2000) can reduce the problem.

Tourism

Treatment wetlands have been effectively marketed for tourism, especially those which provide good natural habitats for birds (Kadlec & Wallace, 2009). If the FTW is operated for tourism the design and operation is likely to have to include walkways, bird viewing areas and education centres. There may also be conflicting aims for depth regulation between habitat provision and treatment.

2. Processes

FTWs, as with other wetland treatment systems, remove pollutants by four main processes (in order of importance): physical; biogeochemical; microbial and plants. These processes are similar in conventional wetlands, so much of the details provided here comes from that research. However, the larger surface area created by plant roots in FTWs tends to increase sedimentation (by filtering), microbiological decomposition, nitrification and denitrification, and also alter the water chemistry i.e. pH and dissolved oxygen (DO) concentrations. Processes will be discussed relative to the effluent constituents being removed.

2.1 Phosphorous removal

Phosphorous within wetland effluents is usually as dissolved orthophosphate (PO_4^{3-}), or organic phosphorus (Masters, 2012). The scarcity of P in natural environments results in efficient nutrient cycling within ecological systems (Kadlec & Wallace, 2009), thus there are few permanent routes for removal of P within treatment wetlands (Figure 1). The major mechanisms for P removal are accretion in peat/soil and soil adsorption.

Settling and peat accretion

Settling is the main process by which phosphorous bound sediments and BOD are removed from the water column (Kadlec & Wallace, 2009). Settling is a physical process whereby phosphate bound in particles sink to the bottom. Settling is increased in FTWs both by the roots (Masters, 2012) which filter the particles from the water column to later slough off to settle on the bottom, and by reducing currents and circulation caused by surface wind disturbance or water movements (e.g. from pumps) (Headley & Tanner, 2006; Chang et al., 2013). The reduction in movement is essential for preventing resuspension of sediment bound phosphorous into the water column, however, this reduction in currents also contributes to the risk that the basin will become anoxic (Van de Moortel et al., 2010). P retention within different conventional wetlands ranges from 40-60%, around 45 to 75 $g/m^2/yr$ (Vymazal, 2007), most of this being due to settling (and associated processes such as accretion and soil adsorption). P removal from FTWs is usually higher due to the additional filtering properties of the roots, reaching 81% (White & Cousins, 2013).

Soil adsorption
Phosphorus is retained in the soils by binding to the soil surface. Soils with high clay content have high P adsorption capacity, which increases with lower pHs. Organic soils also adsorb P, with the adsorption capacity dependent on mineral components (Rhue & Harris, 1999). Al and Fe fix phosphorus in acidic soils, whilst Ca and Mg fix it in alkaline soils (Kadlec & Knight, 1996). This adsorption process is reversible, with an equilibrium between the bound P and the dissolved P in the soil porewater. The soil minerals and binding sites result in a 'phosphate buffering capacity' which determines where this equilibrium exists (Barrow, 1983). This has important implications for P removal, since reducing inflow P can cause P desorption from the sediments, actually producing a higher P outflow than inflow (Belmont et al., 2009).

Precipitation of P
P adsorption occurs in aerobic waters, but as conditions become anoxic (reducing conditions) metals within the soil change valency, becoming soluble. This causes the release of phosphorus as a co-precipitate (precipitating due to the action of a true precipitate) from the soil (Kadlec & Wallace, 2009). In very low oxygen conditions, where the soils are anaerobic (Eh < -200 mV) sulphate reduction occurs (Figure 4). This creates free sulphide which preferentially binds with Fe (as iron sulphide) preventing iron mineralisation of P. Thus, anaerobic conditions promote the release of P back into the water column (Kadlec & Wallace, 2009).
Plant uptake
Plant uptake of P reaches only around 6% (Masters, 2012). If a FTW has a P removal up to 81% (White & Cousins, 2013), this means around 75% is removed predominantly by settling or storage in other sinks. Much of the P in plant uptake is also difficult to remove permanently from the system by harvesting because it is stored in the roots, or it re-enters the system as litter (see Section 2.8 Harvesting). Vymazal (2007) considers that harvesting of conventional wetlands is only useful in low P effluents (e.g. polishing) with around 10-20 g $P/m^2/yr$, where uptake is not limited by growth rate. FTWs may be able to absorb more P, due to their roots being suspended directly in the effluent, and plant roots are more accessible for harvesting, but dredging is still likely to be the most effective method of permanent removal.

Microbial and Algal uptake
Bacteria and algae are important in P cycling within the soils, rhizosphere and water column (Vymazal, 2007). P uptake by microbes in conventional wetlands is very fast, but they store very little (Vymazal, 2007). Thus, having higher surface area and consequently higher microbial mass, microbes in a FTW are likely to be a larger sink of P than in conventional treatment wetlands, however nutrient cycling is likely to result in little net removal, except through sedimentation of dead organic microbial matter.

Fish Uptake
In South East Asia it is common to use fish for nutrient recovery in ponds receiving human effluent (Cairncross & Feachem, 1993). Fish eat periphyton (such as algae, cyanobacteria, heterotrophic microbes, and detritus) (Azim et al., 2005) as well as fungi, protozoa, phytoplankton, zooplankton, invertebrates and invertebrate larvae, and some species are piscivorous. In treatment wetlands fish are usually chosen for their adaptation to low oxygen levels, for example *Gambusia affinis* (mosquito fish) in warm temperate to tropical conditions, and *Notrophus fundulus* (black-stripped top minnow) or *Umbra limi* (central mudminnow) in temperate climates with over 77 different fish species being used in North American treatment wetlands (Kadlec & Wallace, 2009). Sometimes *Oreochromis* spp.

(Tilapia) and Bass have colonised previously unpopulated treatment wetlands (Kadlec & Wallace, 2009).

Li & Li (2009) examined nutrient removal from aquaculture effluent using floating islands (17% cover) planted with the aquatic vegetable *Ipomoea aquatic.* There was artificial aeration and it was populated with *Aristichthys nobilis* (silver carp), *Siniperca chuatsi* (mandarin fish; carnivorous) and *Carassius auratus gibelio (*crucian carp). Around 34% of TN and 18% of TP was removed from the system, and of this around a third (34%) of removed TP and TN was removed by fish. This was around the same that was removed by sedimentation.

Kania (2014, unpublished) suggests that FTW facilitate the sustainable growth of fish and demonstrates that FTW significantly increase fish biomass that can be harvested from the waterway. Fish harvesting enables P removal from the effluent with fish being made into meal which can be used for pork or poultry farming or in pet food. There must be no toxins or toxic metal contaminants in the effluent, especially contaminants that may bioaccumulate. Also, if it is to be sold for human consumption the fish need to be cooked well since there is the potential for contamination by pathogens, particularly the tapeworm *Clonorchis sinensis* (Cairncross & Feachem, 1993).

Fish can disturb bottom sediments, releasing P, particularly those that feed or nest on the bottom e.g. *Cyprinus carpio* (Carp) (Kadlec & Wallace, 2009, p.696).

Problems with phosphorous removal

Generally, wetland treatment only produces temporary storage of P, in contrast to N and C which can be released as gases through microbiological degradation (N_2 and CO_2). Indeed, Yousefi and Mohseni-Bandpei (2010) stated that P can be considered as a conserved entity. Most P is stored in sinks such as sediments (95%; Masters, 2010), plants, microbes and algae, but this P is recycled. These sinks give an initial period of apparent P removal. However, once the wetland is established, nutrient cycling results in similar outflow P levels to inflow. Even regular harvesting of plants only removes around 6% (Masters, 2012) of P inflow, if both the roots and shoots are harvested. Thus, Kavanagh & Keller (2007) concluded that at least 90% of P eventually passes through a wetland system and is released in the effluent.

Some wetland treatment systems can even export more P than they receive, such as a stormwater wetland in North Carolina which had median removal efficiencies of – 95% to 70%; at times exporting twice as much P as it was receiving (Line et al., 2008). This can occur both due to physical disturbance of the sediments releasing P, the re-release of P from biodegradation of organics (Sundaravadivel & Vigneswaran, 2001), or anoxia which can also result in the sudden release of P as a co-precipitate (Maine et al., 2005).

Sudden P releases into the water column can potentially have other detrimental effects. Since P is usually the limiting factor for biological activity in freshwaters (Schindler et al., 2008), a large P release can result in nitrogen becoming limiting. This promotes the growth of Cyanobacteria blooms which as well as producing harmful toxins, also extract N from the atmosphere (Conley et al., 2009).

Masters (2012) is thus emphatic that dredging is important for long term removal of phosphorus from a FTW. Kadlec and Wallace (2009) detail projected working life of different types of wetlands with different soils, ranging from around 10 to 170 years, but dredging around every 10 years (Masters, 2010) would be ideal for sustained P removal with most effluents.

A minor route of P removal is phosphine (PH_3). It is usually found in very low amounts (e.g. 47 ng/m^3 of water in marshes), mostly bound to sediments but with around 10% of this dissolved in the water (Hana et al., 2010). However, it can be released from highly anaerobic wetlands (Eh < - 200mV) (Gassmann & Glindemann, 1993) as phosphine gas. Devai and Delaune (1995) calculated a gaseous release rate of 1.7 g P/m^2/yr from a bulrush wetland treatment system.

Thus, treatment wetlands have various sinks (algae, plants, microbes, soils) which vary in their capacity to absorb P from the effluent based on conditions such as available surface area, soil type, pH and redox potential. FTWs limit the resuspension of particulates since the islands reduce water movement within the wetland and the roots filter out particulates (Borne et al., 2013), thus increasing P sedimentation. However, dredging is essential to long term functioning of a FTW for P removal, and regular harvesting can be useful at low P loadings (Figure 1).

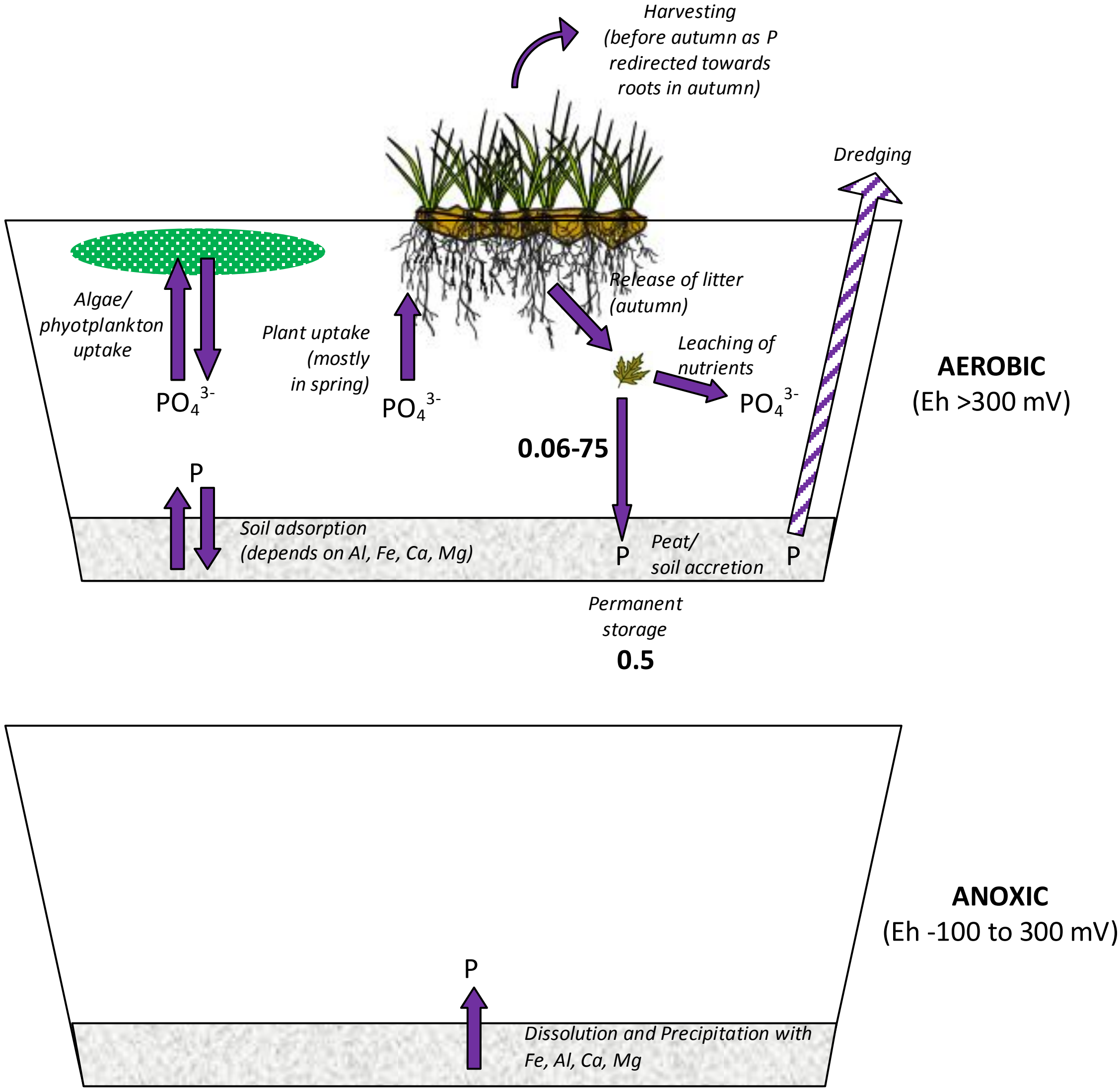

Figure 2. Summary of phosphorus processes in aerobic and anoxic wetlands.
Soil/peat accretion and soil adsorption is the major process and major (95%) sink. However, sorption of P into the soil is reversible. Without harvesting or dredging P removal eventually stops. Numbers in **bold** are g P/m^2/yr that may be removed or added during the processes; *italics* indicate the name of the process, with specific conditions required in brackets.

2.2 Nitrogen removal

Nitrogen is the principal target for treatment in many wetlands. Effluents contain organic nitrogen compounds, which break down principally to ammonia, from which nitrite and nitrate can form through a microbiological nitrification process. Different micro-organisms within anoxic zones can denitrify this nitrate to permanently release N_2 gas from the basin. Agricultural wastes may already have high concentrations of nitrate nitrogen as they enter the wetland. Conversion between different forms of N depends on many factors, including DO, available carbon and pH.

2.2.1 Nitrogen removal in aerobic water

Ammonification (mineralisation)

Dead and decaying organic material is broken down in to ammonia by microbes, either utilising the energy released or absorbing the ammonia for use as microbial biomass. Ammonification increases with temperature, being optimal at 40-60 °C, and with organic compound availability (especially when they have low C/N ratios) (Reddy & Patrick, 1984). Optimum pH is between 6.5 and 8.5 (Vymazal, 2007). Ammonification usually takes place under aerobic conditions (oxidative deamination).

Equation 1: Break down of organic N (example with amino acid) to ammonia

$$RCH(NH_2)COOH + H_20 \rightarrow NH_3 + CO_2$$

Ammonification rates can vary greatly e.g. between 0.004 and 0.53 g $N/m^2/d$ (Reddy & D'Angelo, 1997; Tanner et al., 2002). The root zone of a FTW is likely to be a good location for ammonification.

Ammonia volatilisation

Ammonia exists in an equilibrium between its dissolved ammonium form (NH_4^+) and its gaseous form (NH_3); Equation 2. Below pH 8.0 ammonia loss as gas is negligible (Reddy & Patrick, 1984). At a pH around 9.3 losses due to volatilisation can become significant (Vymazal, 2007). N removal rates due to ammonia volatilisation have been measured at 2.2 g $N/m^2/d$ in wetlands (Stowell et al., 1981).

Equation 2: Conversion of dissolved ammonium to ammonia gas

$$NH_4^+ + OH^- \Leftrightarrow NH_3 + H_2O$$

Algal photosynthesis often elevates pH values during the day (Vymazal, 2007), thus increasing ammonia volatilisation. However, FTWs may inhibit this due to (i) islands shading algae and reducing the area of the air-water interface, and (ii) plants releasing humic acid, which reduces the pH (Van de Moortel et al., 2010).

Nitrification

Within aerobic water micro-organisms convert ammonium to nitrate in a process called nitrification. Directly adjacent to plant roots there is an aerobic zone (Reddy et al., 1989), which means that FTW are likely to have elevated denitrification rates due to the availability of root surface area.

Kadlec & Wallace (2009; p.280) note that nitrification in wetlands is quite different from nitrification in conventional Waste Water Treatment Works (WWTWs). Whilst nitrification is commonly considered a two step process in conventional WWTWs, in natural wetlands it is now believed to have three stages (Bothe et al., 2000); Equation 3.

Equation 3: The three stage nitrification process, converting ammonium to nitrite, then nitrate.

Nitritation (2 stages)

$$NH_3 + O_2 + 2H + 2e^- \xrightarrow{\textit{Nitrosomonas}} NH_2OH + H_20$$

$$NH_2OH + H_20 \xrightarrow{\textit{Nitrosomonas}} NO_2^- + 5H + 4e^-$$

Nitrification (1 stage)

$$2NO_2^- + O_2 \xrightarrow{\textit{Nitrospira or Nitrobacter}} 2NO_3^-$$

Due to the different processes less oxygen and alkalinity is consumed in wetlands during nitrification than in conventional WWTWs (Kadlec & Wallace, 2009). *Nitrospira* is also much more prominent as a nitrifier than *Nitrobacter* in wetlands (Austin et al., 2003).

Nitrification is influenced by temperature (optimum 25-35 °C), pH (optimum 6.6-8), alkalinity, microbial populations present, DO and ammonium concentrations (Vymazal, 1995). Below 4 °C nitrifying bacteria *Nitrosomonas* and *Nitrobacter* do not grow (Paul & Clark, 1996). Kadlec & Wallace (2009; p.280) note, unlike WWTWs, there is little evidence that a low C/N ratio in wetland effluents improves nitrification rates.

In wetlands, for every g of ammonium oxidised to nitrate 2.28 g of oxygen and 7.1 g of alkalinity as calcium carbonate are consumed (Kadlec & Wallace, 2009; p.279) i.e. nitrification requires aerobic conditions and will consume alkalinity and oxygen, becoming increasingly acidic and anaerobic. Wetlands have nitrification rates of 0.01 to 2.15 g $N/m^2/d$ (mean of 0.048) (Reddy & D'Angelo, 1997; Tanner et al., 2002), though this may be much higher for FTWs due to the large root surface area within the aerobic zone.

Low oxygen conditions can result in nitrite (NO_2^-) being produced instead of completing the process toward nitrate (Bernet et al., 2001). The consequence of this is that in a later denitrification stage, some of the nitrite is converted into nitrous oxide (N_2O), a potent greenhouse gas. Sufficient oxygenation in nitrification basins is therefore recommended.

2.2.1 Nitrogen removal in anoxic water

Denitrification

Denitrification is the microbiologically mediated conversion of nitrate into nitrogen gas, which is then released from the wetland into the atmosphere. A carbon source is required for denitrification. The equation can be written in many ways, depending on the source assumed (Equation 4).

Equation 4: Denitrification of methanol, producing nitrogen gas and alkalinity

$$6NO_3^- + 5CH_3OH \longrightarrow 3N_2 + 5CO_2 + 7H_20 + 6OH^-$$

In many ways denitrification is the converse of nitrification, making the water more alkaline and requiring anoxic or anaerobic conditions. Microorganisms denitrify because in the absence of dissolved oxygen for reduction, they reduce nitrate. Although methanol is used for illustration here as a source of carbon, usually it is large organic molecules. It is calculated that per g of NO_3^- around 3.02 g of organic matter (or 2.3g of BOD) is consumed, and around 3g of alkalinity as $CaCO_3$ is produced (Kadlec & Wallace, 2009).

The optimum pH is 6 to 8 (Paul & Clark, 1996) being negligible below pH4 (Vymazal, 2007). Denitrification is very slow below 5 °C, but increases with temperature up to 60 or 75 °C, then decrease rapidly (Paul & Clark, 1996). More nitrate can speed up the process, but the limiting factor in denitrification is often the carbon supply (Kadlec & Wallace, 2009), especially if BOD has settled out in previous treatment basins. A C/N ratio of 5:1 is suggested to ensure carbon does not become limiting (Baker, 1998) although this may be an overestimate if much of the C is labile (Kadlec & Wallace, 2009). Lower pHs can assist with breaking down lignin in cell walls, increasing the litter quality for denitrification processes (Ding et al., 2012).

Often an anaerobic denitrification basin is placed after an aerobic nitrification basin. This enables all the ammonium to be converted to nitrate prior to denitrification, thus maximising total N removal. However, even in a well oxygenated basin there are areas of low mixing, and deeper waters and sediments, where oxygen levels are low enough to produce denitrification (Figure 3, Figure 4), and in anoxic basins nitrification can occur on the surface of roots where the plants have transported oxygen (Kadlec & Wallace, 2009, p.281). Thus both nitrification and denitrification processes can be achieved within a single basin, though controlling the treatment efficiency may be more difficult.

Floating islands can aid denitrification by producing anoxic conditions through the restriction of oxygen diffusion into the water column. Also, roots and plant litter, as well as coconut coir on islands (Baquerizo et al., 2002),can act as sorption sites, with biofilms developing which increase denitrification rates and thus NO_3 removal rates (Vymazal, 2007). Denitrification releases are about 0.003 to 1.02 g N/m^2/d in wetlands (Vymazal, 2007), though this could be higher in FTWs due to more biofilm area and more sorption sites.

Anaerobic Ammonia Oxidation (ANAMMOX)

The bacteria involved in this process were only discovered in 1999. *Planctomycetes Nitrosomonas eutropha* utilises ammonium ions and nitrite (from nitrification of ammonium) to produce nitrogen gas. This can be represented as in Equation 5.

Equation 5. The ANAMMOX process

Formation of nitrite

$$2NH_4^+ + 3O_2 \text{-------------}\rightarrow 2NO_2^- + 4H+ + 2H_2O$$

ANAMMOX

$$NH_4^+ + NO_2^- \text{-------------}\rightarrow N_2 + 2H_2O$$

This denitrification process uses less than half the oxygen (1.94g O per gram of NH_4^+) of the standard denitrification process, and requires no carbon substrate (Kadlec & Wallace, 2009). ANAMMOX processes occur in many types of wetlands when there is severely restricted oxygen. Bishay & Kadlec (2005) found that in a Free Water Surface wetlands there were more ammonia losses than could be accounted for by the oxygen consumed under normal dentification. There was also a lot of nitrite present in this wetland, and very little carbon, suggesting that these conditions were conducive to the ANAMMOX reaction.

Plant uptake

Nitrogen uptake by plants in conventional wetland treatment is low (up to 6-8%) compared to microbial denitrification (up to 61-63%) (Metheson et al., 2002). Vymazal (2007) estimates that for conventional wetland systems plant harvesting is useful for N removal if loading is only around 100-200 g N/m^2/yr. If N removal is a priority, designing and operating the basins to maximise nitrification/denitrification by microorganisms is probably more cost effective.

N is predominantly taken up by plants in the form of ammonia, but also as nitrate. Much of this is returned to the system when tissues senesce (Kadlec & Wallace, 2009).

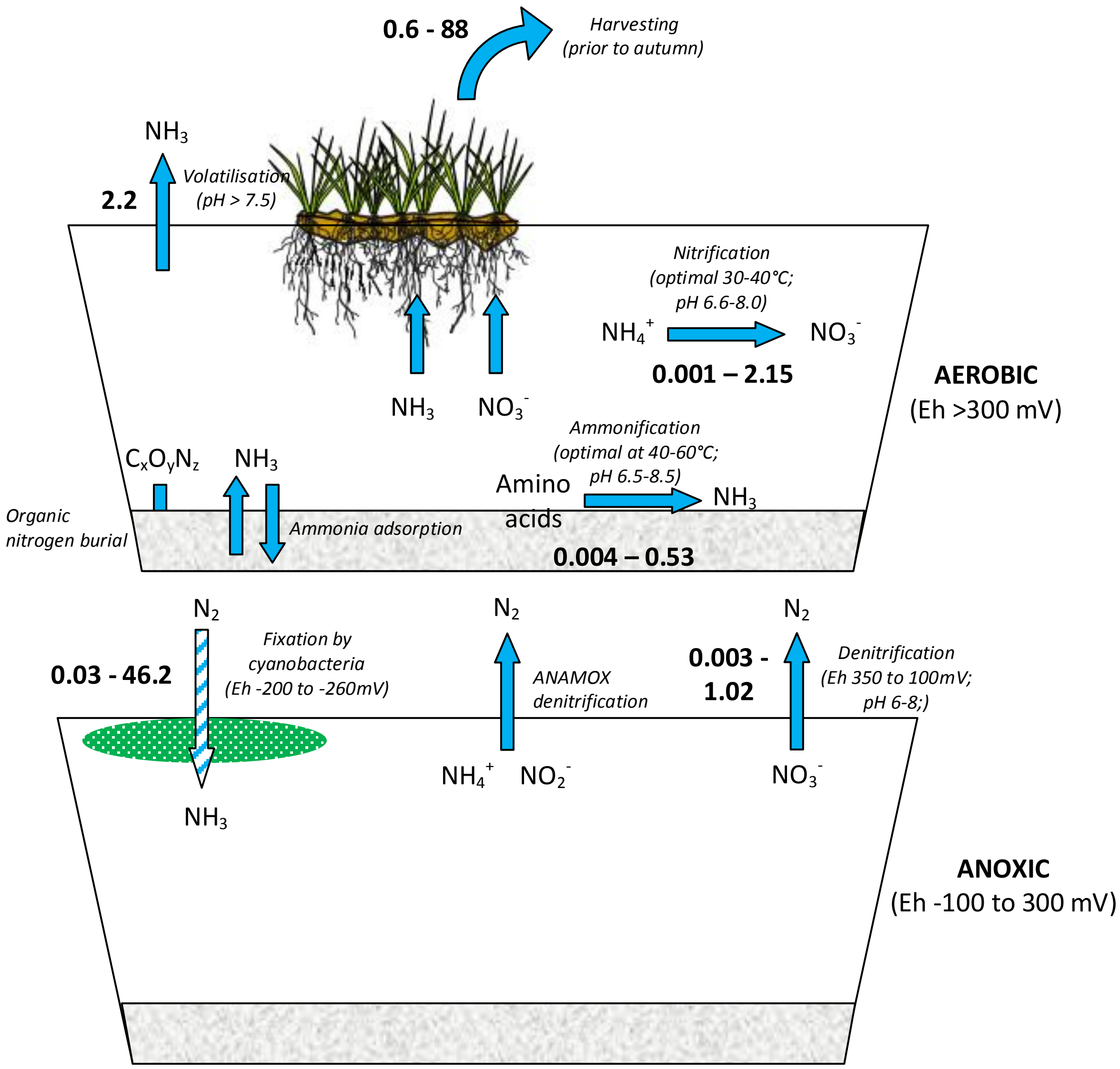

Figure 2. Summary of nitrogen processes in aerobic and anoxic wetlands. Primary settlement of effluent is assumed prior to entering the wetland. Numbers in **bold** are g N/m^2/yr that may be removed or added during the process; *italics* indicates the process, with specific conditions required in brackets. The striped blue arrow indicates nitrogen fixation that would not normally occur unless the anoxic pond becomes anaerobic (Eh < -200 mV).Permanent removal of N is only through ammonia volatilisation (minor), denitrification (including ANAMOX) and harvesting. Organic nitrogen burial (associated with litter) and ammonia adsorption (associated with clay soils) are relatively minor processes.

Problems with nitrogen removal
NH_4 removal rates in conventional wetlands vary between 35 and 50% in Europe (Verhoeven & Meuleman, 1999; Vymazal, 2002). FTWs have shown removal rates from -45% to 75% for NH_4 and between 36% and 40% for total nitrogen (Boutwell, 2002; DeBusk & Hunt, 2005; Gonzalez et al., 2005).

Problems with nitrogen removal are associated with producing the correct microbiological conditions; aerobic for nitrification and anoxic for denitrification, as well as ensuring sufficient carbon supply for the later. These are discussed in the design section.

2.3 Oxygen

Factors influencing oxygen concentrations
Wetlands typically have slow flow, incomplete mixing, and rapidly decreasing oxygen profiles with depth (Figure 3). Anoxic zones develop just below the substrate in shallower basins and also in the lower regions of the water column in deeper basins (Kadlec & Wallace, 2009). Oxygen can be rapidly depleted in wetlands due to microbiological activity, particularly with nitrification and decomposition (Kadlec & Knight, 1996). FTWs exacerbate oxygen depletion both due to high rates of microbiological activity (nitrification) and due to the islands restricting diffusion of oxygen back in to the water i.e. reducing air-water contact area and reducing wind disturbance (Van de Moortel et al., 2010). This makes FTWs particularly susceptible to unwanted drops in DO, especially at high percentage cover of islands.

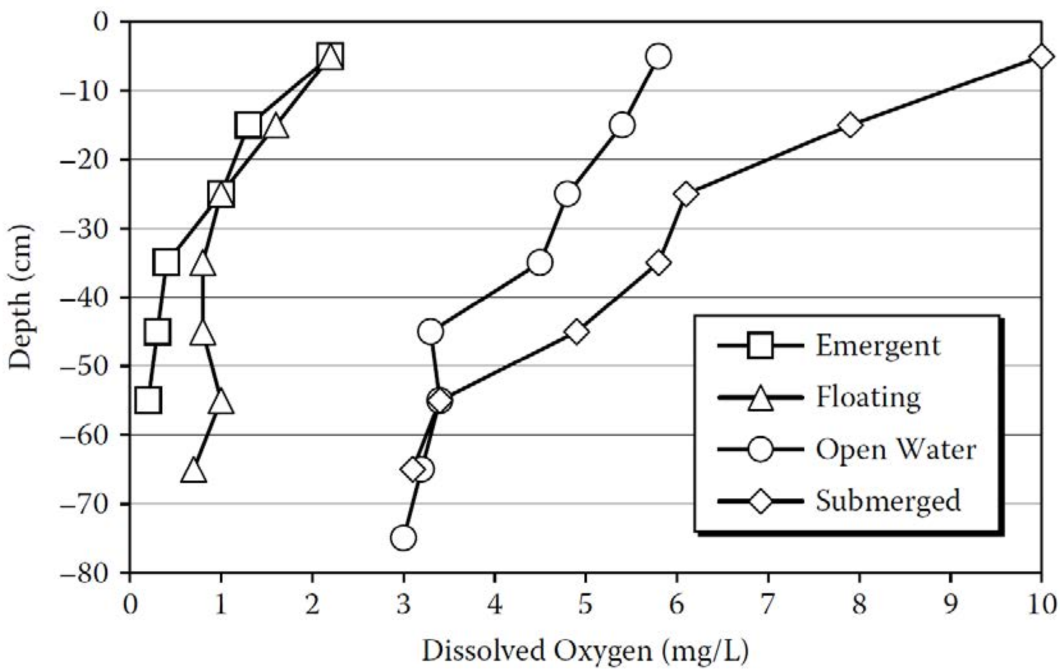

Figure 3. Vertical profiles of dissolved oxygen in various types of FWS (Free Water Surface) wetlands, Florida. Data from 141 profiles collected over a 2½ year period. Data from Chimney et al. (2006), Figure from Kadlec & Wallace (2009). FTWs are most readily compared to floating plant systems.

Temperature affects
Oxygen saturation of water varies with temperature: at 25 °C dissolved oxygen is 8.2 mg/l, and at 5 °C it is 12.8 mg/l. However Kadlec and Wallace (2009) note that the poor mixing of waters limits the dissolution of oxygen such that reaeration is very slow, even in open

wetlands. They estimate that it takes 2 to 4 days to reaerate an open wetland basin from 0 to 90% DO, with typical winds. This is likely to be even slower in FTWs.

Plants

Submerged photosynthesising plants and algae release in the range of 0.26 and 0.96 g/m^2/d of O_2 during photosynthesis (Kadlec & Wallace, 2009, p138), oxygenating the water. Emergent plants bring O_2 to the roots, but O_2 delivery usually matches respiration requirements, so there is little net input into the water column (Brix & Schierup, 1990). Studies by Tanner and Headley (2011) and White and Cousins (2013) both found a high level of oxygen depletion in basins due to floating islands.

Tanner and Headley (2011) not only illustrated how oxygen depletion is higher in FTWs, but also that oxygen depletion is higher when there are plants rather than mats with artificial roots (sisal) (Table 1). This oxygen depletion is likely due to the higher rate of microbiological activity associated with plant roots. Although the relationship between oxygen depletion and root biomass was weak, there was little oxygen depletion due to the floating mat alone and even the mat with artificial roots.

Table 1. Oxygen depletion (%DO) at subsurface and bottom of mesocosms after 7 days due to the effect of Floating Islands, ordered from highest to lowest. Influent DO was 95%, floating island coverage was 50%. Root biomass (dry weight) also shown. Adapted from Tanner and Headley (2011).

	subsurface DO (%)	bottom DO (%)	Root biomass (g/m^2)
Control (no floating mat, but equivalent shading)	87	85	
Floating mat only	85	84	
Mat + soil + artificial roots	85	84	
Mat + soil media	80	79	
Mat + soil + Juncus edgariae	68	66	299
Mat + soil + Schoenoplectus tabernaemontani	68	67	184
Mat + soil + Carex virgata	58	57	533
Mat + soil + Cyperus ustulatus	50	48	329

Van de Moortel et al. (2010) found redox potentials to be decreased due to floating islands: at both 5cm and 60cm depths the FTW has much lower O_2 than an open water basin: at 5cm redox is 68 (open water) cf. -25 (FTW); at 60cm redox is: -93 (open water) cf. -122 (FTW). They did claim that roots can aerate island matting. However, there was little difference between the mat redox potential (72 mV ± 478) and the redox potential 5cm below the surface of an open water basin at (68 mV ±225).

A liability with FTWs is that during summer periods, due to high rates of microbiological activity and insufficient O_2 exchange with the atmosphere, the basin can become anaerobic, causing sulphide toxicity which then kills the plant roots (Lamers et al., 2002) and consequently reducing the effectiveness of treatment. Reduction in treatment efficiency due to anoxia was found in several studies, but usually when the floating islands occupied 50% or more of the surface water area (Van de Moortel et al., 2010; Borne et al., 2013).

2.4 Redox potential

Oxidation is the loss of electrons during a reaction. This is usually through a substance combining with oxygen, as it is energetically the most favourable oxidant. Redox potential is the tendency of a system to oxidise substances i.e. in high redox potential water, incoming organic substances will be rapidly oxidised (an oxidising environment) whereas in low redox potential waters substances will be reduced (a reducing environment). An example of reduction would be where hydrogen combines with carbon to produce methane.

Redox potential is strongly associated with the oxygenation of the water, but it is not identical, since substances other than O_2 can oxidise. Zonation usually occurs in a wetland with oxygen being the oxidiser near the surface, then as DO decreases other substances become oxidisers, with reactions releasing less energy with successively weaker oxidisers. This is in the order O_2, NO_3^-, MnO_2, FeOOH, SO_4^{2-} then CO_2.

The decline in free oxygen reflects the redox potential (Eh), also known as the oxidation-reduction potential (ORP), of the water i.e. the tendency of a chemical to acquire electrons, measured as electric potential (mV). At Eh > 300mV (measured with a platinum electrode) conditions are considered aerobic, at < -100 mV conditions are anaerobic, and between these (near-zero Dissolved Oxygen) conditions are anoxic (Figure 4).

Redox Potential	Reactions		Zone
> +300 mV	Oxygen reduction	I	Aerobic

+ 100 to +300 mV	NO_3^- and Mn_4^+ reduction	II	Anoxic
+100 to – 100 mV	Fe_3^+ and Mn_3^+ reduction	III	Anoxic
-100 to -200 mV	SO_4^{2-} reduction	IV	Anaerobic
< -200 mV	CH_4 formation	V	Anaerobic

Figure 4. Redox zonation in wetlands, based on Kadlec & Wallace (2009). This vertical zonation can be found in deep lentic environments, particularly where there is high oxygen consumption e.g. my microorganisms.

At high redox potential phosphorus can form insoluble complexes with oxidised iron, calcium and aluminium. Organic compounds which comprise most of the BOD are oxidised using oxygen by bacteria, releasing carbon dioxide. At lower redox potentials organic material does not decay quickly. The water is anoxic, with reducing conditions predominating. Manganese and iron are both reduced (Equation 6)

Equation 6. Reduction of manganese and iron in anaerobic conditions

$$Mn^{4+} + 2e^{-} \rightarrow Mn^{2+} \qquad Fe^{3+} + e^{-} \rightarrow Fe^{2+}$$

This reduction causes metals to precipitate out of the sediments back into the water column, bringing P with them, as a co-precipitate (Van de Moortel et al., 2010).

Further decreases in oxygen (below -100mV) result in anaerobic conditions, whereby sulphate is reduced to hydrogen sulphide, which although soluble, can be released as gas at low pH (Kadlec & Wallace, 2009). Usually this reduction is undesirable in wetlands, except in acid mine treatment.

Equation 7. Reduction of sulphate to hydrogen sulphide

$$SO_2^{-4} + 2CH_2O \rightarrow H_2S + SHCO_3^{-}$$

Eventually, at very low redox potential (below -200mV) CO_2 , formate, or acetate, is reduced to methane (CH_4) by bacteria.

Equation 8. Reduction of carbon dioxide to methane.

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_20$$

2.5 BOD, Suspended Solids and Carbon

Biological Oxygen Demand

Biological Oxygen Demand (BOD) is a measure of oxygen consumption by microorganisms due to the oxidation of organic matter; usually measured in the lab over 5 days (BOD_5). BOD of inflows are typically high, unless the treatment basin is being used just for polishing previously treated wastes. BOD decreases rapidly (around 50% decrease within 6 hours) as it passes through a wetland due to decomposition and settling of organic carbon, finally reaching a non-zero plateau (Kadlec & Wallace, 2009). Even if the waters are not aerobic, fermentation and sulphate reduction can remove carbon from the system.

Carbon

Most carbon entering a wetland is organic. Microbiological processes are the main method for removing carbon, through the oxidation of organic compounds, releasing energy. In aerobic waters, respiration takes place (Equation 9), releasing CO_2 to the atmosphere. In anaerobic zones there are four main processes which can take place: (i) fermentation producing either lactic acid or ethanol (ii) methanogenesis producing gaseous methane (iii) sulphate (SO_4^{2-}) reduction producing carbon dioxide and hydrogen sulphide, and (iv) denitrification, producing carbon dioxide and gaseous nitrogen.

Settling is also an important removal method (although the carbon is retained in the sediments). In FTWs plants have been shown to remove around 5.9 g BOD/m^2/day. The large surface area provided by roots can produce a higher rate of microbial decomposition (Brisson & Chazarenc, 2009), but roots also physically entrap particulates onto the biofilm which then fall in clumps and settle out, providing a significant removal pathway for suspended solids (Smith & Kalin, 2000; Headley & Tanner, 2006; Van de Moortel et al., 2010; Borne et al., 2013). Settling is further encouraged by flow resistance through the roots and flow reduction caused by wind shielding of the surface. Particulate carbon, and carbon bound in litter, if it is not decomposed, accumulates in the sediments, particularly where conditions are anaerobic (Kadlec & Wallace, 2009).

Equation 9. Microbiological decomposition of organic compounds.

Respiration

$$C_6H_{12}O_6 \rightarrow CO_2 + H_2O$$

Fermentation

$$C_6H_{12}O_6 \rightarrow 2CHCHOHCOOH \quad \text{(lactic acid)}$$

$$C_6H_{12}O_6 \rightarrow 2CH_3CH_2OH + CO_2 \quad \text{(ethanol)}$$

Methanogenesis

$$\text{(acetate)} \quad CH_3COO^- + 4H_2 \rightarrow 2CH_4 + H_2O + OH^-$$

Sulphate reduction

$$\text{(lactate)} \quad 2CH_3CHOHCOO^- + SO_4^{2-} + H^+ \rightarrow 2CO_2 + 2H_2O + HS^- + 2CH_3COO^- \quad \text{(acetate)}$$

Denitrification (see Equation 4)

Unlike submerged plants, which obtain carbon from the water, carbon uptake by emergent plants is from atmospheric CO_2. Plants thus bring carbon into the system through photosynthesis and the deposition of organic matter. However, the net effect of plants in wetlands is to reduce BOD due to plant respiration, increased settling, and increased decomposition processes (Masters, 2012). Also, where there is carbon limitation in anoxic or anaerobic basins, the C provided by the deposition of litter can be important in increasing denitrification rates (see Section 2.2.1).

Settling of BOD is also affected by basin depth, residence time and water movement (Kadlec & Wallace, 2009). Theoretically higher temperatures should increase microbial decomposition rates. Bacteria have limited activity below 5°C, but in conventional wetlands there is no significant temperature dependence above this (Akratos & Tsihrintzis, 2007; Kadlec & Wallace, 2009). This may be due to limitations in oxygen transfer rates or restricting factors in one or more of the many C processes (Kadlec & Wallace, 2009).

In anoxic (reducing) conditions, the presence of sulphate contributes to the removal of organic matter (BOD/COD) by acting as a coagulant and thus increasing settling rates (Huang 2005).

2.6 Metal removal

Metal removal from wetlands is predominantly through forming complexes with organic matter, and through being coated in iron or manganese oxyhydroxides (Kadlec & Wallace, 2009). This either occurs in the sediments, or they settle out into the sediments. Under anoxic conditions Cu, Zn, Pb, Ni and Ca form insoluble metal sulphides which will settle out. Even in aerobic basins, decomposition of organic matter usually means there is an anoxic layer just below the surface oxic layer (≈1cm) in which these metal sulphides can form.

Predicting metal removal from wetlands can be very difficult, depending on the structure of the sediments and many factors of the water chemistry, with models regularly being wrong by orders of magnitude (Kadlec & Wallace, 2009). Factors that affect metal removal include the Cation Exchange Capacity (CEC) of the sediments, pH (circumneutral usually being optimum), redox potential, the availability of sulphur for the formation of metal sulphides, and the formation of iron and manganese oxyhydroxides (which allow co-precipitation) (Kadlec & Wallace, 2009). Organic soils with humic acids and phenolics increase the CEC and thus adsorption of metals. Sedimentation of metals can result in long term storage, depending on the availability of organics with which metals can complex, although metal accumulation can eventually saturate the soil sink and result in biological toxicity (Kadlec & Wallace, 2009). Thus (careful) dredging is required in the long term to permanently remove metals and ensure the wetland continues to operate effectively.

Uptake by plants is much less important than that by sedimentation, and where metals are taken up, they are mostly stored in the roots. Table 2 shows percentage removal of metals by plants in a conventional wetland and how this is allocated in the roots and shoots.

Table 2. Percentage removal of metals by plants in a conventional treatment wetland and how this is allocated to the roots and shoots (adapted from Nolte and Associates, 1998).

Metal	Roots (%)	Shoots (%)	Total (%)
Ag	2.0	0.0	2.0
As	10.1	0.6	10.7
Cd	13.3	0.0	13.3
Cr	16.8	2.2	19.0
Cu	5.5	0.6	6.1
Hg	6.7	0.0	6.7
Ni	4.7	0.3	5.0
Pb	11.8	2.0	13.8
Zn	6.1	0.4	6.5

Despite plant uptake being low, FTWs have been shown to greatly increase metal removal compared to unvegetated retention ponds. For example Borne et al. (2013) compared treatment in a normal stormwater retention pond with one retrofitted with a floating island. With concentrations of 0.0092 mg Cu/l and 0.035 mg Zn/l in the inflow, particulate Cu and Zn removal was 19% and 40% (respectively) in the normal pond, and 50% and 65% with a floating island. Tanner & Headley (2011) found the removal of dissolved Cu and Zn to be 5% and 1% without a floating island, and 50% and 47% with an island. These authors believe that the benefit of the floating island wasn't principally due to plant uptake. Indeed, Tanner & Headley (2011) found mean plant uptake rates were 0.059-0.114 mg $Cu/m^2/d$ and 1.2-3.3 mg $Zn/m^2/d$, accounting for less than 4% of Cu removal and less than 10% of Zn removal. This was a mesocosm experiment without bottom sediments and with predominantly dissolved metals, so values of plant uptake were probably higher than they would be in a normal FTW.

Tanner & Headley (2011) and Borne et al. (2013) considered that the improved performance with floating islands was due mainly to: (i) interception by the plant roots, (ii) humic acid release from the plants, which reduced alkaline waters to circumneutral pH (Van de Moortel et al., 2010), improving metal complexation and therefore flocculation and settling (Mucha et al., 2008) and (iii) The islands reducing the redox potential to the extent that insoluble metal sulphides formed.

The exact mechanisms of metal removal depend on the specific metal. Most zinc within effluents is in particulate form and is removed predominantly through settling, sorption to organic sediments and chemical precipitation/co-precipitation (Kadlec & Wallace, 2009). It can form precipitates with sulphur (ZnS) and carbonate from the water ($ZnCO_3$) and it co-precipitates with Fe, Mn, Al oxyhydroxides. However, ZnS does not readily precipitate in neutral waters (Younger, 2000), only in more alkaline waters (>7.5). Also, for co-precipiration, the other metals must be present in the effluent, and even then, Fe and Mn oxides are not stable in anoxic waters (Knox et al., 2004). Warmer water temperatures are also correlated with Zn removal, probably due to increased sorption rates (Borne et al., 2013). Aerobic wetlands are expected to absorb about 0.04g $Zn/m^2/d$ (PIRAMID consortium, 2003). Similar to Zn, Cu removal rates increase with temperature, however adsorption is better at more neutral pH (Borne et al., 2013). They also concluded that reduced oxygen resulted in a high production of Cu sulphide precipitates in basins with floating islands.

High loadings of effluent and insufficient adsorption capacity or saturation of the potential sinks (organic carbon, metal hydroxides, high CEC soils) can result in decreasing treatment capacity as well as increasing toxicity. Toxicity can be a biological problem, particularly in open water treatment systems where birds, amphibians and freshwater invertebrates have direct access to the basin (as opposed to subsurface flow systems) (Kadlec & Wallace, 2009). Sorption capacity in studies listed by Kadlec and Wallace estimate between 20 and 780 years operation of a wetland with metal loading. Careful dredging (avoiding resuspension) can be applied to remove contaminated sludges/soils. In mixed wastewater effluents from WWTWs it is likely that the necessity for P removal through regular dredging is higher than that from metal accumulation.

2.7 pH

pH has a profound effect on the functioning of wetlands, as mentioned in previous sections. Several studies have confirmed the effect of floating vegetated islands in reducing pH. In a two year study by White and Cousins (2013) pH decreased from 8.6 to 6.2. After only 11 days Van de Moortel et al. (2010) found a significant pH decrease from 7.5 to 7.0 whilst the control (without an island) stayed constant at around 7.5. Borne et al. (2013) found a difference between the control (8.3) and the FTW (7.3), which aided Cu adsorption. Interestingly Tanner and Headley (2011) didn't notice a drop in pH in mesocosm tanks, although they still found that treatment was enhanced with floating islands, attributing the difference in to the release of bioactive compounds. The researchers who found differences in pH generally agreed that humic compounds were released by the plants, reducing pH. White and Cousins also acknowledged that alkalinity consumed during microbial nitrification on the plant roots could also be a driving force behind dropping pH within aerobic basins.

2.8 Harvesting of Floating Island Plants

FTWs are a relatively new technology with few long term studies, and few details on plant harvesting. The prime functions of plants in FTWs is (i) for their roots to intercept and filter particulates, aiding sedimentation, (ii) to increase the rates of microbiological processes by providing a high surface area on which microorganisms respire, nitrify or denitrify, and (iii) to alter the physic-chemical and chemical environment i.e. increase microbiological processing through the release of humic acids and through reducing DO exchange (acidity and lower oxygen increasing denitrification) and carbon deposition (increasing denitrification). Harvesting is therefore not essential to long term management of FTWs, and although it can help with permanent removal of nutrients and metals, removal rates are typically low. For example, in subsurface flow wetlands plants only removed 2-8% of total nitrogen (Tanner, 2001; Yousefi & Mohseni-Bandpei, 2010) and 3-12% of total phosphorous (Yousefi &

Mohseni-Bandpei, 2010), with microbes believed to be removing the rest of the N, and settling removing the rest of the P. Even with total uptake for N and P estimated at around 6%, all of this is unlikely to be harvested as it is stored in both the roots and shoots, and nutrients are returned back to the wetland through deposition of senescent material.

Practicalities of harvesting

Floating islands facilitate easy harvesting. Often larger islands are able to support the weight of humans, and so cutting could be done directly on the island. Smaller islands can be pulled towards the shore and even lifted out. In contrast with other wetlands where the vegetation is rooted in the sediments, in FTWs both roots and shoots can be removed, and with little disturbance to the sediments. Theoretically a replacement island could be installed immediately, although this may not be cost effective. Also, removal of root mass is likely to be more detrimental to treatment than the gains from permanent removal of the nutrients. For example, FTWs typically increase N and P removal rates by around 20-40% (Table 14), whereas P and N removal by harvesting the whole plant is at the most 6%.

Storage of nutrients in plants

The start of the growing season, in early spring and prior to maximum growth rate, is the time of highest P uptake. However, prior to autumn senescence, much of the P is relocated to the root stock for the following year (Vymazal, 2007). Thus, if removal of P is a priority, harvest timing and frequency is extremely important, with a recommendation that it is done not only prior to senescence, but also during the peak growth period. The P lost in the senescent material re-enters the basin system very rapidly; up to 30% lost through leaching within the first few days of decomposition (Vymazal, 2007).

Although shoot biomass tends to be larger than root biomass (see plants in design section), there is generally more N, P and K stored in the roots than in the shoots, especially when autumn approaches (White & Cousins, 2013; Winston et al., 2013) e.g. Figure 4.

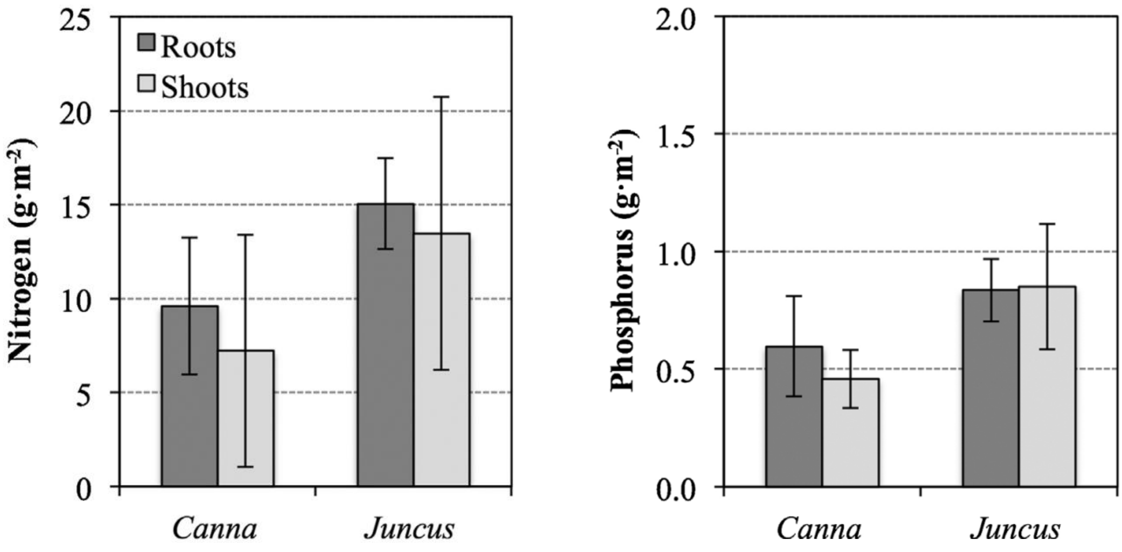

Figure 4. Nitrogen and phosphorus in roots and shoots of *Canna flaccida* and *Juncus effusus* after one summer of growth (harvested 18 September 2008). Nutrients are per m^2 of floating island. Three replicates per bar, with standard error indicated. From White & Cousins (2013).

Storage of metals in plants

Storage of metals tends to show either an even distribution between roots and shoots (e.g. Cu) or predominant storage in the roots (e.g. Zn) (Tanner & Headley, 2011). Table 3 shows uptake of copper and zinc in roots and shoots over a 7 day trial.

Table 3. Uptake of copper and zinc in roots and shoots of four different plant species over 7 days in a FTW, measured as µg/m^2/d. Adapted from: Tanner and Headley (2011).

	Cu		Zn	
Plant species	**roots**	**shoots**	**roots**	**shoots**
Cyprus ustulatus	54	61	3027	282
Carex virgata	54	89	1228	934
Juncus edgariae	38	41	1703	760
Schoenoplectus tabernaemontani	36	24	881	320

Relative importance of different processes

Restricting flow and intercepting particulates on roots is one of the prime benefits of FTWs, consistently removing more BOD and P than open water treatment ponds. However, using synthetic root structure (sisal) Borne et al. (2013) showed that the physical structure alone does not account for most of the benefits of FTWs; water chemistry changes, and to a much lesser extent plant uptake, assist with improving treatment.

3. Treatment Efficiency

Treatment efficiency obtained within a FTW is highly dependent on appropriate design and proper operation, as well as the characteristics of the inflow and the objectives of the treatment. At one end of the scale are FTWs designed for aerobic treatment with mixing or air bubbled into the system, often with low % island coverage and addition of calcium carbonate to aid nitrification reactions. These basins are predominantly to remove ammonium. At the other end of the scale are anaerobic basins with up to 100% island coverage, with addition of carbon in the form of e.g. molasses, to supply the denitrification process. Thus, in aerobic basins, ammonium removal may be high whereas nitrate is produced and may exceed inflow nitrate concentrations. In the latter, denitrification reactions remove nitrate, but ammonium may not be nitrified, resulting in NH_4^+ increasing (due to organic carbon decomposition) such that outflow exceeds inflow. Sometimes floating islands achieve very high rates of removal because of a tightly controlled DO, pH and carbon supply in a hydroponic system. The concentrations of pollutants also affects the removal rate, with higher inflow concentrations often resulting in higher removal rates.

There can be different flow regimes, such as plug flow, where a quantity of effluent is kept in the basin for around 3-7 days, continuous flow, or sporadic flow (such as storm events). Some mesocosm and lab based studies use synthetic effluent with dissolved nutrients, which may exaggerate treatment efficiencies, especially for P and metals which are usually bound to particulates.

Thus, the main considerations when examining performance of a FTW are:

1. Dissolved oxygen: aerobic/anoxic/anaerobic. Natural aeration or artificial aeration through bubblers. With aerobic basins tending to towards nitrification and anaerobic basins tending towards denitrification.
2. Carbon sources: either naturally, through organic carbon, or added artificially, to enhance denitrification rates. Decomposition of organic carbon also results in increased ammonia production within the basin.
3. pH: with alkaline pH increasing nitrification and acidic pH increasing denitrification.
4. Root mass: aiding removal of particulates due to physical filtering and settling processes
5. Mixing: circulation of water to aid the nutrient supply to microbiological processes.
6. Plug flow or continuous flow: affecting residence times and nutrient gradients.
7. Concentrations of inflow pollutants: with higher nutrient supply increasing rates of decomposition/nitrification/denitrification unless limited by another factor.
8. Changes in the FTW chemistry with time. Often pH and redox potential drops due to microbiological processes and restriction of oxygen diffusion from the surface.

Thus, direct comparison between different FTWs has little meaning, and the best comparison is with a relevant control basin. This is often a basin without an island which is receiving the same effluent, however sometimes it is before and after the retrofitting of an island, which doesn't guarantee exactly the same effluent inputs.

New treatment systems can take over a year to stabilise, and even then they can have high variation in treatment efficiency, especially if environmental conditions vary or sinks (such as sediment adsorption) become saturated. However, significantly higher performance of FTWs can be noticed in as little as two days (Van de Moortel et al., 2010), particularly in relation to nitrification/denitrification and other processes which are predominantly dependent on microorganisms, due to their fast response time (Kadlec & Wallace, 2009).

In this section, treatment efficiency from peer-reviewed FTW studies will be summarised separately, with relevant details supplied, and compared to a control where possible. Where complete columns are blank, there was no information.

Abbreviations follow this system: NO_x represents nitrate in the form or either NO_2 or NO_3; TN is Total Nitrogen; N_{org} is organic nitrogen; Cu_{tot} is total copper; Cu_{part} is particulate copper; Cu_{diss} is dissolved copper; DRP is dissolved reactive phosphorus; BOD is biological oxygen demand; COD is chemical oxygen demand; PBP is Particle Bound Phosphorus; TKN is Total Kjeldahl Nitrogen.

Table 4. Removal rates in the study by Van de Moortel *et al.* (2010). Close to 100% coverage of the island resulted in reduced redox potential and anoxic conditions. This resulted in high NO_3 removal rates, but poor NH_4 (thus TN) and P removal rates.

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	2.16	1.77	**18**		1.9	**12**
TN	mg/l	21.8	13.1	**40**	*identical*	19.5	**11**
NH_4	mg/l	16.1	10.8	**33**	*to*	16.5	**-2**
NO_3	mg/l	0.37	0.2	**46**	*control*	0.08	**78**
N_{org}	mg/l	4.31	1.6	**63**	*inflow*	2.87	**33**
TOC	mg/l	27.7	16.4	**41**		23	**17**
COD	mg/l	81.3	46.6	**43**		51.4	**37**
Cu	mg/l	10.0	5.5	**45**		8.4	**16**
Fe	mg/l	454	325	**28**		259	**43**
Mn	mg/l	164	153	**7**		176	**-7**
Ni	mg/l	10.0	6.1	**39**		5.75	**43**
Pb	mg/l	6.10	3.4	**44**		4.58	**25**
Zn	mg/l	57.5	29.7	**48**		47.6	**17**
SO_4^{2-}	mg/l	64.2	49.8	**22**		53.7	**16**
pH	mg/l	7.35	7.08	**4**		7.48	**-2**
cond	µS/cm	1035	1017	**2**		1015	**2**

Table 5. Removal rates in the study by White & Cousins (2013). Troughs of 1.15 m^2 and 3.03 m^2 were used with 100% island coverage and soluble fertiliser added to pond water as the inflow.

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	$mg/m^2/day$				37.2	15.4	**59**
TN	$mg/m^2/day$				320	106	**67**

Table 6. Removal rates in the study by Yang *et al.* (2008). This was a lab based hydroponic study with synthetic effluent (dissolved P and N, but also organic matter used), although the objective was to represent a nursery run-off treatment system, which naturally has few suspended solids. 100% island coverage was used with purposely anaerobic conditions, 3 day batch process, and glucose added to aid denitrification. Thus, high NO_x removal rates were obtained, but NH_3 removal was negative as decomposition of organics was still taking place but with limited or no nitrification.

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l				1.25	1.17	**6**
TN	mg/l				3.76	2.59	**31**
NH3	mg/l				0.93	1.19	**-28**
NO_x	mg/l				1.39	0.12	**91**
COD	mg/l				41.8	34.8	**17**
DO	mg/l				0.01	0	**100**

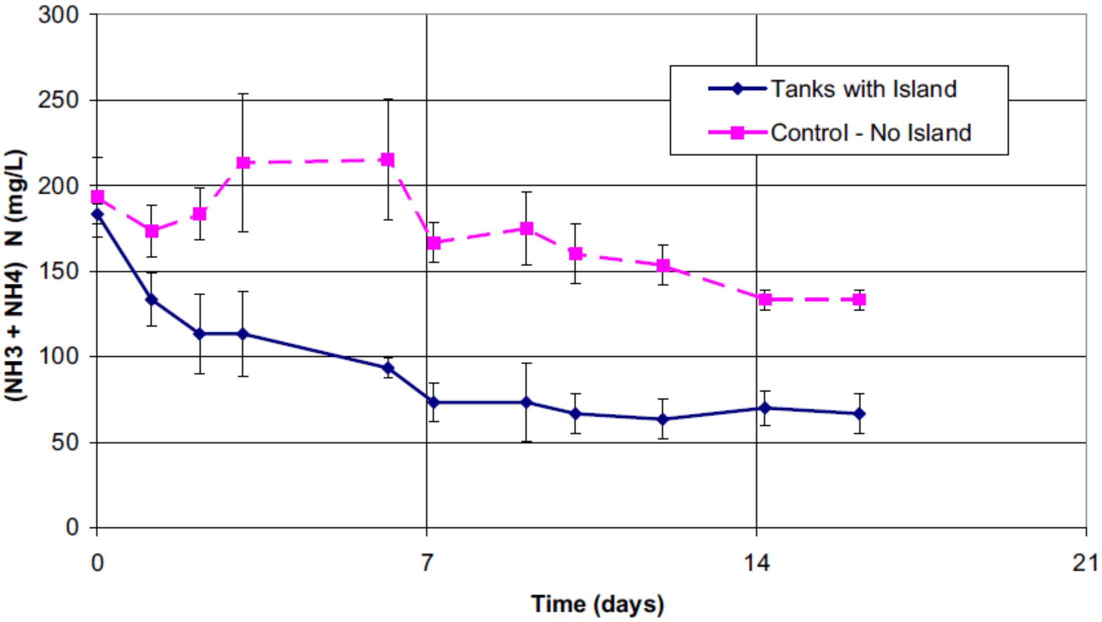

Figure 5. Ammonium removal rates; graph from the study by Stewart *et al.* (2008). An **aerobic** lab experiment with 100% island coverage, calcium carbonate added, and aerated with a bubbler. Synthetic effluent was created using liquid fertiliser (soluble). Conditions were optimised for ammonium removal.

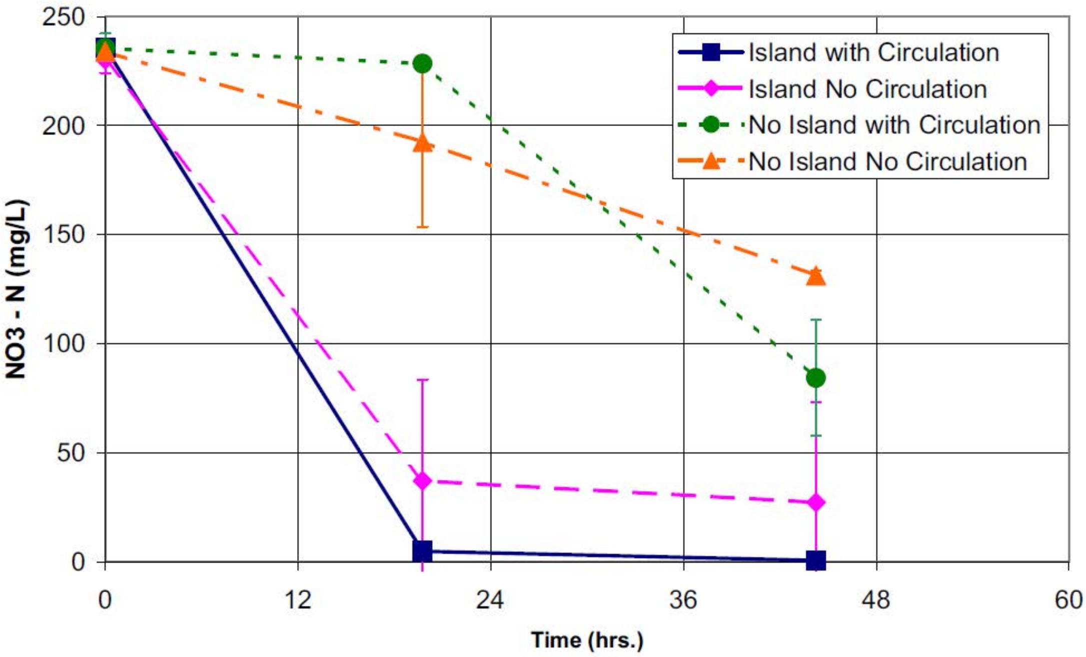

Figure 6. Nitrate removal rates; graph from the study by Stewart *et al.* (2008). An **anaerobic** lab experiment with 100% island coverage and carbon (molasses) added. In some of the replicates water was circulated by a pump. Synthetic effluent was created using liquid fertiliser (soluble). Conditions were optimised for nitrate removal. Redox potential in the control decreased from +200mV to +48mV, but in tanks with islands it decreased to -200mV (much better for denitrification).

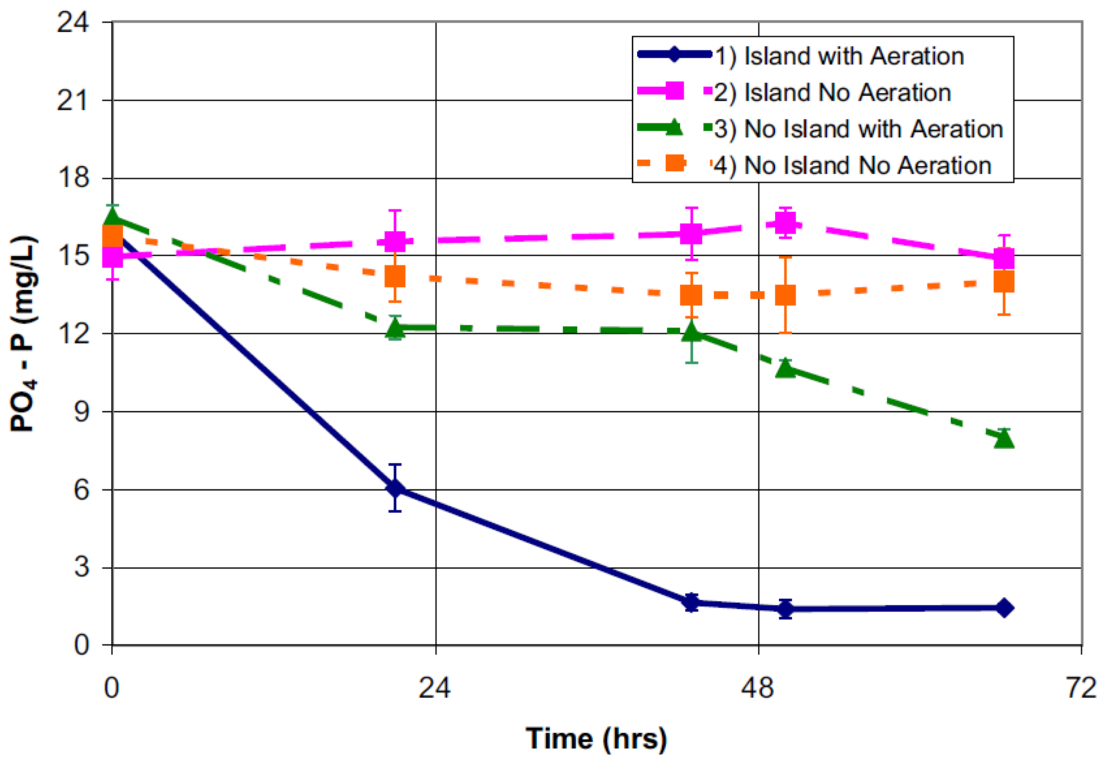

Figure 7. Phosphate removal rates; graph from the study by Stewart *et al.* (2008). Both **anaerobic and aerobic** basins were tested (conditions as in Figures 5 and 6) without islands and with 100% cover of islands. Phosphate removal was best achieved when there was both aeration and floating islands.

Table 7. Removal rates in the study by Borne et al. (2013). A control stormwater retention pond was compared with a retention pond with 50% cover of floating island, receiving the same effluent. Data was retrieved from a graphical presentation of inflow and outflow effluent concentrations.

	control			FTW		
	inflow	outflow	**% removal**	inflow	outflow	**% removal**
TSS	30	24	**20**	*identical*	12	**60**
Cu_{tot}	0.0090	0.0075	**17**	*to*	0.0057	**37**
Cu_{part}	0.0035	0.0030	**14**	*control*	0.0019	**46**
Cu_{diss}	0.0049	0.0044	**10**	*inflow*	0.0038	**22**
Zn_{tot}	0.035	0.022	**37**		0.013	**63**
Zn_{part}	0.027	0.017	**37**		0.010	**63**
Zn_{diss}	0.006	0.005	**17**		0.005	**17**

Table 8. Removal rates in the study by Tanner & Headley (2011). After 7 days batch experiment with 1m x 1m mesocosms and 36% island cover. Artificial stormwater used. FTW results are from the plant species which gave best results (*Cyperus ustulatus*).

		control			**FTW**	
	inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP			**3**			**58**
DRP			**-5**			**60**
Cu_{tot}			**7**			**57**
Cu_{diss}			**5**			**50**
Zn_{tot}			**-1**			**19**
Zn_{diss}			**1**			**37**
Turbidity - subsurface			**24**			**67**
Turbidity - bottom			**24**			**67**
DO (subsurface)			**8**			**39**
DO (bottom)			**11**			**40**

Table 9. Removal rates in the study by Stefani *et al.* (2011) based on median values. Effluent was from aquaculture, following conventional activated sludge treatment. There was a 19% cover of islands and a continuous flow (0.09 m/s).

			control			**FTW**	
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l				0.55	0.19	**65**
SS	mg/l				350	320	**9**
COD	mg/l				15	5	**67**
BOD	mg/l				4.2	2	**52**
pH					7.3	7.2	**1**
cond	µS/cm				645	645	**0**

Table 10. Removal rates in the study by Winston *et al.* (2013). The study examined a stormwater retention pond before (control) and after (FTW) retrofitting an **18%** coverage of floating island. Data is a mean over different storm events.

			control			**FTW**	
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	0.26	0.11	**58**	0.41	0.05	**88**
PBP	mg/l	0.13	0.04	**69**	0.17	0.03	**82**
OP	mg/l	0.13	0.07	**46**	0.24	0.02	**92**
TN	mg/l	1.01	0.41	**59**	3.49	0.43	**88**
NH_3	mg/l	0.10	0.05	**50**	1.6	0.04	**98**
TKN	mg/l	0.88	0.35	**60**	3.32	0.37	**89**
NO_x	mg/l	0.12	0.06	**50**	0.17	0.06	**65**
N_{org}	mg/l	0.89	0.34	**62**	1.72	0.33	**81**
TSS	mg/l	216	24	**89**	252	13	**95**

Table 11. Removal rates in the study by Winston *et al.* (2013). The study examined a stormwater retention pond before (control) and after (FTW) retrofitting an **9%** coverage of floating island. Data is a mean over different storm events.

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	0.26	0.17	**35**	0.19	0.12	**37**
PBP	mg/l	0.13	0.05	**62**	0.07	0.05	**29**
OP	mg/l	0.14	0.12	**14**	0.12	0.07	**42**
TN	mg/l	1.64	1.05	**36**	1.17	0.61	**48**
NH_3	mg/l	0.12	0.11	**8**	0.11	0.05	**55**
TKN	mg/l	1.43	0.97	**32**	0.84	0.55	**35**
NO_x	mg/l	0.20	0.08	**60**	0.34	0.06	**82**
N_{org}	mg/l	1.50	0.93	**38**	0.72	0.5	**31**
TSS		354	30	**92**	101	22	**78**

Table 12. Removal rates in the study by Chang *et al.* (2013) **during** storm events in a functioning stormwater retention pond; assessed before (control) and after (FTW) fitting floating islands with 8.7% cover. Nutrient concentrations are given as the means over several different storm events. The pond contained a fountain.

		Control			FTW		
		inflow	Outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	0.028	0.027	**4**	0.058	0.050	**14**
OP	mg/l	0.006	0.006	**0**	0.021	0.010	**52**
TN	mg/l	0.300	0.377	**-26**	0.626	0.526	**16**
NH_3	mg/l	0.048	0.052	**-8**	0.102	0.104	**-2**
NO_x	mg/l	0.006	0.017	**-183**	0.062	0.029	**53**

Table 13. Removal rates in the study by Chang *et al.* (2013) **outside** of storm events in a functioning stormwater retention pond; assessed before (control) and after (FTW) fitting floating islands with 8.7% cover. Nutrient concentrations are given as the means over different sampling times. Notice that the treatment rates are much higher than during storm events (probably due to lower flows and thus higher retention times).

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	0.037	0.034	**8**	0.055	0.029	**47**
OP	mg/l	0.003	0.002	**33**	0.020	0.004	**80**
TN	mg/l	0.303	0.349	**-15**	0.655	0.552	**16**
NH_3	mg/l	0.121	0.103	**15**	0.208	0.102	**51**
NO_x	mg/l	0.025	0.022	**12**	0.032	0.025	**22**

Overview

With more cover of floating islands there is a tendency for redox potential to drop due to reduced O_2 diffusion from atmosphere. This leads to denitrification processes dominating in which there is high removal of NO_3 but low removal of NH_4 (Yang et al., 2008; Van de Moortel et al., 2010). Indeed NH_4 removal may be negative due to decomposition of organics to NH_4 without subsequent removal by nitrification (Table 4 and 6). Aeration can prevent this NH_4 accumulation by encouraging nitrification, and it also prevents P release from sediments that can occur at low redox potentials (Figure 7).

Low % island cover had detrimental effects on treatment efficiency, as did lower residence times. For example, TN removal was 48% with 9% island cover in the Winston et al. study (2013), but this increased to 88% TN removal with 18% cover (Tables 10 and 11). Similarly Chang *et al.* (2013) found only 14% TP removal with 9% cover during storms, but outside of storm flows this removal increased to 47% (Tables 12 and 13).

Around 20% cover seems optimal if the basin is to be maintained as an aerobic system without artificial aeration, and still achieve good removal efficiency. Beyond this point it is probably worth using 100% cover, with a choice between a high nitrate removal anaerobic basin, or artificial aeration (bubbling) to produce a high treatment rate aerobic basin. Stewart et al. (2008) illustrates how tightly controlled conditions and addition of calcium carbonate (nitrification) or carbon (denitrification) can be used to optimise treatment rates. Stewart et al. (2008) also showed that nitrification and denitrification processes can be achieved in a single aerobic tank if tightly controlled. Treatment efficiencies noted by Stewart and White & Cousins (2013) are likely to be around the maximum achievable in FTWs due to the use of soluble fertilisers in their experiments and their tightly controlled hydroponic systems. Therefore when assessing potential performance of a new FTW we must decide whether it will be a tightly controlled situation or more of a field based FTW.

Table 14 summarises these studies. The improvements through using Floating Islands, as discussed, vary due to conditions, however we can expect between around 2 and 55% increase in P removal compared to a Free Water Surface wetland, and a 12 to 42% increase in N removal. Metal removal is also considerably higher in FTWs (20-50% higher). Most importantly, if conditions are tailored for denitrification (anaerobic and sufficient carbon supply) NO_3 removal can be up to 100% in FTWs.

Table 14. Summary of % removal rates in different studies for main nutrients and metals. 'Improvement' examines the increase of treatment efficiency of FTWs beyond the control wetlands (FST wetlands). The Van de Moortel study is excluded from the fianl comparison since anaerobic conditions produced P release and is not an example of good FTW management.

	Study	Moortel	White	Yang	Stewart*	Stewart*	Borne*	Tanner	Stefani	Winston	Winston	Chang	Chang	**RANGE**
	% cover	**100**	**100**	**100**	**100**	**100**	**50**	**50**	**19**	**18**	**9**	**8.7**	**8.7**	**8.7-100**
	notes	anaerobic	anaerobic	anaerobic	aerobic	anaerobic						storm	non-storm	
FTW	TP	12	59	6	91			58	65	88	37	14	47	**6-91**
FTW	TN	11	67	31						88	48	16	16	**11-88**
FTW	NH_4	-2		-28	66					98	55	-2	51	**-28-66**
FTW	NO_3/NO_x	78		91		100				65	82	53	22	**22-100**
FTW	Cu_{tot}	16					37	57						**16-57**
FTW	Zn_{tot}	17					63	19						**17-63**
controls	TP	18			53			3		58	35	4	8	**3-58**
controls	TN	40								59	36	-26	-15	**-26-59**
controls	NH_4	33			26					50	8	-8	15	**-8-33**
controls	NO_3/NO_x	46				42				50	60	-183	12	**-183-60**
controls	Cu_{tot}	45					17	7						**7-45**
controls	Zn_{tot}	48					37	-1						**-1-48**
improvement	TP				38			55		30	2	10	39	**2-55**
improvement	TN									29	12	42	31	**12-42**
improvement	NH_4	*study*			40					48	47	6	36	**6-48**
improvement	NO_3/NO_x	*excluded*				58				15	22	236	10	**10-236**
improvement	Cu_{tot}						20	50						**20-50**
improvement	Zn_{tot}						26	20						**20-26**

Notes:
(*) indicates data extracted from graphs.
In Stewart study, PO_4^{3-} was assessed instead of P

3.1 Seasonal Variation

Seasonal variation in FTW is due to (i) temperature variations, which affect plant and especially microbial productivity, (ii) consequent DO variations due to increased oxygen demand when there is increased microbiological activity, and to some extent the solubility of oxygen in water at different temperatures, and (iii) seasonal growth patterns in plants.

The effect of season on treatment efficiency depends on the main processes involved in their removal, particularly how temperature and oxygen variations affect these processes. For example, spring and autumn are peak P uptake periods for vegetation in wetlands (Kadlec & Wallace, 2009) however, the main process of P removal is settling and adsorption, so seasonal P removal was found to vary less than that of nitrogen (Wittgren & Maehlum, 1997).

Studies have shown conflicting results over how variable treatment efficiency is over different seasons, particularly with N removal, but this is likely to be due to differences in limiting factors. As previously mentioned, plant uptake as NO_3 or NH_4^+ tends to be relatively small compared to microbiological processes (Riley et al., 2005). Thus, studies have found N removal to be affected by seasonal temperature variation (Spieles & Mitsch, 2000; Picard et al., 2005). However, Maehlum and Stalnacke (1999) and Mander et al. (2000) found little difference in N removal between warm and cold climates and Van de Moortel et al. (2010) found more variation due to temperature in P than in N. It is likely that these differences are due to other factors that may be limiting, particularly anoxia. For example, in the study by Van de Moortel et al. (2010) there was low NH_4 removal as 100% island coverage produced low DO and reducing conditions, nullifying any further potential N removal increases due to increased temperature. Also, as mentioned previously, in practice decomposition is not found to be highly temperature dependent in wetlands (Akratos & Tsihrintzis, 2007; Kadlec & Wallace, 2009) and therefore ammonia production rates are not likely to change much with temperature. Thus, interactions between season, light, temperature and DO mean that an individual variable is not a good predictor of activity, and the net effect can be counter-intuitive (Stein & Hook, 2005; Kadlec & Wallace, 2009). However, despite these interacting effects very low temperatures (5 °C) certainly restrict microbiological activity and plant growth (Mitsch and Gosselink 1993).

Rainfall

Rainfall can have a large and varied effect on pollutants entering a basin. If the inflow is from a combined sewer system rainfall events can massively increase dilution and flow rates into the wetland. Van de Moortel et al. (2010) found that heavy rainfall caused a significant reduction in inflow conductivity from 1102 µS to 733 µS, and total nitrogen from 23.1 mg TN/l to 16.9 mg TN/l. Then, after rain events although other constitutions remained diluted in the pond, ammonium and nitrate concentrations actually increased (probably due to microbiological activity). With stormwater treatment ponds, the inflow comes from road run-off which has often had an accumulation of metals during the dry period, so initial concentrations during a storm are usually high, as the metals and particulates get washed off the road, but then rapidly decrease as the storm continues and the concentrations become diluted (Barbosa & Dodkins, 2010).

Rainfall and evaporation also have an effect on dilution within the basins (Kadlec & Wallace, 2009). The addition of rainwater can alter the water chemistry (oxygen, pH), rates of microbiological activity, and affect physical processes e.g. increased depth increasing settling. It is also important to consider that when measuring inflow and outflow concentrations, differences may be due to changes in dilution, rather than any removal within the basin, and loading capacities must take rainfall input and evaporation (and drainage) into consideration (Kadlec & Wallace, 2009).

Shading and Temperature
Floating islands can significantly reduce water temperature (by shading) in the warmer months, and also reduce temperature variation if there is sufficient cover (Van de Moortel et al., 2010). However, Winston et al. (2013) with only 18% island cover, found there was little water temperature reduction (preventing them producing conditions for trout to live in the FTW). Van de Moortel et al (2010) also found that although summer tempertures were lower in FTW compared to open water wetlands, winter temperatures were not lower. However, ice still persisted longer in FTWs during the winter due to reduced wind disturbance at the surface in FTWs.

4. Design Considerations

To achieve treatment objectives careful consideration must be taken in design and operation of the wetland. These must be specific to the flow volume, flow variation, the concentrations of pollutant and the required characteristics of outflow. Wetlands can easily be overloaded with sludge so pre-treatment (removal of large material by bar screen or settling of grit and stones) and primary treatment (sedimentation) are essential for domestic effluents prior to entering the wetland. Good design of these initial stages is also extremely important in maximising the treatment efficiency and the cost of running a FTW and to prevent them becoming unnecessarily clogged by high sludge loadings.

4.1 Island Cover

Since floating islands can restrict oxygen diffusion from the air into the water (Smith & Kalin, 2000), island coverage is an extremely important design factor. For example, an almost complete coverage by islands resulted in poor P retention in sediments due to anoxia (Van de Moortel et al., 2010).

The percentage cover of a pond by the island is one of the most important considerations in FTW design. High cover (>50%) can cause anoxia but low cover (9 to 18%) may produce little additional treatment effect (e.g. Winston et al., 2013). The anoxia is not only caused by islands reducing air-water contact, but also because of the high rate of microbiological processes such as nitrification and decomposition. Thus, the optimum size of the island to prevent anoxia is likely to be dependent on the quality of the influent, particularly ammonia/nitrate and organic carbon concentrations. Flow design, mixing and aeration will also be major factors (See section 3. Treatment Efficiency).

Using more island coverage should increase microbiological activity due to the larger root area, however if high rates of aerobic microbiological activity is to be maintained (e.g. nitrification) oxygen consumption will necessarily be high. To maintain high island coverage without depleting oxygen, bubblers can be installed (Stewart et al., 2008). This requires investment and energy costs, and therefore their use depends on a cost-benefit analysis, although energy can be provided by e.g. solar power. Baffles have also been introduced in some FTWs to increase circulation around the roots. Mixing waters to promote aeration should be done with care as disturbance of sediments can liberate trapped P.

4.2 Optimising for N removal

Since P is effectively conservative, but N can be released as gas through correctly managing the microbial environment, strategies for permanently removing N are very different from those for removing P.

Aerobic and Anaerobic basins

Floating islands may increase denitrification by increasing anoxia, although it is preferable to have an oxygenated basin with a high residence time as a first stage to convert most of the ammonia to nitrate in the nitrification process.

Thus, with N the main objective is to convert as much ammonium as possible to nitrate, usually through an aerobic 1st stage, and then to convert as much of this nitrate to N_2 gas, through an anoxic 2nd stage. Oxygen in the aerobic stage can be rapidly depleted with high coverage of FIs and high rates of microbiological activity, so FI cover has to be carefully managed, or artificial aeration has to be included. Sufficient alkalinity must also be available for nitrification, which can be achieved through the addition of $CaCO_3$ (Stewart et al., 2008).

Nitrification reduces DO and pH, though these conditions are ideal for the next (anoxic) denitrification stage. FI cover can be much higher at this stage. Yang et al. (2008) achieved 97% N removal rates with 0% DO in a hydroponic system, though in more natural systems anoxia can cause sulphide toxicity (Lamers et al., 2002) that kill or restrict root growth.

Recycling
Denitrification is predominantly limited by C supply, with a recommended C:N loading ratio of 5:1 (Bishay & Kadlec, 2005). In a two stage system Carbon limitation often occurs because much of the organic C is removed by settling in the earlier aerobic stage (Kadlec & Wallace, 2009), thus releasing nitrate. Additional C can be supplied artificially, e.g. as glucose syrup (Yang et al., 2008) but for most effluent treatment systems it is cheaper and more practical to seed the anoxic basin with raw effluent that has not gone through the anaerobic stage (Kadlec & Wallace, 2009).

Recycling is also used to return anaerobic outflow back to the aerobic stage; denitrification makes the effluent more alkaline, ideal for further nitrification of ammonia (Kadlec & Wallace, 2009). Recycling is now in Danish treatment wetland guidelines (Brix & Schierup, 1990). Figure 5 gives an example of how FTW wetlands could be designed for treatment of domestic effluent, including recycling.

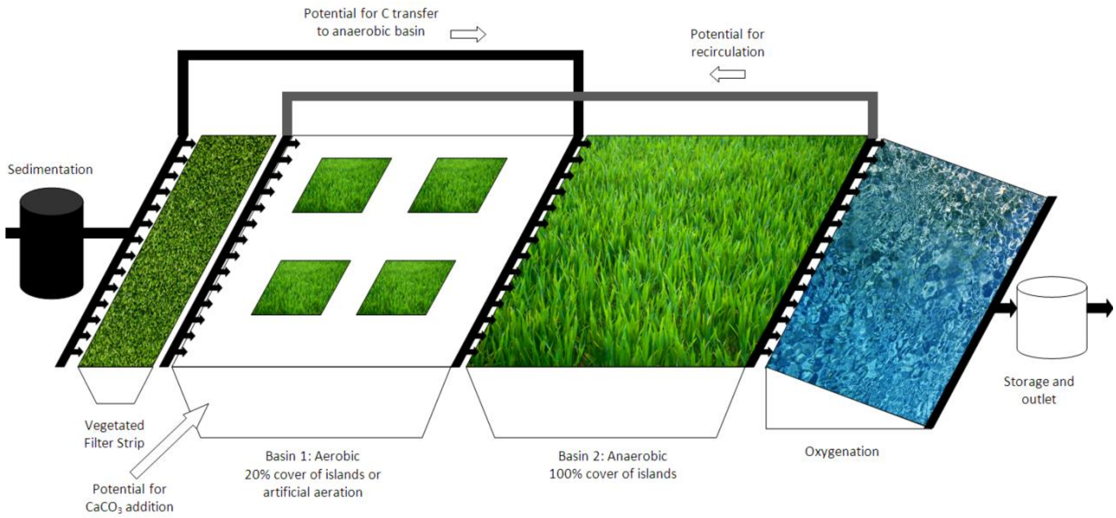

Figure 5. A theoretical design using FTWs for treating low volume domestic effluent (mixture of P, BOD, NH_4^+ and NO_3^- inputs) showing basic design features of combined basins. The vegetated filter strip would have to be adapted/increased/removed depending on solids input from sedimentation.

4.3 Plants

The treatment potential within a FTW depends mostly on the filtering capacity of the roots (root depth and density) and their surface area as a microbiological habitat. Choice of plant species will also affect the rates of nutrient and metal uptake, root/shoot biomass division, growth rates and the way in which the basin water chemistry is altered due to the release of humic acids and protons by plant roots.

Plant dimensions
Tanner & Headley (2011) examined four species growing on floating islands in mesocosms, providing detailed measurements. 90th percentile of root depth averaged between 24 and 48cm, depending on species. The root surface area was between 4.6 and 9.3 m^2/m^2 of floating mat. Above mat biomass was between 834 and 2350 g/m^2 and root biomass of 184-

533 g/m^2 (see Table 3 for more details) with shoot to root ratios of between 3.7 and 4.5. Winston *et al.* (2013) found that *Hibiscus* had shoot:root ratio of 6.3. Indeed, most species have an above mat biomass greater than the below mat biomass, except for Carex spp. such as Carex stricta (Winston et al., 2013), Carex virgata and Cyperus ustulatus (Tanner & Headley, 2011) (Table 3).

Table 5. Mean biomass and shoot:root ratio for FTW plants.

Plant species	Shoots: Mean above mat biomass (g/m^2)	Roots: Mean below mat biomass (g/m^2)	Biomass ratio
[1]*Juncus spp.*	86.3	43.4	2.0
[1]*Carex stricta*	131.4	207.6	0.6
[1]*Spartina pectinata*	121.7	48.1	2.5
[1]*Hibiscus moscheutos*	269	58.9	4.6
[1]*Pontederia cordata*	72	57.7	1.2
[2]*Cyperus ustilatus*	1528	329	4.6
[2]*Carex virgate*	2350	533	4.4
[2]*Juncus edgariae*	1113	299	3.7
[2]*Schoenoplectus tabernaemontani*	834	184	4.5

[1] Winston et al. (2013) in stormwater retention pond
[2] Tanner & Headley (2011) in mesocosm with much more intensive planting

Plant uptake appears to be more associated with total plant biomass rather than root density (Tanner & Headley, 2011), although White & Cousins (2013) found that uptake of N and P by *Juncus effusus* (60.6 N and 3.71 g P/m^2/growing season) was higher than that of *Canna flaccida* (3.71 N and 2.27 g P/m^2/growing season) despite having a similar shoot length, and this was attributed to the much longer roots of *J. effusus*. Nutrient uptake by *J. effusus* was also found to be much higher than that of *Pontederia cordata* in a study by (Chang et al., 2013).

Floating islands are usually allowed 6 months of plant growth to establish before assessing efficiency e.g. (Borne et al., 2013). Once plant growth has reached a maximum (maximum density and shoot biomass) there is no additional net uptake of nutrients by the plants i.e. litter deposition is equal to growth (Kadlec & Wallace, 2009). However, although some of this litter will accumulate on the island, some will sink into the basin with some nutrient release but also with some C and P storage in the sediments.

White & Cousins (2013) found that an increase in nutrient loading increased shoot growth (but not root growth) and suggested that this may be due to a shift in allocation strategy towards shoots when nutrients are plentiful, following Muller, Shmid & Weiner (2000). However, harvesting of the shoots does not appear to affect the root biomass (Borne et al., 2013).

Plant establishment

Vogel (2011) noted that floating island plants have more establishment success and establish quicker, with more cover, when the starting biomass is higher. She recommends planting of as much biomass stock as possible at the start, to aid establishment. The growth rate for some plants may be higher in the first year of establishment, whilst other plants may have higher growth rate in the second year (Svengsouk & Mitsch, 2001).

Buoyancy of islands
Buoyancy of vegetated islands changes seasonally, with mats sinking several centimetres during the spring and summer as the biomass increases (Hogg and Wein 1988 a). Seasonal effects become less pronounced with age, as dead biomass accumulates and decomposition increases (matching biomass accumulation).

4.4 Activated Carbon

Activated carbon is being considered by Frog Environmental Ltd. as a possibility for improving floating island performance prior to the complete establishment of plants by incorporating the carbon within the floating island material. Performance of floating islands usually relates to their ability to increase removal rates for P and N as well as to remove metals, commonly Cu and Zn.

Activated carbon is used in water filters and chemical purification processes. It is highly porous carbon with a high surface area which has been treated by oxygen or sulphuric acid to increase adsorption. It has a surface area of 300-2,000 m^2/g and can adsorb a wide range of pollutants including large organic molecules. Because adsorption works by chemically binding the impurities to the carbon, the active sites in the carbon eventually become filled and adsorption stops. The effectiveness of activated carbon depends on pore size, the carbon source and the manufacturing process.

Typically activated carbon is used to remove metals or organic pollutants rather than nutrients. This is because the surface of activated carbon is negatively charged, attracting positive ions (e.g. Cu^{2+}, Zn^{2+}) rather than negative ions (NO_2^-, NO_3^-). Bhatnagar & Sillanpää (2011) reviewed the adsorption of nitrate on to various carbon substances. Results vary with 1mg/g (Mizuta et al., 2004), 1.7 mg/g (Bhatnagara* et al., 2008) and 4 mg/g (Oztürk & Bektaş, 2004) adsorption of NO_3^-, although these studies are all done in lab conditions and are better than can be expected in the field. Biochar (a form of charcoal) has been tested in field for nitrate removal, though it has tended to have low effectiveness except where nut shells have been the carbon source of biochar (Knowles et al., 2011; Yao et al., 2012).

Nitrate adsorption depends on contact time. Oztürk and Bektaş (2004) achieved complete adsorption within 1 hour at pH<5.0 and 25 °C. Optimal pH for activated carbon adsorption of nitrate occurs at pH2. This is because H+ ions bind to surface and reduce –ve charge, increasing uptake of –ve ions (NO_3^-). Problems with extrapolating these results to the field include (i) the nitrate could be bound to other substances within the water column or sediments, (ii) there would be a diffusion gradient between the site of adsorption (the island) and the bottom of the basin, (iii) the effluent is not being passed through the carbon, so adsorption is passive (iv) optimal pHs for adsorption would not be suitable for a treatment basin, which should be kept around neutral pH.

Ammonia adsorption is around 5.08 mg NH_3/g of carbon at 20 °C increasing to 5.80 mg/g at 60 °C (Long et al., 2008). The temperature of activation of the carbon also affects the adsorption capacity, with higher activation temperatures increasing ammonia adsorption (Ghauri et al., 2012).

P removal in wetlands tends to be predominantly through physical sedimentation processes, which are aided by particle interception by plant roots. When P removal was tested with activated carbon adsorption capacity was 1.11 mg /g at high P concentrations, decreasing with lower P concentrations (Liang et al., 2011). Optimum pH for adsorption ranges between 6 and 10 (Kumarab et al., 2010), which is ideal for FTWs, although again, these studies use data for filtration of nutrients rather than passive adsorption at the surface of the basin.

Adsorption rates of different nutrients are dependent on the nutrient concentration in the effluent, and at low concentrations close to 100% removal is theoretically possible. However, with low circulation and a diffusion gradient within a treatment basin it is unlikely that high percentage removal rates are possible. In addition, we would expect to use around 100 times more carbon (by weight) than the nutrient we are reclaiming, which is likely to be prohibitively expensive.

Activated carbon can provide a carbon source for improving denitrification when C is limiting (Isaacs & Henze 1995; Yang *et al.* 2008). This may be particularly important prior to the establishment of vegetation, which would then provide a source of carbon through decaying organic matter. However, addition between the layers of the floating island may be less useful than simply mixing the powdered activated carbon into the effluent as it enters the basin. Also, a soluble carbon source such as glucose (Yang et al., 2008), acetate or hydrolysate (Isaacs & Henze, 1995) may be better for encouraging denitrification than powdered activated carbon.

5. Conclusions and Recommendations

The main function of floating islands in removing pollutants from effluents is:

- Plant roots assisting in filtering and settling processes for P
- Plant roots acting as a large surface area for micro-organism activity in: decomposition, nitrification, and denitrification (removal of BOD and N).
- Mild acidification of water due to release of humic acids, and a C input from senescent vegetation; assisting denitrification.

P removal is predominantly a physical process. It binds to particulates and removal is assisted by the reduced water movement and the filtering effect of roots on these particulates. This sloughs off to the bottom sediments. Metals are also removed predominantly through binding to particles and sedimentation. Reduced DO in the basin and disturbance of the sediments can result in release of P and metals from the sediments. P is effectively conservative, and if dredging of the sediments is not done (around every 10 years is suggested) the sediment bound P and dissolved P will reach an equilibrium whereby there is no net P removal (and potential for pulses of P in the outflow which are higher than that in the inflow).

N removal is predominantly a microbiological process with NH_4^+ being nitrified to NO_3^- in aerobic basins by nitrifying bacteria, then NO_3^- being denitrified to N_2 gas (and thus released) in anaerobic basins by denitrifying bacteria. FTWs have excellent potential for removing N from effluents. An initial aerobic basin (up to 20% island cover or 100% with aeration) can be used for nitrification and then a second anaerobic basin (100% island cover) can be used for denitrification. Up to 100% N removal is possible, with more tightly controlled conditions increasing the ability to remove N. At the aerobic stage the addition of $CaCO_3^-$ can assist with nitrification (as alkalinity is used up during this process). At the anaerobic stage the addition of C can assist with denitrification (as carbon compounds are used during this process). This C may be added as e.g. glucose or molasses, or as BOD fed from the FTW inlet.

With good management but without hydroponic conditions (i.e. aeration, $CaCO_3$ or artificial C addition) we could expect a FTW wetland to achieve around 60% removal of TP, 75% removal of TN, 50% removal of NH_4^+, 80% removal of NO_3 and 40% removal of metals. All these are expected to be significant improvements (around 20-40% higher) than with basins without islands, depending on specific conditions. More controlled conditions could considerably increase the treatment rates.

Plant uptake only accounts for up to 6% of nutrient (N and P) removal in FTWs. This is also recycled into the system through decomposition unless harvesting is undertaken. Although concentrations of nutrients and some metals (e.g. Zn) are higher in the roots, shoot biomass of plants tends to be higher. Thus shoot harvesting often removes a little over half of the nutrients taken up by the plants. Floating islands also provide access for root harvesting, but harvesting of roots is unlikely to be beneficial as it is more time consuming and also reduces the filtering capacity and microbiological activity associated with the root network: the principal mechanisms of nutrient removal in FTWs. Evidence suggests that removal of shoots does not negatively affect the roots.

FTWs have other advantages over conventional Free Water Surface Wetlands:

- They can adjust to varying water levels

- A higher retention time is possible as they can be made deeper without submerging the vegetation
- Habitat value for birds/amphibians

Recommendations for domestic effluent treatment:
Domestic effluent usually has high BOD, NH_4^+, NO_3^- and P although specific operation and design of FTWs should be tailored to the specific characteristics of the inflow.

1. N removal is the principal benefit of FTWs:
   - An aerobic basin for nitrification is required to convert ammonia to nitrate
   - An anaerobic basin for denitrification is required to convert nitrate to N_2 gas.
   - Although nitrification and denitrification can be achieved in the same basin, separate aerobic/anaerobic basins can be used to more easily control the processes.
2. Depending on cost considerations and inflow water alkalinity, $CaCO_3$ can be added in the aerobic basin to aid nitrification.
3. C can be added in the anaerobic basin to aid denitrification. In smaller treatment systems requiring high water quality outflow a hydroponic system with glucose or molasses addition can be used. For larger treatment systems with greater costs considerations, input of C can come from a controlled input of BOD directly from the FTW inflow.
4. 100% cover of islands, with mixing, is optimal for N reduction in the anaerobic basin.
5. 100% cover of islands with aeration (bubbling) is optimal for aerobic nitrification. If cost considerations prevent aeration, 20% island cover is recommended in the aerobic basin to prevent anoxia occurring.
6. A recycling system from the anaerobic to the aerobic basin, although not always necessary, may be useful when there is excessive NH_4^+ in the outflow i.e. to increase nitrification rates.
7. Aeration is required after the denitrification basin to prevent the release of anoxic waters to the environment.
8. Circum-neutral pH should be maintained in anaerobic and aerobic basins. If pH drops considerably there is a danger of P release.
9. Dredging, particularly of the first (aerobic) basin is recommended every 10 years to remove P trapped in sediments, as well as accumulating metals. Alternative (dormant) treatment basins may be required to be made operational treat effluent as dredging operations are undertaken in the main basin.
10. Plants with high root surface area and high plant biomass are recommended for the floating islands e.g. *Juncus effusus*. Ecological considerations may result in other species being chosen or plant mixtures being used.
11. Harvesting should be done, but only of the shoots.

Use of Activated Carbon
The use of activated carbon between layers of floating island material to assist in pollutant removal will probably have limited effectiveness. This is due to a diffusion gradient between the surface of the basin and the bottom of the basin, and the passive nature of adsorption i.e. the effluent is not being filtered through the medium. At the very most (with high retention times and full adsorption) N and P removal is likely to be about 1g for every 100g of activated carbon used. The proper establishment of plants, a focus on correct basin design, and water chemistry control, is likely to be a much more effective use of resources.

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Rev 4-01-14 Sorted – Most Recent first.
Journal Articles first, Magazine and On-Line Articles second, Reports third.

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(1) Articles Published in Scientific Journals (typically peer-reviewed)

Floating Treatment Wetlands (FTWs) in Wastewater Treatment: Treatment efficiency and potential benefits of activated carbon.

Type	Review of Published Articles
Author	Dodkins, I; Mendzil, AF; O'Dea, L.
Publication	unpublished
Volume	n/a
Pages	n/a
Date	March 2014

Notes:

This report was prepared by Swansea University staff for FROG Environmental, Ltd. It includes historical information and discussions of treatment efficacies for common contaminants studied by numerous researchers. Detailed information is provided for the various pathways for phosphorus removal (Section 2.1), nitrogen removal (Section 2.2), oxygen issues and effects (Section 2.3 – 2.4); BOD, suspended solids and carbon (Section 2.5), metal removal (Section 2.6), plants (Section 2.8), and other factors related to FTW operation.

The report describes methods for optimizing removal of specific contaminants by controlling aeration and pH, determining percent island coverage, using chemical additives, etc.

Executive Summary:
Floating Treatment Wetlands (FTWs) have many benefits over Free Water Surface (FWS) wetlands:
1. Plant roots assisting in filtering and settling processes for sediment bound P and metals
2. Plant roots acting as a large surface area for micro-‐organism activity in: decomposition, nitrification, and denitrification (removal of BOD and N).
3. Mild acidification of water due to release of humic acids; and a C input from senescent vegetation, assisting denitrification.
4. They can adjust to varying water levels
5. A higher retention time is possible as they can be made deeper without submerging the vegetation
Percentage removal of nutrients and metals from effluent is around 20-‐40% higher in FTWs than in conventional FWS ponds. Removal efficiency, particularly of nitrogen, can be further increased with tighter control on the water chemistry (aeration; adding CaCO3; adding a carbon source). 20% coverage of islands is optimal for aerobic basins. 100% cover is optimal for anaerobic basins or aerobic basins where there is artificial aeration. The design the FTW and the control of basin water chemistry is essential for optimising treatment efficiencies. The passive use of activated carbon within layers of floating islands is unlikely to be cost effective.

Exploring hydrobiogeochemical processes of floating treatment wetlands in a subtropical stormwater wet detention pond

Type	Journal Article
Author	Chang, N.B, Z. Xuan. Z Marimon, K. Islam, P. Wanielista
Publication	Ecological Engineering
Volume	54
Pages	66-76
Date	2013

Notes:

This peer-reviewed article covers a study of BioHaven islands and Beemat products that was reported in a previous University of Central Florida report.

Abstract:

Floating treatment wetland (FTW) is one of the emerging best management practices (BMPs) for stormwater treatment where *macrophytes provide a suitable root-zone environment for microorganisms that allow the plants to remove nutrients through direct uptake into their tissue. In this study, four floating mats with native Florida aquatic macrophytes were deployed in a 340 m2 subtropical stormwater wet detention pond. A fountain in the pond and peat moss used to hold the substrate for plant species on the floating mats are both assumed to add nutrients to the water column. The aim of this study was to evaluate the performance of nutrient removal through the four floating mats and explore associated effects of simultaneous hydrological and biological controls related to various hydrobiogeochemical processes for nutrient removal in a multimedia pond environment. Nutrient concentrations in both inlet and outlet were monitored continuously over 13 months, with episodic (storm events) and routine (non-storm events) sampling plans carried out in parallel to justify the efficacy of the FTWs. Nutrient values within the water column and the sediment were compared before (Phase I) and after (Phase II) the deployment of the FTWs to prove the proposed hypotheses. An additional phase (Phase III) after the removal of the FTWs was added to enhance the understanding of ecosystem response. For non-storm events, phosphorus removal was substantial because of the increase in the initial concentrations, presumably due to resuspension of nutrients into the water column from the fountain operation; about 47.7% total phosphorus (TP) and 79.0% orthophosphate (OP) were removed. The removal rates of total nitrogen (TN), nitrite– and nitrate–nitrogen (NOx–N = NO−2 –N + NO−3 –N), and ammonia–nitrogen (NH3–N) were also calculated as 15.7, 20.6, and 51.1%, respectively. Without the uptake by plants, the nutrient removal decreased to different degrees when comparing those in non-storm events during Phase II. Considering plant species, nutrient uptake and assimilation by soft rush (Juncus effusus) was much higher than that by pickerelweed (Pontederia cordata) through both leaves and roots in this case. For soft rush, uptake rate in spring is much higher than that in fall. About 77.0 g N and 8.8 g P were removed from pond water via uptake and assimilation during the second phase. Despite organic nitrogen accumulation due to the pickerelweed leaf debris sedimentation, the organic nitrogen concentration in pond water was still kept at a low level, which implies that the ecosystem is capable of efficiently managing the withered plants and circulating nutrients.*

Floating Treatment Wetland Retrofit to Improve Stormwater Pond Performance for Suspended Solids, Copper and Zinc.

Type Journal Article
Author Borne, Karine E., E.A. Fassman, C.C. Tanner
Publication Ecological Engineering
Volume 54
Pages 173-182
Date 2013

Notes:

This peer-reviewed article summarizes a sophisticated stormwater study that utilized automatic samplers, remote flow measurement sensors, complex math analyses, etc.

The study shows that, when plant roots are long enough to reach near the bottom of a pond and water is flowing through the pond, results are good. There was a plant die-off observed at the end of summer, which produced toxic conditions beneath the mat. This was attributed to severe deoxygenation, which would have affected biofilm performance as well. FTW coverage of the pond was 50%.

This was a controlled study with two parallel ponds. The pond with FTW had 41% better removal for TSS than the control pond, 40% for particulate ZN, and 39% for particulate copper.

Abstract:
A field trial study with side-by-side monitoring of two parallel stormwater treatment ponds, one of which contained a floating treatment wetland (FTW), has been carried out to assess the benefit of retrofitting a conventional retention pond with a FTW. Inflow and outflow event mean concentrations (EMCs) were quantified and used to assess the overall pollutant removal efficiency of each system. Findings show that a FTW can significantly improve the runoff water quality and thus reduce the impact on the receiving environment. The present study reveals that a pond retrofit with a FTW would be more efficient than a conventional retention pond, exhibiting a 41% (for total suspended solids – TSS), 40% (for particulate zinc – PZn), 39% (for particulate copper – PCu) and 16% (for dissolved copper – DCu) lower effluent EMC. Physical entrapment of the particulate pollutants into the roots' biofilm seems be a significant removal pathway, which could be impacted by the inflow volume. Due to higher humic content, lower dissolved oxygen and more neutral water column pH induced by the FTW, there was increased potential for adsorption processes and/or precipitation as

insoluble copper sulphides, in addition to the direct Cu uptake by the plants. The dissolved zinc (DZn) inlet EMCs, which already met the Australian and New Zealand Environment Conservation Council (ANZECC) water quality guidelines and could correspond to an irreducible concentration of the system, were too low to differentiate the performance of either pond.

http://hdl.handle.net/2292/21002

Evaluation of floating treatment wetlands as retrofits to existing stormwater retention ponds

Type	Journal Article
Author	Winston, R.J., W.F. Hunt, S.G. Kennedy, L.S. Merriman, J. Chandler, D. Brown
Publication	Ecological Engineering
Volume	54 (2013)
Pages	254-265
Date	March 2013

Notes:

This peer-reviewed article covers the study previously reported by William Hunt, et al.

Abstract:

Thousands of existing wet retention ponds have been built across the United States, primarily for the mitigation of peak flow and removal of sediment. These systems struggle to mitigate soluble nutrient loads from urban watersheds. A simple retrofit for improvement of pond performance for nitrogen and phosphorus removal could become popular. Floating treatment wetlands (FTWs), one such retrofit, are a hydroponic system that provides a growing medium for hydrophytic vegetation, which obtain nutrients from the stormwater pond. Installation of FTWs does not require earth moving, eliminates the need for additional land to be dedicated to treatment, and does not detract from the required storage volume for wet ponds (because they float). To test whether FTWs reduce nutrients and sediment, two ponds in Durham, NC, were monitored pre- and post-FTW installation. At least 16 events were collected from each pond during both monitoring periods. The distinguishing characteristic between the two ponds postretrofit was the fraction of pond surface covered by FTWs; the DOT pond and Museum ponds had 9% and 18%, respectively, of their surface area covered by FTWs. A very small fraction of N and P was taken up by wetland plants, with less than 2% and 0.2%, respectively, of plant biomass as N and P. Temperature measurements at three depths below FTWs and at the same depths in open water showed no significant difference in mean daily temperatures, suggesting little shading benefit from FTWs. The two ponds produced effluent temperatures that exceeded trout health thresholds. Both the pre- and post-FTW retrofit ponds performed well from a pollutant removal perspective. One pond had extremely low total nitrogen (TN) effluent concentrations (0.41 mg/L and 0.43 mg/L) during both pre- and post-FTW retrofit periods, respectively. Floating treatment wetlands tended to improve pollutant capture within both ponds, but not always significantly. Mean effluent concentrations of TN were reduced at the DOT pond from 1.05 mg/L to 0.61 mg/L from pre- to post-retrofit. Mean total phosphorus (TP) effluent concentrations were reduced at both wet ponds from pre- to post-retrofit [0.17 mg/L to 0.12 mg/L (DOT pond) and 0.11 mg/L to 0.05 mg/L (Museum pond)]. The post-retrofit effluent concentrations were similar to those observed for bioretention cells and constructed stormwater wetlands in North Carolina. The DOT pond showed no significant differences between pre- and post-retrofit effluent concentrations for all nine analytes. The Museum pond had a statistically significant improvement post-retrofit (when compared to the pre-retrofit period) for both TP and total suspended solids (TSS). Wetland plant root length was measured to be approximately 0.75 m, which had the benefit of stilling water flow, thereby increasing sedimentation. Results suggested that greater percent coverage of FTWs produced improved pollutant removal.

Floating treatment wetland aided remediation of nitrogen and phosphorus from simulated stormwater runoff

Type	Journal Article
Author	White, S.A., M.M. Cousins
Publication	Ecological Engineering
Volume	61 (2013)
Pages	207-215
Date	2013

Notes:

A peer-reviewed article of a Beemat study conducted at Clemson University.

Abstract:

Floating treatment wetlands are potential alternatives to traditional constructed wetlands for remediating nutrient-rich water. This study examined the remediation efficacy of floating treatment wetlands planted with Canna flaccida and Juncus effusus in a replicated trough system over two growing seasons at two nutrient loading rates. Plant growth parameters were measured on a biweekly basis, and water quality parameters were monitored weekly. Plant shoots and roots were harvested at the end of the first growing season, and biomass was dried, ground, and analyzed for nutrient content. Juncus plants fixed 28.5±3.4 gNper m2 and 1.69±0.2 gP per m2, while Canna fixed 16.8±2.8 gN per m2 and 1.05±0.2 g P per m2. More N and P were fixed in the below-mat biomass of both species than in the above-mat biomass, thus whole-plant harvest may be a critical management strategy for floating treatment wetlands. During the first season, when nutrient addition rates simulated stormwater loading conditions, effluent nutrient concentrations were very low and averaged 0.14±0.04mgL−1 total N and 0.02±0.01mgL−1 total P. During the second season, nutrient-loading rate into treatment wetlands was doubled to simulate a more nutrient-rich runoff, and effluent nutrient concentrations averaged 0.79±0.3mgL−1 total N and 0.12±0.03mgL−1 total P. Floating treatment wetlands may prove most effective in low nutrient environments where it is necessary to polish water quality to extremely low P concentrations.

Constructed Wetlands with Floating Emergent Macrophytes: An Innovative Stormwater Treatment Technology

Type	Journal Article
Author	Headley, T.R. and C.C. Tanner
Publication	Critical Reviews in Environmental Science and Technology
Volume	42
Pages	2261-2310
Date	2012

Notes:

This peer reviewed article describes various uses for FTWs (stormwater, sewage, acid mine drainage, animal waste, eutrophic lakes, and water supply reservoirs) and focuses on stormwater. Numerous macrophyte species are described. Nutrient removal data for numerous studies and numerous parameters are compiled. This paper provides an excellent summary of research methods and results conducted by major researchers in the field of FTWs.

Abstract:

The treatment of urban stormwater poses numerous technical and operational challenges, particularly due to the intermittent and highly variable nature of hydrologic and pollutant inputs. Floating emergent macrophyte treatment wetlands (FTWs) are a hybridization of ponds and wetlands that offer potential advantages for treatment of these highly variable flows. FTWs utilize rooted, emergent macrophytes growing on a mat or raft floating on the surface of the water rather than rooted in the sediments. Thus, they can tolerate the widely fluctuating water depths typical of stormwater systems, without the risk of the plants drowning. The roots hang beneath the floating mat and provide a large surface area for biofilm attachment. The authors provide a review of the FTW concept, structure, function, and treatment efficiency reported to date and discuss the potential advantages of this emerging technology for stormwater applications. Although still limited, the available data from mesocosm and pilot studies on removal of key pollutants such as organic matter, suspended solids, nutrients, and metals shows that they can significantly enhance performance of pond systems, and provide similar or better performance than surface flow wetlands for a range of polluted waters. Further studies are needed to verify the apparent potential of FTWs treating stormwater at full scale.

http://dx.doi.org/10.1080/10643389.2011.574108

The Ability of Vegetated Floating Islands to Improve Water Quality in Natural and Constructed Wetlands, A Review

Type	Journal Article
Author	Masters, B.
Publication	Water Practice & Technology
Volume	Vol 7 No. 1

Pages n/a
Date 2012

Notes:

This peer-reviewed article provides a detailed overview of the history and various mechanisms of FTWs for water quality improvement, and provides summaries of data obtained by FII in laboratory and field-scale tests. The report also lists suppliers of FTWs worldwide and has a comprehensive reference list of technical articles describing various aspects of FTWs.

Abstract:

Constructed and natural wetlands are widely used to improve many water quality parameters. Vegetated floating islands (VFIs) placed on the surface of these wetlands significantly enhance the efficiency of natural processes that reduce nutrients, suspended solids, heavy metals and other pollutants. Pollutant reduction in VFIs, particularly nutrients such as nitrogen and phosphorous, occurs primarily through the actions of bacterial biofilms growing within the island matrix and on plant roots hanging below the islands. Direct uptake of nutrients by plants is minor, although plants are essential as they provide additional substrate for biofilm development while supplying oxygen and carbon for use by the bacteria.
Nitrogen-based nutrients are primarily removed from wetlands as nitrogen gas. Phosphorous is mostly deposited as organic-rich sediment which accumulates within or beneath the floating islands. This material can become anoxic and return its contained phosphorous to the water column, making it biologically available for algal or bacterial blooms that degrade water quality. Physical removal of this P-rich material is an essential wetland management action. VFIs can remove phosphorous at up to 4.6 g/m2/day and ammonia at up to 8.1 g/m2/day with simultaneous denitrification of nitrate to nitrogen gas. VFIs can significantly increase the efficiency of pollutant removal from natural and constructed wetlands.

Floating Treatment Wetlands for Domestic Wastewater Treatment

Type Journal Article
Author Faulwetter, J.L., M.D. Burr, A.B. Cunningham, F.M. Stewart, A.K. Camper and O.R. Stein
Publication Water Science & Technology
Volume 64.10
Pages 2089-2095
Date 2011

Notes:

A peer-reviewed report that describes the DNA analysis of the bacteria types that were studied in the second Montana-based MBRCT grant. The DNA was identified by denaturing gradient gel electrophoresis (DGGE) by targeting specific functional genes from bacteria samples collected at the top, center, and bottom of island matrix growing on simulated wastewater.

Abstract:

Floating islands are a form of treatment wetland characterized by a mat of synthetic matrix at the water surface into which macrophytes can be planted and through which water passes. We evaluated two matrix materials for treating domestic wastewater, recycled plastic and recycled carpet fibers, for chemical oxygen demand (COD) and nitrogen removal. These materials were compared to pea gravel or open water (control). Experiments were conducted in laboratory scale columns fed with synthetic wastewater containing COD, organic and inorganic nitrogen, and mineral salts. Columns were unplanted, naturally inoculated, and operated in batch mode with continuous recirculation and aeration. COD was efficiently removed in all systems examined (>90% removal). Ammonia was efficiently removed by nitrification. Removal of total dissolved N was ~50% by day 28, by which time most remaining nitrogen was present as NO3-N. Complete removal of NO3-N by denitrification was accomplished by dosing columns with molasses. Microbial communities of interest were visualized with denaturing gradient gel electrophoresis (DGGE) by targeting specific functional genes. Shifts in the denitrifying community were observed post-molasses addition, when nitrate levels decreased. The conditioning time for reliable nitrification was determined to *be approximately three months. These results suggest that floating treatment wetlands are a viable alternative for domestic wastewater treatment.*

The Taxonomy of Treatment Wetlands: A Proposed Classification and Nomenclature System

Type	Journal Article
Author	Headley, T.R. and N. Fonder
Publication	Journal of Environmental Engineering
Volume	(in press)
Pages	n/a
Date	2011

Notes:

This peer-reviewed paper provides good descriptions and pictures of the various types of treatment wetlands.

Abstract :

This paper proposes a structured foundation for classifying and naming different treatment wetland (TW) design alternatives, based on observable physical design traits. A classification hierarchy is organised like a polychotomous key, from general classification criteria to wetland type identification. Three characteristics are typical of all TW: the presence of macrophytic vegetation; the existence of water-logged or saturated substrate conditions for at least part of the time; and inflow of contaminated water with constituents to be removed. Treatment Wetlands are further classified based on hydrology and vegetation characteristics. Hydrological traits relate to water position, flow direction, degree of saturation and position of influent loading. Based on the predominant position of water in the system, two main groups are identified: those with Surface Flow above a benthic substrate and those with Subsurface Flow through a porous media. The systems with surface flow are divided into three standard types, differentiated by vegetation type: Surface Flow (SF), Free-Floating Macrophyte (FFM), and Floating Emergent Macrophyte (FEM) TWs. Subsurface flow systems always contain sessile emergent macrophytes and are divided into four standard types, based on flow direction: Horizontal Sub-Surface Flow (HSSF), Vertical Flow (VF), Up Flow (UF) and Fill and Drain (FaD) TWs. Standard types are described with their main applications. Associated variants are identified. An overview of intensified variants, which have elevated energy, chemical or operational inputs in order to increase efficiency or overcome process limitations, is also provided.

Components of floating emergent macrophyte wetlands influencing removal of stormwater pollutants

Type	Journal Article
Author	Tanner, C.C. and T.R. Headley
Publication	Ecological Engineering
Volume	37
Issue	**3**
Pages	474-486
Date	2011

Notes:

This peer-reviewed article describes the findings of tank-scale controlled experiments comparing FTWs planted with and without macrophytes for the removal of metals, phosphorus, and turbidity. The study examines the contribution of each of the major constituents of the FTWs.

ABSTRACT

Floating treatment wetlands planted with emergent macrophytes (FTWs) provide an innovative option for treating urban stormwaters. Emergent plants grow on a mat floating on the water surface, rather than rooted in the bottom sediments. They are therefore able to tolerate the wide fluctuations in water depths that are typical of stormwater ponds. To better understand the treatment capabilities of FTWs,
a series of replicated (n = 3) mesocosm experiments (12×0.7m3 tanks using 0.36m2 floating mats) were conducted over seven day periods to examine the influence of constituent components of FTWs (floating mat, soil media, and four different emergent macrophyte species) for removal of copper, zinc, phosphorus and fine suspended solids (FSS) from synthetic stormwater. The presence of a planted floating mat significantly (P < 0.05) improved removal of copper (>6-fold), fine suspended particles (#3-fold reduction in turbidity) and dissolved reactive P (in the presence of FSS) compared to the control. Living

plants provided a large submerged root surface-area (4.6–9.3m2 of primary roots m−2 mat) for biofilm development and played a key role in the removal of Cu, P and FSS. Uptake of Cu and P into plant tissues
during the trials could only account for a small fraction of the additional removal found in the planted FTWs, and non-planted floating mats with artificial roots providing similar surface area generally did not provide equivalent benefits. These responses suggest that release of bioactive compounds from the plant roots, or changes in physico-chemical conditions in the water column and/or soils in
the planted FTWs indirectly enhanced removal processes by modifying metal speciation (e.g. stimulating complexation or flocculation of dissolved fractions) and/or the sorption characteristics of biofilms.
The removal of dissolved zinc was enhanced by the inclusion of a floating mat containing organic soil media, with reduced removal when vegetated with all except one of the test species. The results indicate
that planted FTWs are capable of achieving dissolved Cu and Zn mass removal rates in the order of 5.6–7.7mgm−2 d−1 and 25–104mgm−2 d−1, respectively, which compare favourably to removal rates
reported for conventional surface flow constructed wetlands treating urban stormwaters. Although not directly measured in the present study, the removal of particulate-bound metals is also likely to be
high given that the FTWs removed approximately 34–42% of the turbidity associated with very fine suspended particulates within three days. This study illustrates the promise of FTWs for stormwater treatment, and supports the need for larger-scale, longer-term studies to evaluate their sustainable treatment performance.

The Effects of Artificial Floating Wetlands on Water Quality in a Eutrophic Lake

Type	Master's Thesis
Author	Jangrell-Bratli, A. S.
Publication	University of Florida St. Petersburg
Volume	n/a
Pages	n/a
Date	July 2011

Notes:

This 160-page thesis describes the results of a pond-scale study using FTWs comprised of plastic flotation pipe and netting, with macrophytes. This study is related to the 2011 study by J.A. Vogel.

Effects of Vegetation, Season and Temperature on the Removal of Pollutants in Experimental Floating Treatment Wetlands

Type	Journal Article
Author	Van de Moortel, A.M.K., E. Meers, N. De Pauw, F.M.G. Tack
Publication	Water Air Soil Pollution
Volume	212
Pages	281-297
Date	Feb 2010

Notes:

This peer-reviewed article describes a comparative study of FTWs with and without plants.

Abstract:

The research and interest towards the use of constructed floating wetlands for (waste)water treatment is emerging as more treatment opportunities are
marked out, and the technique is applied more often. To evaluate the effect of a floating macrophyte mat and the influence of temperature and season on
physico-chemical changes and removal, two constructed floating wetlands (CFWs), including a floating macrophyte mat, and a control, without emergent vegetation, were built. Raw domestic wastewater from a wastewater treatment plant was added on day 0. Removal of total nitrogen, NH4–N, NO3–N, P, chemical oxygen demand (COD), total organic carbon
and heavy metals (Cu, Fe, Mn, Ni, Pb and Zn) was studied during 17 batch-fed testing periods with a retention time of 11 days (February–March 2007 and
August 2007–September 2008). In general, the CFWs performed better than the control. Average removal efficiencies for NH4–N, total nitrogen, P and COD were respectively 35%, 42%, 22% and 53% for the CFWs, and 3%, 15%, 6% and 33% for the control. The pH was significantly lower in the CFWs (7.08± 0.21) than in the control (7.48±0.26) after 11 days. The removal efficiencies of NH4–N, total

nitrogen and COD were significantly higher in the CFWs as the presence of the floating macrophyte mat influenced positively their removal. Total nitrogen, NH4– N and P removal was significantly influenced by temperature with the highest removal between 5°C and 15°C. At lower and higher temperatures, removal relapsed. In general, temperature seemed to be the steering factor rather than season. The presence of the floating macrophyte mat restrained the increase of the water temperature when air temperature was > 15°C. Although the mat hampered oxygen diffusion from the air towards the water column, the redox potential measured in the rootmat was higher than the value obtained in the control at the same depth, indicating that the release of oxygen from the roots could stimulate oxygen consuming reactions within the root mat, and root oxygen release was higher than oxygen diffusion from the air.

Floating Treatment Wetlands: An Innovative Option for Stormwater Quality Applications

Type Conference Article
Author Headley, T.R. and C.C. Tanner
Publication 11th International Conference on Wetland Systems for Water Pollution Control (India)
Volume Nov 1-7
Pages 1101-1106
Date 2008

Notes:

This peer-reviewed article is a short version of the 2012 report by Headley and Tanner (listed below). It describes various types of FTWs with macrophytes.

ABSTRACT
Floating Treatment Wetlands (FTWs) are an innovative variant of the more traditional constructed wetland and pond technologies that offer great potential for treatment of urban stormwaters. FTWs employ rooted, emergent macrophytes (similar to those used in surface and subsurface flow wetlands) growing on a mat floating on the surface of the water rather than rooted in the sediments. Thus, they can tolerate the wide water depth fluctuations typical in stormwater systems, without the risk of the plants becoming inundated and stressed. In many aspects, FTWs are a hybrid between a pond and a wetland; they behave hydraulically similar to a stormwater detention pond, whilst imparting similar treatment processes to that of a wetland. The plant roots hang beneath the floating mat and provide a large surface area for biofilm growth which forms an important part of the treatment reactor. This paper provides a review of the FTW concept, structure and function, and discusses some of the potential advantages of this emerging technology for stormwater applications.

Purification of nitrate-rich agricultural runoff by a hydroponic system

Type Journal Article
Author Yang, Z., S. Zheng, J. Chen, M. Sun
Publication Bioresource Technology
Volume 99 (2008)
Pages 8049 - 8053
Date April 2008

Notes:

Nutrient removal in runoff water was studied . Young O. javanica seedlings were evenly transplanted to 4-cm thick foam sheets that fully covered the overall water surface in the floating rafts.

Abstract:

The purification of nitrate-rich agricultural runoff by a floating-raft (FR) hydroponic system was investigated at 3-, 2- and 1-d hydraulic retention times (HRTs) with particular emphasis on nitrogen conversion and removal through the system. The FR system has a dissolved oxygen (DO) environment similar to the horizontal subsurface flow system, generally 0.00 mg L_1 that facilitates denitrification. An efficient nitrate–nitrite–nitrogen (NOx–N) removal, 91%, 97% and 71% on average at 3-, 2- and 1-d HRT, respectively, was frequently achieved. The mean retentions were 17–47% for chemical oxygen demand, 31–64% for total nitrogen, and 8–15% for total phosphorus for the FR system. Mass balance analysis implied that the detectable DO concentration in the reactor, as low as 0.7 mg L_1, played a very important role in the conversion and removal of NH3–N and NOx–N, which finally affected the NOx–N removal at 3-d HRT.

Floating Islands as an Alternative to Constructed Wetlands for Treatment of Excess Nutrients from Agricultural and Municipal Wastes -- Results of Laboratory-Scale Tests

Type	Journal Article
Author	Stewart, F.M., T. Mulholland, A. Cunningham, B. Kania, M. Osterlund
Publication	Land Contamination and Reclamation
Volume	16
Number	1
Pages	25-33
Date	February 2008

Notes:

This peer-reviewed article describes several of the tank-scale FTW experiments that measured bacterial biofilm removal of nitrate, ammonium and phosphate, and compares these removal rates to those achieved by other researchers, who were primarily focused on plant-based removal of nutrients on FTWs.

http://www.floatingislandinternational.com/wp-content/plugins/fii/research/9.pdf

Abstract

Constructed wetlands are recognized as effective mechanisms of water treatment and are employed in a variety of applications. Wetlands comprise diverse and complex systems of interacting plants and animals that remove contaminants from the water column by mechanical filtration and biochemical conversion. A major component of the wetland environment is microbial, with bacteria and other microorganisms proliferating upon all available submerged surfaces (i.e. substrate). In these wetland environments, microbial activity is limited by substrate surface area and nutrient flux. Consequently, the microbial contribution to wetland efficacy can be improved by increasing a wetland's substrate surface area and increasing water circulation rates through that substrate. Various studies have investigated the use of floating wetland platforms to enhance wetland capacity; however, none of those studies has determined the specific contributions of microbes. In our study, we quantified the microbial component of BioHaven® Floating Islands for aerobic removal of ammonium, anoxic removal of nitrate, and simultaneous aerobic/anoxic removal of ammonium, nitrate and phosphate. This study establishes tank-scale standards to which other microbial data can be compared. In doing this, it has been determined that the microbes growing within a unit volume of BioHaven® Floating Island material are capable of removing 10 600 mg of nitrate per day, 273 mg of ammonium per day, and 428 mg of phosphate per day, where the unit island volume is defined as having a top surface area of 1.0 ft^2 and a thickness of 0.6 ft.

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(2) Articles Published in Professional and Trade Magazines

Floating Treatment Wetlands Improve Stormwater Quality

Type	Professional Magazine Article
Author	Reinsel, Mark
Publication	Environmental Science and Engineering
Volume	Volume 26, No. 3 May/June 2013
Pages	40-44
Date	2013

Notes:

This article provides an overview of case studies in North Carolina and Montana, and provides removal rate and concentration data for eight common pollutants associated with stormwater.

http://ese.dgtlpub.com/2013/2013-06-30/home.php

Floating Wetlands Help Boost Nitrogen Removal in Lagoons

Type	Trade Magazine Article
Author	Reinsel, M.
Publication	WaterWorld
Contract	June
Pages	n/a
Date	2012

Notes:

This article gives a short summary of removal of ammonia, nitrate, and some other nutrients at Rehberg Ranch (MT) , Wiconisco (PA) , McLean's Pit (NZ) and the MBRCT test ponds (MT).

http://www.waterworld.com/articles/print/volume-28/issue-6/editorial-features/floating-wetlands-help-boost--nitrogen-removal-in-lagoons.html

Wesstown Lake: Floating Wetland Islands

Type	Trade Magazine Article
Author	Lubnow, F.
Publication	Lakeline
Volume	Spring
Pages	31-35
Date	2012

Notes:

This five-page article describes the phosphorus uptake mechanisms in FTWs (plants, microbes, filtration of particles, moving up the food chain). In a first pond experiment with FTWs, uptake of P and N were estimated by analyzing plant mass and nutrient concentration in the plants. In a second pond experiment (Mermaid Pool, New Jersey) P uptake was measured for a range of inflow values before and after FTW installation. Graphical results are presented for P uptake at the inlet and outlet of Mermaid Pool during 2011.

Floating Islands for Tertiary Nutrient Removal and Circulators for Primary & Secondary Treatment at an STP

Type	Trade Magazine Article
Author	Ambulkar, A., S. Zeller and D. Klinger
Publication	Everything About Water
Volume	May
Pages	85-87
Date	2012

Notes:

Photographs and descriptions of BioHaven FTWs at the Wiconisco site, with additional information related to Solar Bee circulators also deployed at the site.

Floating Treatment Wetlands Mitigate Lake Eutrophication

Type	Professional Magazine Article
Author	Reinsel, M.
Publication	Environmental Science & Engineering Magazine
Volume	May/June
Pages	38-41
Date	2012

Notes:

This article describes how D.O. and temperatures were improved for fish habitat at Fish Fry Lake, Shepherd, Montana, using a Leviathan™ system.

Using Floating Islands for Tertiary Nutrient Removal

Type	Professional Magazine Article
Author	Ambulkar, A., S. Zeller and D. Klinger
Publication	Environmental Science and Engineering Magazine
Volume	Summer

Pages 24-27
Date 2010

Notes:

A summary of the first pilot-scale deployment of BioHaven FTWs in a controlled test of wastewater treatment in a municipal lagoon setting at Wiconisco, PA. Measured parameters described in the article include BOD, TN, and TP.

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(3) Articles Published in Popular Magazines, Books or On-Line

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Innovative Alternative for Waste Water Impoundment Treatment

Type On-Line Magazine Article
Author Waguespack, Nicole (Martin Ecosystems)
Publication Land and Water
Volume May/June 2013
Pages 41 - 44
Date 2013

Notes:

This on-line article provides a description of the floating island project for an unaerated wastewater lagoon the Elayn Hunt Correctional Facility in Louisiana. Photographs of the islands at various stages of plant growth are included. Descriptions of island placement, maintenance and repairs, planted species, and numerical results are provided.

Fishing out Phosphorus

Type On-Line Magazine Article
Author Fox, Andrea
Publication WEF News
Volume Jan 21
Pages 1-6
Date 2013

Notes:

A description of the "phosphorus-to-fish" process that is being managed by Bruce Kania at Fish Fry Lake in Shepherd, MT.

http://news.wef.org/fishing-out-phosphorus

Floating Treatment Wetlands Mitigate Lake Eutrophication

Type On-Line Magazine Article
Author Kania, B, M. Reinsel, F. Stewart
Publication Water Online
Volume August 14
Pages 1-7
Date 2012

Notes:

This article provides an overview of the stewardship at Fish Fry Lake in Shepherd MT, for the improvement of water quality and fishing. Descriptions of the Leviathan® system and plots of DO and temperature versus depth (before and after stewardship) are presented. A photo of Bruce Kania with a string of large perch is included.

Floating Islands – an alternative to urban wetlands

Type Chapter in a book
Author Hwang, L. and A. LePage
Publication Wetlands – Integrating Multidisciplinary Concepts, (LePage, B.A. ed. Springer Netherlands)
Volume n/a
Pages n/a
Date 2011

Floating Treatment Wetlands: An Innovative Option for Stormwater Quality Applications

Type Conference Article
Author Headley, T.R. and C.C. Tanner
Publication 11th International Conference on Wetland Systems for Water Pollution Control (India)
Volume Nov 1-7
Pages 1101-1106
Date 2008

Notes:

This peer-reviewed article is a short version of the 2012 report by Headley and Tanner (listed below). It describes various types of FTWs with macrophytes.

ABSTRACT
Floating Treatment Wetlands (FTWs) are an innovative variant of the more traditional constructed wetland and pond technologies that offer great potential for treatment of urban stormwaters. FTWs employ rooted, emergent macrophytes (similar to those used in surface and subsurface flow wetlands) growing on a mat floating on the surface of the water rather than rooted in the sediments. Thus, they can tolerate the wide water depth fluctuations typical in stormwater systems, without the risk of the plants becoming inundated and stressed. In many aspects, FTWs are a hybrid between a pond and a wetland; they behave hydraulically similar to a stormwater detention pond, whilst imparting similar treatment processes to that of a wetland. The plant roots hang
beneath the floating mat and provide a large surface area for biofilm growth which forms an important part of the treatment reactor. This paper provides a review of the FTW concept, structure and function, and discusses some of the potential advantages of this emerging technology for stormwater applications.

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(4) Reports

Floating Treatment Wetland Technologies

Type Unpublished Report for Distribution
Author Reinsel, Mark
Publication Floating Island International, Inc
Pages 1-25
Date November 2012

Notes:

This report describes the various FII treatment alternatives (BioHavens, Leviathan, Coral, and BioSwales) and discusses design, installation, and costs.

BioHaven technology: Where Human Endeavor and Nature Come Together

Type Unpublished Report for Distribution
Author FII Staff
Publication Floating Island International, Inc
Pages 1-4
Date September 2012

Notes:

This article was prepared for the U.S. Navy. It provides an overview of FII, as well as a review of various applications for FTW technology. The article contains several photographs of FTWs in different settings around the world, including a tern nesting island, a Leviathan® stream channel, a framework for a circular FTW under construction, and a 21,000 sf archipelago in Singapore. The article also provides a future vision statement for the FTW products.

Floating Wetland Systems for Nutrient Removal in Stormwater Ponds – FDOT Project BDK78 985-01

Type University Project with Florida DOT Supervision – Final Report
Project Manager Rick Renna, P.E., State Hydraulics Engineer
Author Wanielista, M.P.; NB Chang, M. Chopra, Z. Xuan, K. Islam, Z. Marimon
Publication n/a

Date September 2012

Notes:

This 182-page final report describes tank-scale and pond-scale experiments for BioHaven islands and Beemat products. The pond-scale experiments were affected by nutrient uptake by duckweed and algae in the controls. The study recommends giving a 12% credit for deploying FTWs in stormwater ponds. The report incorporates numerous photographs and numerical data.

Waterway Stewardship through Floating Islands

Type Unpublished Report for Distribution
Author Reinsel, Mark, F. Stewart
Publication Floating Island International, Inc

Pages 1-10
Date June 2012

Notes:

This 10-page report provides an overview of floating treatment wetlands (FTWs). The article covers biofilm basics, nutrient removal mechanisms, a comparison of nutrient removal efficacy by FTWs compared to other stormwater BMP methods, and fishery enhancement by FTWs. A bullet list summary of important facts from Azim's periphyton ecology textbook is provided. A short discussion of the Fish Fry Lake case study is also included.

Final report: Evaluation of Floating Wetland Islands (FWIs) as a Retrofit to Existing Stormwater Detention Basins

Type Final Report
Author Hunt, W.F; R.J. Winston, S.G. Kennedy
Publication North Carolina DENR
Contract 1653
Pages 1-71
Date March 22, 2012

Notes:

This report describes research conducted on real-world stormwater ponds under controlled conditions by the University of North Carolina under a grant funded by the NC DENR. Measured parameters included TN, TP, and TSS. The data were statistically analyzed.

Control of Microbial Processes for Enhanced Water Treatment Using Floating Island Treatment Systems

Type Grant Final Report
Author Cunningham, A.B., A. Camper, M. Burr and F.M. Stewart
Publication Montana Board of Research and Commercialization Technology
Date 2010

Notes:

The final report for the second Montana-based MBRCT grant. The report includes data from controlled laboratory-scale studies at the Center for Biofilm Engineering at Montana State University. The lab experiments compared organic carbon, ammonium and nitrate removal in simulated wastewater for various matrix types including PET and recycled carpet fibers. The experiments also compared various aeration cycling regimes to optimize for combined aerobic and anoxic bacterial removal. DNA analysis was performed on the bacterial biofilms to determine how the nitrifiers and denitrifiers were distributed within the matrix columns.

The project also included field-scale components, which comprised a comparison of wind-electric and solar-electric-power for circulating and aerating outdoor islands. In addition, a 1300-sf island was installed in a wastewater lagoon in Billings, MT, and a controlled experiment was started that tracked removal of TN, TP, nitrate, ammonium, phosphate, and COD in an island lagoon and a control lagoon. Removal data are presented in the report.

This report is available on the FII website.

Floating Vegetated Islands for Stormwater Treatment: Removal of Copper, Zinc and Fine Particulates

Type	Report
Author	Headley, T.R. and C.C. Tanner
Publication	NIWA Client Report HAM2007-175
Source	Auckland (NZ) Regional Council
Pages	1-28
Date	Nov 2007

Notes:

The first part of this report is a study of suitable NZ native plant species suitable for use on FTWs. The second part of this report describes experiments that measured removal rates of copper, zinc, and fine particulates in 1 m^3 test tanks run in triplicate. Numerical data of removal rates are presented in the report. The report provides a good description of a well-run experiment and contains photographs of the tanks, FTWs, plant tops and roots.

Application of Floating Wetlands for Enhanced Stormwater Treatment: A Review

Type	Report
Author	Headley, T.R. and C.C. Tanner
Publication	NIWA Client Report HAM2006-123
Source	Auckland (NZ) Regional Council
Pages	1-100
Date	Nov 2006

Notes:

This report provides descriptions and photographs of a wide range of FTWs that were commercially available in 2006, including the FII products. It does not include experimental results that were obtained after 2006 (see the 2011 and 2012 papers for more recent data).

Wittenberg, Joyce

From: Suplee, Mike
Sent: Monday, April 07, 2014 5:00 PM
To: Greeley, Carrie
Subject: FW: A few things....

From: Brian Sugden [mailto:Brian.Sugden@plumcreek.com]
Sent: Monday, April 07, 2014 2:01 PM
To: Suplee, Mike
Subject: A few things....

Mike,
At the NW Science meeting last week in Missoula, I ran into a limnology student from Western Washington University, who is doing some work on Lake Whatcom, up near Bellingham, WA. He sent me a link to their website, and thought you might be interested. I like the way they had their annual reports and data organized…

http://www.wwu.edu/iws/

From web link above, click on "Lake Whatcom" tab on the left and navigate from there.

I can't help but note that this academic institution doesn't have a problem sharing data…

By the way, nice job in the public hearings presenting the "tsunami of science" supporting the rules.

Our family just got back from a week in Utah – backpacking in Canyonlands NP and some mountain biking. Very nice time. In my mad scramble to get out of town, I didn't manage to get any written comments in on the proposed rules. Not sure that I would have had anything more to say that I did in the oral testimony. EXCEPT… that issue I raised earlier in an email to you regarding the endnote to Table 12A-1 (see attached email). Is this a problem that I forgot to include this in comments?

-Brian

Brian Sugden
Forest Hydrologist, Plum Creek
PO Box 1990
Columbia Falls, MT 59912
406-892-6368
Brian.Sugden@plumcreek.com

From:	Suplee, Mike
To:	Brian Sugden
Subject:	RE: Question on Circular 12, Table 12A-1
Date:	Thursday, January 30, 2014 2:18:55 PM
Attachments:	image001.jpg

Hi Brian;

At this stage it is best that you provide a comment during the upcoming BER public comment period, then I can address it in the final.

Thanks, Mike

From: Brian Sugden [mailto:Brian.Sugden@plumcreek.com]
Sent: Thursday, January 30, 2014 2:16 PM
To: Suplee, Mike
Subject: RE: Question on Circular 12, Table 12A-1

Yes, I think a clarification is needed. Please let me know what you would suggest as clarification on this Endnote… and if the Department is going to work that change in, or if some of us should comment on that.
Thanks,
-Brian

From: Suplee, Mike [mailto:msuplee@mt.gov]
Sent: Thursday, January 30, 2014 2:13 PM
To: Brian Sugden
Subject: RE: Question on Circular 12, Table 12A-1

Hi Brian;

Hope all is well.

You are correct that endnote 4 applies to MPDES permitting, not surface water assessment methodology; the same procedures we have been using since 2010 for assessment will continue even after adoption of the nutrient standards.

Perhaps a clarifying sentence is needed.

It is not a 30 day rolling, its monthly.

-Mike

From: Brian Sugden [mailto:Brian.Sugden@plumcreek.com]
Sent: Thursday, January 30, 2014 11:17 AM
To: Suplee, Mike

Cc: Douglas Parker (DParker@hydrometrics.com); Mark Lambrecht (tsria@mt.net)
Subject: Question on Circular 12, Table 12A-1

Mike – On Table 12A-1 for the criteria in in Circular 12 there is an Endnote 4 referenced that states "***The 30-day average concentration of these parameters may not be exceeded more than once in any five-year period, on average***."

A couple of questions:

1. I assume this applies to permit effluent limitations and not assessment method for listing/delisting. If so, this should be clarified in this endnote.
2. Also with regard to this endnote, is this based on a rolling 30-day average or monthly values?

Thanks,
-Brian

Brian Sugden
Forest Hydrologist, Plum Creek
PO Box 1990
Columbia Falls, MT 59912
406-892-6368
Brian.Sugden@plumcreek.com

Wittenberg, Joyce

From: Bruce Kania <bruce@floatingislandinternational.com>
Sent: Tuesday, April 01, 2014 12:39 PM
To: DEQ WQP Admin
Subject: Proposed Variance Rules
Attachments: Comments to DEQ--Bruce.docx; Dodkins report swansea.pdf

Dear Carrie,

Attached are some brief comments that describe an alternative technology poised to facilitate successful compliance with DEQ's proposed Variances rules, and a supporting document.

Best Regards,
Bruce Kania
CEO Floating Island Int.
1-800-450-1088

Ms. Carrie Greeley
Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901
deqwqpadmin@mt.gov

1 April 2014

To Whom It May Concern,

As CEO of Floating Island International, a Montana company actively engaged in the international water quality market, I commend Montana's DEQ for implementation of new nutrient standards for Montana. We are also very pleased to note DEQ's solutions-oriented policy, as exemplified by nutrient credit trading language in the standards.

A summary report written by Dodkins, attached below, tracks an extensive series of peer-reviewed papers that quantify our modular technology's internationally proven efficacy around nitrogen cycling. As such, we welcome the new parameters as the right stimulus to garner integration of floating treatment wetlands as a viable and cost-effective alternative technology across Montana.

We would also like to note that science is building around both cold weather efficacy and strategies for phosphorus uptake in association with our embodiments of floating treatment wetlands. On another environmentally positive tact, our islands are also being launched in both stormwater and recreational pond settings, and so are beginning to demonstrate a realistic ability to address watershed TSS and nonpoint nutrient loads. Our technology, born in Montana and the development of which was co-funded by a Montana public research entity, promises to enhance DEQ's commendable efforts towards environmental sustainability in our great state!

Sincerely,
Bruce Kania
CEO Floating Island Int.

SEACAMS Swansea University

Enterprise Assist:

Floating Treatment Wetlands (FTWs) in Wastewater Treatment: Treatment efficiency and potential benefits of activated carbon.

I Dodkins & AF Mendzil

March 2014

Prepared for: FROG Environmental Ltd, Ban y Berllan, Llansadwrn, Llanwrda, SA19 8NA.

Sustainable Expansion of the Applied Coastal
And Marine Sectors (SEACAMS)
Prifysgol Abertawe/Swansea University
Abertawe/Swansea
Cymru/Wales
SA2 8PP
www.seacams.ac.uk
Ebost/Email: I.R.Dodkins@swansea.ac.uk

ERDF
Llywodraeth Cynulliad Cymru
Welsh Assembly Government
Ewrop & Chymru: Buddsoddi yn eich Dyfodol
Cronfa Datblygu Rhanbarthol Ewrop
Europe & Wales: Investing in your Future
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Literature Review:

Floating Treatment Wetlands (FTWs) in Wastewater Treatment: Treatment efficiency and potential benefits of activated carbon.

Dr. Ian Dodkins; Anouska Mendzil; Leela O'Dea

Executive Summary

Floating Treatment Wetlands (FTWs) have many benefits over Free Water Surface (FWS) wetlands:

1. Plant roots assisting in filtering and settling processes for sediment bound P and metals
2. Plant roots acting as a large surface area for micro-organism activity in: decomposition, nitrification, and denitrification (removal of BOD and N).
3. Mild acidification of water due to release of humic acids; and a C input from senescent vegetation, assisting denitrification.
4. They can adjust to varying water levels
5. A higher retention time is possible as they can be made deeper without submerging the vegetation

Percentage removal of nutrients and metals from effluent is around 20-40% higher in FTWs than in conventional FWS ponds. Removal efficiency, particularly of nitrogen, can be further increased with tighter control on the water chemistry (aeration; adding $CaCO_3$; adding a carbon source). 20% coverage of islands is optimal for aerobic basins. 100% cover is optimal for anaerobic basins or aerobic basins where there is artificial aeration. The design the FTW and the control of basin water chemistry is essential for optimising treatment efficiencies. The passive use of activated carbon within layers of floating islands is unlikely to be cost effective.

Introduction

Definition

Floating Treatment Wetlands (FTWs) comprise of wetland basins or cells, on which there are artificial mats containing emergent plants (Figure 1). This is not to be confused with treatment using floating leaved plants such as *Eichhornia crassipes* (Water Hyacinth), *Pistia stratiotes* (water lettuce), *Lemna* spp. (duck weed) or *Azolla* spp. (water fern) e.g. Reddy & Smith (1987); Kivaisi (2001), or where natural floating islands have established. Floating Treatment Wetlands are also referred to as Constructed Floating Wetlands (CFWs) or Floating Mat Constructed Wetlands, but we will use FTW throughout the review. Floating Islands (FIs) will be used to refer only to the islands within the treatment system. 'Effluent' refers to the water being treated at any stage within the wetland and 'inflow' refers to effluent entering the wetland, and 'outflow' as effluent leaving the wetland. Comparison will regularly be made between FTWs and other wetlands. Where 'conventional wetlands' is referred to, this means other treatment wetlands in general. Basins where there is open water but no islands, are known as Free Water Surface (FWS) wetlands.

The core of this review assesses process, performance and design of FTWs and includes a section on the potential for incorporating activated carbon into FTWs.

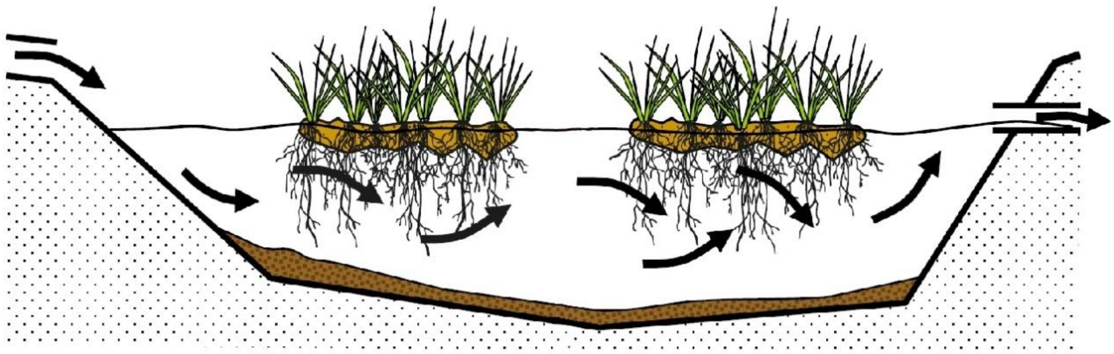

Figure 1. A Floating Treatment Wetland (FTW). Emergent plants are grow within a floating artificially constructed material. The roots are directly in contact with the effluent and can intercept suspended particles. The roots also provide a high surface area for microbiological activity. Image: Headley and Tanner (2006).

1.1 History

Floating islands are a natural occurrence, and can be found where emergent aquatic plants have broken from the land, sometimes developing in highly nutrient rich or sulphurous pools (Duzer, 2004). Floating leaved plants for treatment date back to the 11th Century, when the floating *Azolla* fern was used by Chinese and Vietnamese farmers to extract dissolved nutrients from wetlands and rice paddies, after which it was dried and applied as soil fertiliser (Whitton & Potts, 2002). The use of Water Hyacinth (*Eichhornia crassipes*) to remove nutrients also developed in South East Asia, and both have been used for centuries for water treatment within this region (Whitton & Potts, 2002). *E. crassipes* was suggested for use in the early 20th Century in both Auckland (Australia) and Yorkshire (UK) (Dymond, no date), and then in 1975 NASA used it to treat a sewage lagoon in the USA (Wolverton & Mcdonald, 1978).

Constructed floating islands were first developed in Japan in the 1990s, with *Cana generalis* being grown in floating beds to absorb nutrients from fish ponds and treatment basins (Wu et al., 2000). Twenty percent coverage of soilless artificial floating islands, again using *C. generalis*, was later recommended to improve water quality in China (Bing & Chen, 2001).

Floating mats have also developed unintentionally in many open water treatment systems. Sometimes detrimental effects were observed, such as in Florida, where mats which had grown to 50% coverage moved with the wind across a shallow basin; scraping the bottom and disturbing sediments, resulting in increased outflow turbidity and phosphorus release (Kadlec & Wallace, 2009).

1.2 Range of applications

FTWs can be specifically designed or they can be installed in currently operating open water wetlands i.e. retrofitting. Potentially FTWs can be used with the same waste streams as conventional wetland systems. Some examples from the literature are:

Domestic wastewater treatment

Conventional vegetated wetlands have often been advocated for wastewater polishing, rather than heavy nutrient loads, since they can become clogged and plants are very good at removing low concentrations of nutrients. However, given that there is primary sedimentation, FTWs can potentially deal with larger nutrient loadings since they have higher P and N removal capacity compared to conventional wetland treatment systems. The exposed roots aid sediment deposition, thus reducing turbidity, and there is greater surface area available for microbiologically mediated nitrification/denitrification reactions. Treatment with floating islands has been done on domestic waste in a highly controlled environment i.e. as a hydroponic system (Vaillant et al., 2003).

Metals treatment

Wet detention ponds are used as a Best Management Practice for stormwater run-off in the USA (Chang et al., 2013). FTWs have thus become a popular choice as a retrofit for stormwater run-off treatment in these ponds (Chimney et al., 2006; Headley & Tanner, 2006; Tanner & Headley, 2008, 2011; Hwang & Lepage, 2011; Chang et al., 2012; Borne et al., 2013; White & Cousins, 2013; Winston et al., 2013). FIs are beneficial since they can treat water effectively even with the large fluctuations in water depth that occur during storms.

Strosnider & Nairn (2010) stated that FTWs are ideal for acid mine drainage, particularly if anaerobic conditions are maintained using high island cover. The resulting anaerobic conditions and the decomposing plant material aids denitrification, making the water more alkaline.

Agricultural waste

The enhanced nitrate removal rate of FTWs makes them appealing in reducing pollution from agricultural run-off (Stewart et al., 2008; Yang et al., 2008) as well as for more concentrated wastewaters, such as from swine effluent (Hubbard et al., 2004).

Habitats

FIs have sometimes been constructed specifically to create habitats e.g. to protect birds from land-based predators (Hancock, 2000), including a huge floating island of 3 700 m^2 in Sheepy lake (California) as a habitat for nesting Caspian Terns (Patterson, 2012). Only islands designed for effluent treatment will be covered in this review, however FIs do provide habitats as a secondary function. Emergent grasses can attract waterfowl and terrestrial birds because of the seeds, nesting material, nesting cover and available water. Fish have been introduced into some open water wetlands, however those that feed or nest on the bottom have been found to disturb sediments, increasing suspended solids (Kadlec & Wallace, 2009, p.779).

Although usually not problematic, there have been incidences where large bird communities have contaminated open water treatment wetlands with faeces (Orosz-Coghlan et al., 2006), or have disturbed sediments, increasing turbidity (Knowlton et al., 2002). Geese herbivory can devastate the establishment of wetland plants, especially if planted during the spring or autumn migratory period (Kadlec & Wallace, 2009). However, a benefit of floating islands is that other herbivores e.g. rabbits, cannot usually access the islands. Mosquitoes may also be a problem with an open water system, particularly where monotypic vegetation such as cattail, bulrush and common reed restrict predator access (Knight et al., 2004). However, removing leaf litter, and ensuring that water depth is greater than 40cm (Sinclair et al., 2000) can reduce the problem.

Tourism
Treatment wetlands have been effectively marketed for tourism, especially those which provide good natural habitats for birds (Kadlec & Wallace, 2009). If the FTW is operated for tourism the design and operation is likely to have to include walkways, bird viewing areas and education centres. There may also be conflicting aims for depth regulation between habitat provision and treatment.

2. Processes

FTWs, as with other wetland treatment systems, remove pollutants by four main processes (in order of importance): physical; biogeochemical; microbial and plants. These processes are similar in conventional wetlands, so much of the details provided here comes from that research. However, the larger surface area created by plant roots in FTWs tends to increase sedimentation (by filtering), microbiological decomposition, nitrification and denitrification, and also alter the water chemistry i.e. pH and dissolved oxygen (DO) concentrations. Processes will be discussed relative to the effluent constituents being removed.

2.1 Phosphorous removal

Phosphorous within wetland effluents is usually as dissolved orthophosphate (PO_4^{3-}), or organic phosphorus (Masters, 2012). The scarcity of P in natural environments results in efficient nutrient cycling within ecological systems (Kadlec & Wallace, 2009), thus there are few permanent routes for removal of P within treatment wetlands (Figure 1). The major mechanisms for P removal are accretion in peat/soil and soil adsorption.

Settling and peat accretion
Settling is the main process by which phosphorous bound sediments and BOD are removed from the water column (Kadlec & Wallace, 2009). Settling is a physical process whereby phosphate bound in particles sink to the bottom. Settling is increased in FTWs both by the roots (Masters, 2012) which filter the particles from the water column to later slough off to settle on the bottom, and by reducing currents and circulation caused by surface wind disturbance or water movements (e.g. from pumps) (Headley & Tanner, 2006; Chang et al., 2013). The reduction in movement is essential for preventing resuspension of sediment bound phosphorous into the water column, however, this reduction in currents also contributes to the risk that the basin will become anoxic (Van de Moortel et al., 2010). P retention within different conventional wetlands ranges from 40-60%, around 45 to 75 $g/m^2/yr$ (Vymazal, 2007), most of this being due to settling (and associated processes such as accretion and soil adsorption). P removal from FTWs is usually higher due to the additional filtering properties of the roots, reaching 81% (White & Cousins, 2013).

Soil adsorption
Phosphorus is retained in the soils by binding to the soil surface. Soils with high clay content have high P adsorption capacity, which increases with lower pHs. Organic soils also adsorb P, with the adsorption capacity dependent on mineral components (Rhue & Harris, 1999). Al and Fe fix phosphorus in acidic soils, whilst Ca and Mg fix it in alkaline soils (Kadlec & Knight, 1996). This adsorption process is reversible, with an equilibrium between the bound P and the dissolved P in the soil porewater. The soil minerals and binding sites result in a 'phosphate buffering capacity' which determines where this equilibrium exists (Barrow, 1983). This has important implications for P removal, since reducing inflow P can cause P desorption from the sediments, actually producing a higher P outflow than inflow (Belmont et al., 2009).

Precipitation of P
P adsorption occurs in aerobic waters, but as conditions become anoxic (reducing conditions) metals within the soil change valency, becoming soluble. This causes the release of phosphorus as a co-precipitate (precipitating due to the action of a true precipitate) from the soil (Kadlec & Wallace, 2009). In very low oxygen conditions, where the soils are anaerobic (Eh < -200 mV) sulphate reduction occurs (Figure 4). This creates free sulphide which preferentially binds with Fe (as iron sulphide) preventing iron mineralisation of P. Thus, anaerobic conditions promote the release of P back into the water column (Kadlec & Wallace, 2009).
Plant uptake
Plant uptake of P reaches only around 6% (Masters, 2012). If a FTW has a P removal up to 81% (White & Cousins, 2013), this means around 75% is removed predominantly by settling or storage in other sinks. Much of the P in plant uptake is also difficult to remove permanently from the system by harvesting because it is stored in the roots, or it re-enters the system as litter (see Section 2.8 Harvesting). Vymazal (2007) considers that harvesting of conventional wetlands is only useful in low P effluents (e.g. polishing) with around 10-20 g $P/m^2/yr$, where uptake is not limited by growth rate. FTWs may be able to absorb more P, due to their roots being suspended directly in the effluent, and plant roots are more accessible for harvesting, but dredging is still likely to be the most effective method of permanent removal.

Microbial and Algal uptake
Bacteria and algae are important in P cycling within the soils, rhizosphere and water column (Vymazal, 2007). P uptake by microbes in conventional wetlands is very fast, but they store very little (Vymazal, 2007). Thus, having higher surface area and consequently higher microbial mass, microbes in a FTW are likely to be a larger sink of P than in conventional treatment wetlands, however nutrient cycling is likely to result in little net removal, except through sedimentation of dead organic microbial matter.

Fish Uptake
In South East Asia it is common to use fish for nutrient recovery in ponds receiving human effluent (Cairncross & Feachem, 1993). Fish eat periphyton (such as algae, cyanobacteria, heterotrophic microbes, and detritus) (Azim et al., 2005) as well as fungi, protozoa, phytoplankton, zooplankton, invertebrates and invertebrate larvae, and some species are piscivorous. In treatment wetlands fish are usually chosen for their adaptation to low oxygen levels, for example *Gambusia affinis* (mosquito fish) in warm temperate to tropical conditions, and *Notrophus fundulus* (black-stripped top minnow) or *Umbra limi* (central mudminnow) in temperate climates with over 77 different fish species being used in North American treatment wetlands (Kadlec & Wallace, 2009). Sometimes *Oreochromis* spp.

(Tilapia) and Bass have colonised previously unpopulated treatment wetlands (Kadlec & Wallace, 2009).

Li & Li (2009) examined nutrient removal from aquaculture effluent using floating islands (17% cover) planted with the aquatic vegetable *Ipomoea aquatic.* There was artificial aeration and it was populated with *Aristichthys nobilis* (silver carp), *Siniperca chuatsi* (mandarin fish; carnivorous) and *Carassius auratus gibelio (*crucian carp). Around 34% of TN and 18% of TP was removed from the system, and of this around a third (34%) of removed TP and TN was removed by fish. This was around the same that was removed by sedimentation.

Kania (2014, unpublished) suggests that FTW facilitate the sustainable growth of fish and demonstrates that FTW significantly increase fish biomass that can be harvested from the waterway. Fish harvesting enables P removal from the effluent with fish being made into meal which can be used for pork or poultry farming or in pet food. There must be no toxins or toxic metal contaminants in the effluent, especially contaminants that may bioaccumulate. Also, if it is to be sold for human consumption the fish need to be cooked well since there is the potential for contamination by pathogens, particularly the tapeworm *Clonorchis sinensis* (Cairncross & Feachem, 1993).

Fish can disturb bottom sediments, releasing P, particularly those that feed or nest on the bottom e.g. *Cyprinus carpio* (Carp) (Kadlec & Wallace, 2009, p.696).

Problems with phosphorous removal

Generally, wetland treatment only produces temporary storage of P, in contrast to N and C which can be released as gases through microbiological degradation (N_2 and CO_2). Indeed, Yousefi and Mohseni-Bandpei (2010) stated that P can be considered as a conserved entity. Most P is stored in sinks such as sediments (95%; Masters, 2010), plants, microbes and algae, but this P is recycled. These sinks give an initial period of apparent P removal. However, once the wetland is established, nutrient cycling results in similar outflow P levels to inflow. Even regular harvesting of plants only removes around 6% (Masters, 2012) of P inflow, if both the roots and shoots are harvested. Thus, Kavanagh & Keller (2007) concluded that at least 90% of P eventually passes through a wetland system and is released in the effluent.

Some wetland treatment systems can even export more P than they receive, such as a stormwater wetland in North Carolina which had median removal efficiencies of – 95% to 70%; at times exporting twice as much P as it was receiving (Line et al., 2008). This can occur both due to physical disturbance of the sediments releasing P, the re-release of P from biodegradation of organics (Sundaravadivel & Vigneswaran, 2001), or anoxia which can also result in the sudden release of P as a co-precipitate (Maine et al., 2005).

Sudden P releases into the water column can potentially have other detrimental effects. Since P is usually the limiting factor for biological activity in freshwaters (Schindler et al., 2008), a large P release can result in nitrogen becoming limiting. This promotes the growth of Cyanobacteria blooms which as well as producing harmful toxins, also extract N from the atmosphere (Conley et al., 2009).

Masters (2012) is thus emphatic that dredging is important for long term removal of phosphorus from a FTW. Kadlec and Wallace (2009) detail projected working life of different types of wetlands with different soils, ranging from around 10 to 170 years, but dredging around every 10 years (Masters, 2010) would be ideal for sustained P removal with most effluents.

A minor route of P removal is phosphine (PH_3). It is usually found in very low amounts (e.g. 47 ng/m^3 of water in marshes), mostly bound to sediments but with around 10% of this dissolved in the water (Hana et al., 2010). However, it can be released from highly anaerobic wetlands (Eh < - 200mV) (Gassmann & Glindemann, 1993) as phosphine gas. Devai and Delaune (1995) calculated a gaseous release rate of 1.7 g P/m^2/yr from a bulrush wetland treatment system.

Thus, treatment wetlands have various sinks (algae, plants, microbes, soils) which vary in their capacity to absorb P from the effluent based on conditions such as available surface area, soil type, pH and redox potential. FTWs limit the resuspension of particulates since the islands reduce water movement within the wetland and the roots filter out particulates (Borne et al., 2013), thus increasing P sedimentation. However, dredging is essential to long term functioning of a FTW for P removal, and regular harvesting can be useful at low P loadings (Figure 1).

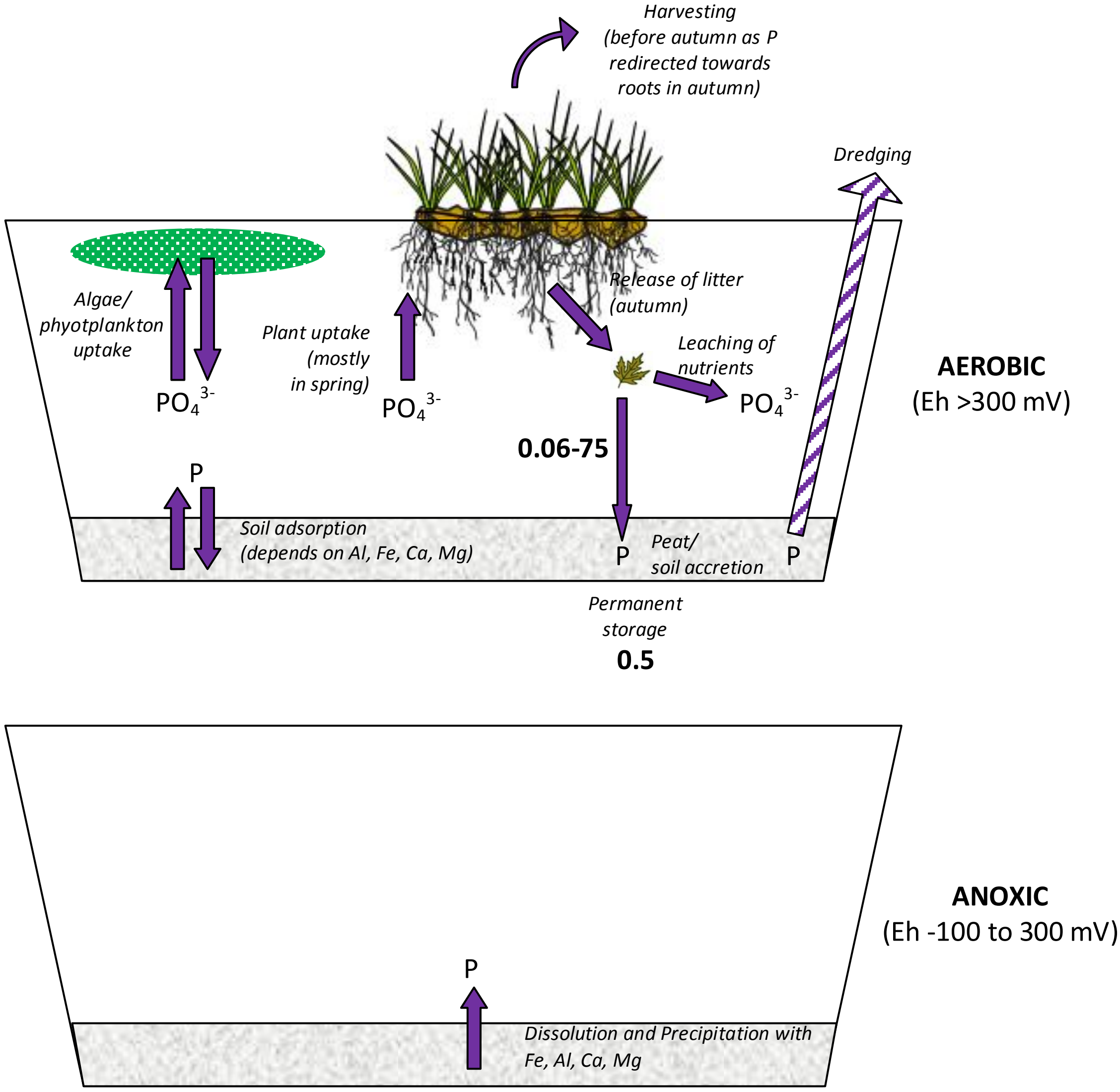

Figure 2. Summary of phosphorus processes in aerobic and anoxic wetlands.
Soil/peat accretion and soil adsorption is the major process and major (95%) sink. However, sorption of P into the soil is reversible. Without harvesting or dredging P removal eventually stops. Numbers in **bold** are g P/m^2/yr that may be removed or added during the processes; *italics* indicate the name of the process, with specific conditions required in brackets.

2.2 Nitrogen removal

Nitrogen is the principal target for treatment in many wetlands. Effluents contain organic nitrogen compounds, which break down principally to ammonia, from which nitrite and nitrate can form through a microbiological nitrification process. Different micro-organisms within anoxic zones can denitrify this nitrate to permanently release N_2 gas from the basin. Agricultural wastes may already have high concentrations of nitrate nitrogen as they enter the wetland. Conversion between different forms of N depends on many factors, including DO, available carbon and pH.

2.2.1 Nitrogen removal in aerobic water

Ammonification (mineralisation)

Dead and decaying organic material is broken down in to ammonia by microbes, either utilising the energy released or absorbing the ammonia for use as microbial biomass. Ammonification increases with temperature, being optimal at 40-60 °C, and with organic compound availability (especially when they have low C/N ratios) (Reddy & Patrick, 1984). Optimum pH is between 6.5 and 8.5 (Vymazal, 2007). Ammonification usually takes place under aerobic conditions (oxidative deamination).

Equation 1: Break down of organic N (example with amino acid) to ammonia

$$RCH(NH_2)COOH + H_20 \rightarrow NH_3 + CO_2$$

Ammonification rates can vary greatly e.g. between 0.004 and 0.53 g $N/m^2/d$ (Reddy & D'Angelo, 1997; Tanner et al., 2002). The root zone of a FTW is likely to be a good location for ammonification.

Ammonia volatilisation

Ammonia exists in an equilibrium between its dissolved ammonium form (NH_4^+) and its gaseous form (NH_3); Equation 2. Below pH 8.0 ammonia loss as gas is negligible (Reddy & Patrick, 1984). At a pH around 9.3 losses due to volatilisation can become significant (Vymazal, 2007). N removal rates due to ammonia volatilisation have been measured at 2.2 g $N/m^2/d$ in wetlands (Stowell et al., 1981).

Equation 2: Conversion of dissolved ammonium to ammonia gas

$$NH_4^+ + OH^- \Leftrightarrow NH_3 + H_2O$$

Algal photosynthesis often elevates pH values during the day (Vymazal, 2007), thus increasing ammonia volatilisation. However, FTWs may inhibit this due to (i) islands shading algae and reducing the area of the air-water interface, and (ii) plants releasing humic acid, which reduces the pH (Van de Moortel et al., 2010).

Nitrification

Within aerobic water micro-organisms convert ammonium to nitrate in a process called nitrification. Directly adjacent to plant roots there is an aerobic zone (Reddy et al., 1989), which means that FTW are likely to have elevated denitrification rates due to the availability of root surface area.

Kadlec & Wallace (2009; p.280) note that nitrification in wetlands is quite different from nitrification in conventional Waste Water Treatment Works (WWTWs). Whilst nitrification is commonly considered a two step process in conventional WWTWs, in natural wetlands it is now believed to have three stages (Bothe et al., 2000); Equation 3.

Equation 3: The three stage nitrification process, converting ammonium to nitrite, then nitrate.

Nitritation (2 stages)

$$NH_3 + O_2 + 2H + 2e^- \xrightarrow{\textit{Nitrosomonas}} NH_2OH + H_20$$

$$NH_2OH + H_20 \xrightarrow{\textit{Nitrosomonas}} NO_2^- + 5H + 4e^-$$

Nitrification (1 stage)

$$2NO_2^- + O_2 \xrightarrow{\textit{Nitrospira or Nitrobacter}} 2NO_3^-$$

Due to the different processes less oxygen and alkalinity is consumed in wetlands during nitrification than in conventional WWTWs (Kadlec & Wallace, 2009). *Nitrospira* is also much more prominent as a nitrifier than *Nitrobacter* in wetlands (Austin et al., 2003).

Nitrification is influenced by temperature (optimum 25-35 °C), pH (optimum 6.6-8), alkalinity, microbial populations present, DO and ammonium concentrations (Vymazal, 1995). Below 4 °C nitrifying bacteria *Nitrosomonas* and *Nitrobacter* do not grow (Paul & Clark, 1996). Kadlec & Wallace (2009; p.280) note, unlike WWTWs, there is little evidence that a low C/N ratio in wetland effluents improves nitrification rates.

In wetlands, for every g of ammonium oxidised to nitrate 2.28 g of oxygen and 7.1 g of alkalinity as calcium carbonate are consumed (Kadlec & Wallace, 2009; p.279) i.e. nitrification requires aerobic conditions and will consume alkalinity and oxygen, becoming increasingly acidic and anaerobic. Wetlands have nitrification rates of 0.01 to 2.15 g $N/m^2/d$ (mean of 0.048) (Reddy & D'Angelo, 1997; Tanner et al., 2002), though this may be much higher for FTWs due to the large root surface area within the aerobic zone.

Low oxygen conditions can result in nitrite (NO_2^-) being produced instead of completing the process toward nitrate (Bernet et al., 2001). The consequence of this is that in a later denitrification stage, some of the nitrite is converted into nitrous oxide (N_2O), a potent greenhouse gas. Sufficient oxygenation in nitrification basins is therefore recommended.

2.2.1 Nitrogen removal in anoxic water

Denitrification

Denitrification is the microbiologically mediated conversion of nitrate into nitrogen gas, which is then released from the wetland into the atmosphere. A carbon source is required for denitrification. The equation can be written in many ways, depending on the source assumed (Equation 4).

Equation 4: Denitrification of methanol, producing nitrogen gas and alkalinity

$$6NO_3^- + 5CH_3OH \longrightarrow 3N_2 + 5CO_2 + 7H_20 + 6OH^-$$

In many ways denitrification is the converse of nitrification, making the water more alkaline and requiring anoxic or anaerobic conditions. Microorganisms denitrify because in the absence of dissolved oxygen for reduction, they reduce nitrate. Although methanol is used for illustration here as a source of carbon, usually it is large organic molecules. It is calculated that per g of NO_3^- around 3.02 g of organic matter (or 2.3g of BOD) is consumed, and around 3g of alkalinity as $CaCO_3$ is produced (Kadlec & Wallace, 2009).

The optimum pH is 6 to 8 (Paul & Clark, 1996) being negligible below pH4 (Vymazal, 2007). Denitrification is very slow below 5 °C, but increases with temperature up to 60 or 75 °C, then decrease rapidly (Paul & Clark, 1996). More nitrate can speed up the process, but the limiting factor in denitrification is often the carbon supply (Kadlec & Wallace, 2009), especially if BOD has settled out in previous treatment basins. A C/N ratio of 5:1 is suggested to ensure carbon does not become limiting (Baker, 1998) although this may be an overestimate if much of the C is labile (Kadlec & Wallace, 2009). Lower pHs can assist with breaking down lignin in cell walls, increasing the litter quality for denitrification processes (Ding et al., 2012).

Often an anaerobic denitrification basin is placed after an aerobic nitrification basin. This enables all the ammonium to be converted to nitrate prior to denitrification, thus maximising total N removal. However, even in a well oxygenated basin there are areas of low mixing, and deeper waters and sediments, where oxygen levels are low enough to produce denitrification (Figure 3, Figure 4), and in anoxic basins nitrification can occur on the surface of roots where the plants have transported oxygen (Kadlec & Wallace, 2009, p.281). Thus both nitrification and denitrification processes can be achieved within a single basin, though controlling the treatment efficiency may be more difficult.

Floating islands can aid denitrification by producing anoxic conditions through the restriction of oxygen diffusion into the water column. Also, roots and plant litter, as well as coconut coir on islands (Baquerizo et al., 2002),can act as sorption sites, with biofilms developing which increase denitrification rates and thus NO_3 removal rates (Vymazal, 2007). Denitrification releases are about 0.003 to 1.02 g N/m^2/d in wetlands (Vymazal, 2007), though this could be higher in FTWs due to more biofilm area and more sorption sites.

Anaerobic Ammonia Oxidation (ANAMMOX)
The bacteria involved in this process were only discovered in 1999. *Planctomycetes Nitrosomonas eutropha* utilises ammonium ions and nitrite (from nitrification of ammonium) to produce nitrogen gas. This can be represented as in Equation 5.

Equation 5. The ANAMMOX process

Formation of nitrite

$$2NH_4^+ + 3O_2 \text{ -------------}\rightarrow 2NO_2^- + 4H+ + 2H_2O$$

ANAMMOX

$$NH_4^+ + NO_2^- \text{ -------------}\rightarrow N_2 + 2H_2O$$

This denitrification process uses less than half the oxygen (1.94g O per gram of NH_4^+) of the standard denitrification process, and requires no carbon substrate (Kadlec & Wallace, 2009). ANAMMOX processes occur in many types of wetlands when there is severely restricted oxygen. Bishay & Kadlec (2005) found that in a Free Water Surface wetlands there were more ammonia losses than could be accounted for by the oxygen consumed under normal dentification. There was also a lot of nitrite present in this wetland, and very little carbon, suggesting that these conditions were conducive to the ANAMMOX reaction.

Plant uptake

Nitrogen uptake by plants in conventional wetland treatment is low (up to 6-8%) compared to microbial denitrification (up to 61-63%) (Metheson et al., 2002). Vymazal (2007) estimates that for conventional wetland systems plant harvesting is useful for N removal if loading is only around 100-200 g N/m^2/yr. If N removal is a priority, designing and operating the basins to maximise nitrification/denitrification by microorganisms is probably more cost effective.

N is predominantly taken up by plants in the form of ammonia, but also as nitrate. Much of this is returned to the system when tissues senesce (Kadlec & Wallace, 2009).

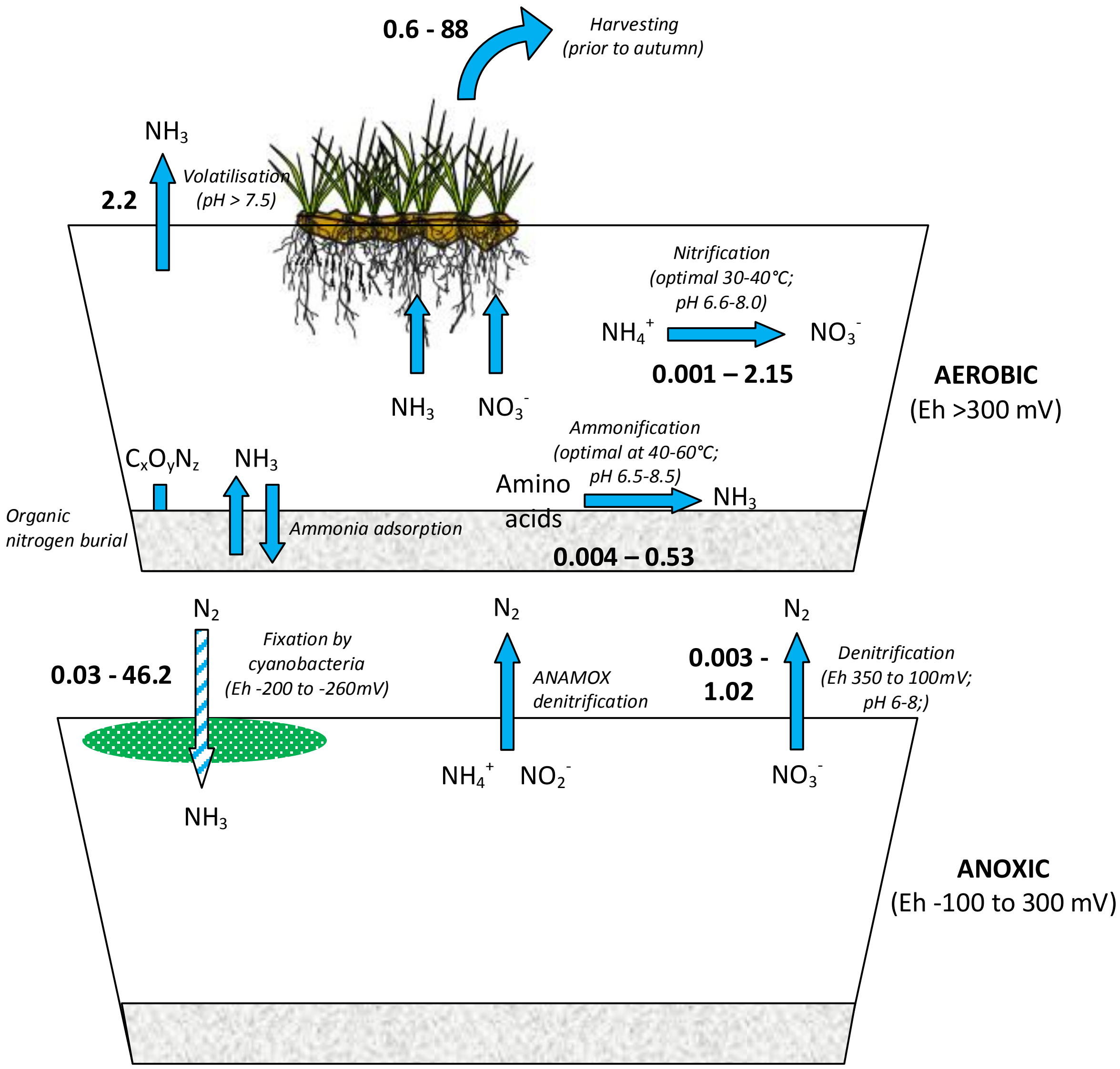

Figure 2. Summary of nitrogen processes in aerobic and anoxic wetlands. Primary settlement of effluent is assumed prior to entering the wetland. Numbers in **bold** are g N/m^2/yr that may be removed or added during the process; *italics* indicates the process, with specific conditions required in brackets. The striped blue arrow indicates nitrogen fixation that would not normally occur unless the anoxic pond becomes anaerobic (Eh < -200 mV).Permanent removal of N is only through ammonia volatilisation (minor), denitrification (including ANAMOX) and harvesting. Organic nitrogen burial (associated with litter) and ammonia adsorption (associated with clay soils) are relatively minor processes.

Problems with nitrogen removal
NH_4 removal rates in conventional wetlands vary between 35 and 50% in Europe (Verhoeven & Meuleman, 1999; Vymazal, 2002). FTWs have shown removal rates from -45% to 75% for NH_4 and between 36% and 40% for total nitrogen (Boutwell, 2002; DeBusk & Hunt, 2005; Gonzalez et al., 2005).

Problems with nitrogen removal are associated with producing the correct microbiological conditions; aerobic for nitrification and anoxic for denitrification, as well as ensuring sufficient carbon supply for the later. These are discussed in the design section.

2.3 Oxygen

Factors influencing oxygen concentrations
Wetlands typically have slow flow, incomplete mixing, and rapidly decreasing oxygen profiles with depth (Figure 3). Anoxic zones develop just below the substrate in shallower basins and also in the lower regions of the water column in deeper basins (Kadlec & Wallace, 2009). Oxygen can be rapidly depleted in wetlands due to microbiological activity, particularly with nitrification and decomposition (Kadlec & Knight, 1996). FTWs exacerbate oxygen depletion both due to high rates of microbiological activity (nitrification) and due to the islands restricting diffusion of oxygen back in to the water i.e. reducing air-water contact area and reducing wind disturbance (Van de Moortel et al., 2010). This makes FTWs particularly susceptible to unwanted drops in DO, especially at high percentage cover of islands.

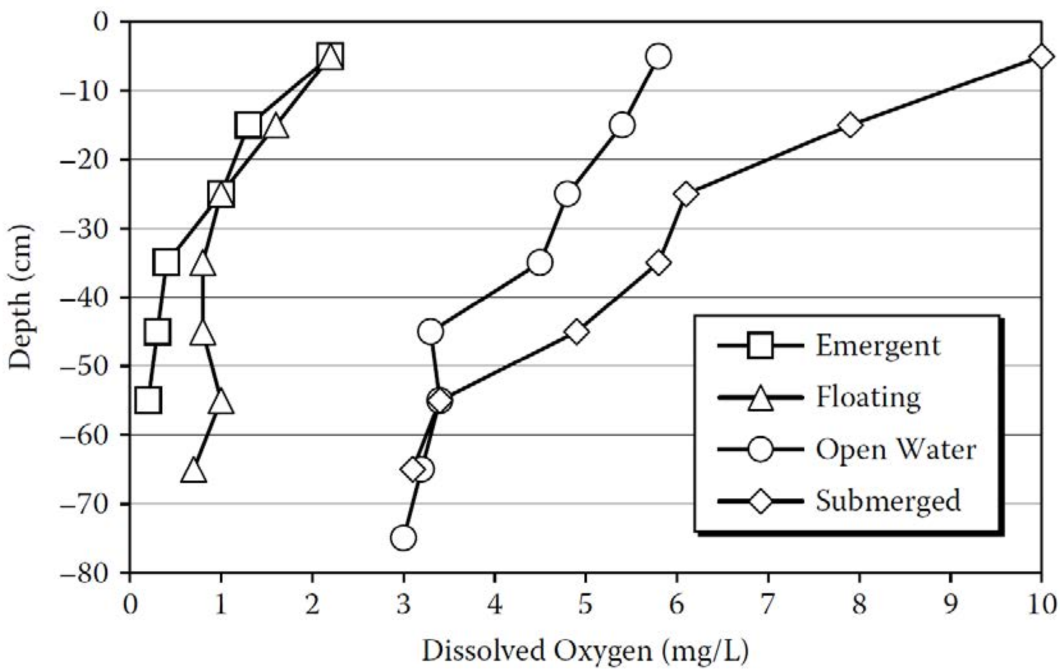

Figure 3. Vertical profiles of dissolved oxygen in various types of FWS (Free Water Surface) wetlands, Florida. Data from 141 profiles collected over a 2½ year period. Data from Chimney et al. (2006), Figure from Kadlec & Wallace (2009). FTWs are most readily compared to floating plant systems.

Temperature affects
Oxygen saturation of water varies with temperature: at 25 °C dissolved oxygen is 8.2 mg/l, and at 5 °C it is 12.8 mg/l. However Kadlec and Wallace (2009) note that the poor mixing of waters limits the dissolution of oxygen such that reaeration is very slow, even in open

wetlands. They estimate that it takes 2 to 4 days to reaerate an open wetland basin from 0 to 90% DO, with typical winds. This is likely to be even slower in FTWs.

Plants

Submerged photosynthesising plants and algae release in the range of 0.26 and 0.96 $g/m^2/d$ of O_2 during photosynthesis (Kadlec & Wallace, 2009, p138), oxygenating the water. Emergent plants bring O_2 to the roots, but O_2 delivery usually matches respiration requirements, so there is little net input into the water column (Brix & Schierup, 1990). Studies by Tanner and Headley (2011) and White and Cousins (2013) both found a high level of oxygen depletion in basins due to floating islands.

Tanner and Headley (2011) not only illustrated how oxygen depletion is higher in FTWs, but also that oxygen depletion is higher when there are plants rather than mats with artificial roots (sisal) (Table 1). This oxygen depletion is likely due to the higher rate of microbiological activity associated with plant roots. Although the relationship between oxygen depletion and root biomass was weak, there was little oxygen depletion due to the floating mat alone and even the mat with artificial roots.

Table 1. Oxygen depletion (%DO) at subsurface and bottom of mesocosms after 7 days due to the effect of Floating Islands, ordered from highest to lowest. Influent DO was 95%, floating island coverage was 50%. Root biomass (dry weight) also shown. Adapted from Tanner and Headley (2011).

	subsurface DO (%)	bottom DO (%)	Root biomass (g/m^2)
Control (no floating mat, but equivalent shading)	87	85	
Floating mat only	85	84	
Mat + soil + artificial roots	85	84	
Mat + soil media	80	79	
Mat + soil + Juncus edgariae	68	66	299
Mat + soil + Schoenoplectus tabernaemontani	68	67	184
Mat + soil + Carex virgata	58	57	533
Mat + soil + Cyperus ustulatus	50	48	329

Van de Moortel et al. (2010) found redox potentials to be decreased due to floating islands: at both 5cm and 60cm depths the FTW has much lower O_2 than an open water basin: at 5cm redox is 68 (open water) cf. -25 (FTW); at 60cm redox is: -93 (open water) cf. -122 (FTW). They did claim that roots can aerate island matting. However, there was little difference between the mat redox potential (72 mV ± 478) and the redox potential 5cm below the surface of an open water basin at (68 mV ±225).

A liability with FTWs is that during summer periods, due to high rates of microbiological activity and insufficient O_2 exchange with the atmosphere, the basin can become anaerobic, causing sulphide toxicity which then kills the plant roots (Lamers et al., 2002) and consequently reducing the effectiveness of treatment. Reduction in treatment efficiency due to anoxia was found in several studies, but usually when the floating islands occupied 50% or more of the surface water area (Van de Moortel et al., 2010; Borne et al., 2013).

2.4 Redox potential

Oxidation is the loss of electrons during a reaction. This is usually through a substance combining with oxygen, as it is energetically the most favourable oxidant. Redox potential is the tendency of a system to oxidise substances i.e. in high redox potential water, incoming organic substances will be rapidly oxidised (an oxidising environment) whereas in low redox potential waters substances will be reduced (a reducing environment). An example of reduction would be where hydrogen combines with carbon to produce methane.

Redox potential is strongly associated with the oxygenation of the water, but it is not identical, since substances other than O_2 can oxidise. Zonation usually occurs in a wetland with oxygen being the oxidiser near the surface, then as DO decreases other substances become oxidisers, with reactions releasing less energy with successively weaker oxidisers. This is in the order O_2, NO_3^-, MnO_2, FeOOH, SO_4^{2-} then CO_2.

The decline in free oxygen reflects the redox potential (Eh), also known as the oxidation-reduction potential (ORP), of the water i.e. the tendency of a chemical to acquire electrons, measured as electric potential (mV). At Eh > 300mV (measured with a platinum electrode) conditions are considered aerobic, at < -100 mV conditions are anaerobic, and between these (near-zero Dissolved Oxygen) conditions are anoxic (Figure 4).

Redox Potential	Reactions		Zone
> +300 mV	Oxygen reduction	I	Aerobic

+ 100 to +300 mV	NO_3^- and Mn_4^+ reduction	II	Anoxic
+100 to – 100 mV	Fe_3^+ and Mn_3^+ reduction	III	
-100 to -200 mV	SO_4^{2-} reduction	IV	Anaerobic
< -200 mV	CH_4 formation	V	

Figure 4. Redox zonation in wetlands, based on Kadlec & Wallace (2009). This vertical zonation can be found in deep lentic environments, particularly where there is high oxygen consumption e.g. my microorganisms.

At high redox potential phosphorus can form insoluble complexes with oxidised iron, calcium and aluminium. Organic compounds which comprise most of the BOD are oxidised using oxygen by bacteria, releasing carbon dioxide. At lower redox potentials organic material does not decay quickly. The water is anoxic, with reducing conditions predominating. Manganese and iron are both reduced (Equation 6)

Equation 6. Reduction of manganese and iron in anaerobic conditions

$$Mn^{4+} + 2e^{-} \rightarrow Mn^{2+} \qquad Fe^{3+} + e^{-} \rightarrow Fe^{2+}$$

This reduction causes metals to precipitate out of the sediments back into the water column, bringing P with them, as a co-precipitate (Van de Moortel et al., 2010).

Further decreases in oxygen (below -100mV) result in anaerobic conditions, whereby sulphate is reduced to hydrogen sulphide, which although soluble, can be released as gas at low pH (Kadlec & Wallace, 2009). Usually this reduction is undesirable in wetlands, except in acid mine treatment.

Equation 7. Reduction of sulphate to hydrogen sulphide

$$SO_2^{-4} + 2CH_2O \rightarrow H_2S + SHCO_3^{-}$$

Eventually, at very low redox potential (below -200mV) CO_2 , formate, or acetate, is reduced to methane (CH_4) by bacteria.

Equation 8. Reduction of carbon dioxide to methane.

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_20$$

2.5 BOD, Suspended Solids and Carbon

Biological Oxygen Demand

Biological Oxygen Demand (BOD) is a measure of oxygen consumption by microorganisms due to the oxidation of organic matter; usually measured in the lab over 5 days (BOD_5). BOD of inflows are typically high, unless the treatment basin is being used just for polishing previously treated wastes. BOD decreases rapidly (around 50% decrease within 6 hours) as it passes through a wetland due to decomposition and settling of organic carbon, finally reaching a non-zero plateau (Kadlec & Wallace, 2009). Even if the waters are not aerobic, fermentation and sulphate reduction can remove carbon from the system.

Carbon

Most carbon entering a wetland is organic. Microbiological processes are the main method for removing carbon, through the oxidation of organic compounds, releasing energy. In aerobic waters, respiration takes place (Equation 9), releasing CO_2 to the atmosphere. In anaerobic zones there are four main processes which can take place: (i) fermentation producing either lactic acid or ethanol (ii) methanogenesis producing gaseous methane (iii) sulphate (SO_4^{2-}) reduction producing carbon dioxide and hydrogen sulphide, and (iv) denitrification, producing carbon dioxide and gaseous nitrogen.

Settling is also an important removal method (although the carbon is retained in the sediments). In FTWs plants have been shown to remove around 5.9 g BOD/m^2/day. The large surface area provided by roots can produce a higher rate of microbial decomposition (Brisson & Chazarenc, 2009), but roots also physically entrap particulates onto the biofilm which then fall in clumps and settle out, providing a significant removal pathway for suspended solids (Smith & Kalin, 2000; Headley & Tanner, 2006; Van de Moortel et al., 2010; Borne et al., 2013). Settling is further encouraged by flow resistance through the roots and flow reduction caused by wind shielding of the surface. Particulate carbon, and carbon bound in litter, if it is not decomposed, accumulates in the sediments, particularly where conditions are anaerobic (Kadlec & Wallace, 2009).

Equation 9. Microbiological decomposition of organic compounds.

Respiration

$$C_6H_{12}O_6 \rightarrow CO_2 + H_2O$$

Fermentation

$$C_6H_{12}O_6 \rightarrow 2CHCHOHCOOH \quad \text{(lactic acid)}$$
$$C_6H_{12}O_6 \rightarrow 2CH_3CH_2OH + CO_2 \quad \text{(ethanol)}$$

Methanogenesis

$$\text{(acetate)} \quad CH_3COO^- + 4H_2 \rightarrow 2CH_4 + H_2O + OH^-$$

Sulphate reduction

$$\text{(lactate)} \quad 2CH_3CHOHCOO^- + SO_4^{2-} + H^+ \rightarrow 2CO_2 + 2H_2O + HS^- + 2CH_3COO^- \quad \text{(acetate)}$$

Denitrification (see Equation 4)

Unlike submerged plants, which obtain carbon from the water, carbon uptake by emergent plants is from atmospheric CO_2. Plants thus bring carbon into the system through photosynthesis and the deposition of organic matter. However, the net effect of plants in wetlands is to reduce BOD due to plant respiration, increased settling, and increased decomposition processes (Masters, 2012). Also, where there is carbon limitation in anoxic or anaerobic basins, the C provided by the deposition of litter can be important in increasing denitrification rates (see Section 2.2.1).

Settling of BOD is also affected by basin depth, residence time and water movement (Kadlec & Wallace, 2009). Theoretically higher temperatures should increase microbial decomposition rates. Bacteria have limited activity below 5°C, but in conventional wetlands there is no significant temperature dependence above this (Akratos & Tsihrintzis, 2007; Kadlec & Wallace, 2009). This may be due to limitations in oxygen transfer rates or restricting factors in one or more of the many C processes (Kadlec & Wallace, 2009).

In anoxic (reducing) conditions, the presence of sulphate contributes to the removal of organic matter (BOD/COD) by acting as a coagulant and thus increasing settling rates (Huang 2005).

2.6 Metal removal

Metal removal from wetlands is predominantly through forming complexes with organic matter, and through being coated in iron or manganese oxyhydroxides (Kadlec & Wallace, 2009). This either occurs in the sediments, or they settle out into the sediments. Under anoxic conditions Cu, Zn, Pb, Ni and Ca form insoluble metal sulphides which will settle out. Even in aerobic basins, decomposition of organic matter usually means there is an anoxic layer just below the surface oxic layer (≈1cm) in which these metal sulphides can form.

Predicting metal removal from wetlands can be very difficult, depending on the structure of the sediments and many factors of the water chemistry, with models regularly being wrong by orders of magnitude (Kadlec & Wallace, 2009). Factors that affect metal removal include the Cation Exchange Capacity (CEC) of the sediments, pH (circumneutral usually being optimum), redox potential, the availability of sulphur for the formation of metal sulphides, and the formation of iron and manganese oxyhydroxides (which allow co-precipitation) (Kadlec & Wallace, 2009). Organic soils with humic acids and phenolics increase the CEC and thus adsorption of metals. Sedimentation of metals can result in long term storage, depending on the availability of organics with which metals can complex, although metal accumulation can eventually saturate the soil sink and result in biological toxicity (Kadlec & Wallace, 2009). Thus (careful) dredging is required in the long term to permanently remove metals and ensure the wetland continues to operate effectively.

Uptake by plants is much less important than that by sedimentation, and where metals are taken up, they are mostly stored in the roots. Table 2 shows percentage removal of metals by plants in a conventional wetland and how this is allocated in the roots and shoots.

Table 2. Percentage removal of metals by plants in a conventional treatment wetland and how this is allocated to the roots and shoots (adapted from Nolte and Associates, 1998).

Metal	Roots (%)	Shoots (%)	Total (%)
Ag	2.0	0.0	2.0
As	10.1	0.6	10.7
Cd	13.3	0.0	13.3
Cr	16.8	2.2	19.0
Cu	5.5	0.6	6.1
Hg	6.7	0.0	6.7
Ni	4.7	0.3	5.0
Pb	11.8	2.0	13.8
Zn	6.1	0.4	6.5

Despite plant uptake being low, FTWs have been shown to greatly increase metal removal compared to unvegetated retention ponds. For example Borne et al. (2013) compared treatment in a normal stormwater retention pond with one retrofitted with a floating island. With concentrations of 0.0092 mg Cu/l and 0.035 mg Zn/l in the inflow, particulate Cu and Zn removal was 19% and 40% (respectively) in the normal pond, and 50% and 65% with a floating island. Tanner & Headley (2011) found the removal of dissolved Cu and Zn to be 5% and 1% without a floating island, and 50% and 47% with an island. These authors believe that the benefit of the floating island wasn't principally due to plant uptake. Indeed, Tanner & Headley (2011) found mean plant uptake rates were 0.059-0.114 mg Cu/m^2/d and 1.2-3.3 mg Zn/m^2/d, accounting for less than 4% of Cu removal and less than 10% of Zn removal. This was a mesocosm experiment without bottom sediments and with predominantly dissolved metals, so values of plant uptake were probably higher than they would be in a normal FTW.

Tanner & Headley (2011) and Borne et al. (2013) considered that the improved performance with floating islands was due mainly to: (i) interception by the plant roots, (ii) humic acid release from the plants, which reduced alkaline waters to circumneutral pH (Van de Moortel et al., 2010), improving metal complexation and therefore flocculation and settling (Mucha et al., 2008) and (iii) The islands reducing the redox potential to the extent that insoluble metal sulphides formed.

The exact mechanisms of metal removal depend on the specific metal. Most zinc within effluents is in particulate form and is removed predominantly through settling, sorption to organic sediments and chemical precipitation/co-precipitation (Kadlec & Wallace, 2009). It can form precipitates with sulphur (ZnS) and carbonate from the water ($ZnCO_3$) and it co-precipitates with Fe, Mn, Al oxyhydroxides. However, ZnS does not readily precipitate in neutral waters (Younger, 2000), only in more alkaline waters (>7.5). Also, for co-precipiration, the other metals must be present in the effluent, and even then, Fe and Mn oxides are not stable in anoxic waters (Knox et al., 2004). Warmer water temperatures are also correlated with Zn removal, probably due to increased sorption rates (Borne et al., 2013). Aerobic wetlands are expected to absorb about 0.04g $Zn/m^2/d$ (PIRAMID consortium, 2003). Similar to Zn, Cu removal rates increase with temperature, however adsorption is better at more neutral pH (Borne et al., 2013). They also concluded that reduced oxygen resulted in a high production of Cu sulphide precipitates in basins with floating islands.

High loadings of effluent and insufficient adsorption capacity or saturation of the potential sinks (organic carbon, metal hydroxides, high CEC soils) can result in decreasing treatment capacity as well as increasing toxicity. Toxicity can be a biological problem, particularly in open water treatment systems where birds, amphibians and freshwater invertebrates have direct access to the basin (as opposed to subsurface flow systems) (Kadlec & Wallace, 2009). Sorption capacity in studies listed by Kadlec and Wallace estimate between 20 and 780 years operation of a wetland with metal loading. Careful dredging (avoiding resuspension) can be applied to remove contaminated sludges/soils. In mixed wastewater effluents from WWTWs it is likely that the necessity for P removal through regular dredging is higher than that from metal accumulation.

2.7 pH

pH has a profound effect on the functioning of wetlands, as mentioned in previous sections. Several studies have confirmed the effect of floating vegetated islands in reducing pH. In a two year study by White and Cousins (2013) pH decreased from 8.6 to 6.2. After only 11 days Van de Moortel et al. (2010) found a significant pH decrease from 7.5 to 7.0 whilst the control (without an island) stayed constant at around 7.5. Borne et al. (2013) found a difference between the control (8.3) and the FTW (7.3), which aided Cu adsorption. Interestingly Tanner and Headley (2011) didn’t notice a drop in pH in mesocosm tanks, although they still found that treatment was enhanced with floating islands, attributing the difference in to the release of bioactive compounds. The researchers who found differences in pH generally agreed that humic compounds were released by the plants, reducing pH. White and Cousins also acknowledged that alkalinity consumed during microbial nitrification on the plant roots could also be a driving force behind dropping pH within aerobic basins.

2.8 Harvesting of Floating Island Plants

FTWs are a relatively new technology with few long term studies, and few details on plant harvesting. The prime functions of plants in FTWs is (i) for their roots to intercept and filter particulates, aiding sedimentation, (ii) to increase the rates of microbiological processes by providing a high surface area on which microorganisms respire, nitrify or denitrify, and (iii) to alter the physic-chemical and chemical environment i.e. increase microbiological processing through the release of humic acids and through reducing DO exchange (acidity and lower oxygen increasing denitrification) and carbon deposition (increasing denitrification). Harvesting is therefore not essential to long term management of FTWs, and although it can help with permanent removal of nutrients and metals, removal rates are typically low. For example, in subsurface flow wetlands plants only removed 2-8% of total nitrogen (Tanner, 2001; Yousefi & Mohseni-Bandpei, 2010) and 3-12% of total phosphorous (Yousefi &

Mohseni-Bandpei, 2010), with microbes believed to be removing the rest of the N, and settling removing the rest of the P. Even with total uptake for N and P estimated at around 6%, all of this is unlikely to be harvested as it is stored in both the roots and shoots, and nutrients are returned back to the wetland through deposition of senescent material.

Practicalities of harvesting

Floating islands facilitate easy harvesting. Often larger islands are able to support the weight of humans, and so cutting could be done directly on the island. Smaller islands can be pulled towards the shore and even lifted out. In contrast with other wetlands where the vegetation is rooted in the sediments, in FTWs both roots and shoots can be removed, and with little disturbance to the sediments. Theoretically a replacement island could be installed immediately, although this may not be cost effective. Also, removal of root mass is likely to be more detrimental to treatment than the gains from permanent removal of the nutrients. For example, FTWs typically increase N and P removal rates by around 20-40% (Table 14), whereas P and N removal by harvesting the whole plant is at the most 6%.

Storage of nutrients in plants

The start of the growing season, in early spring and prior to maximum growth rate, is the time of highest P uptake. However, prior to autumn senescence, much of the P is relocated to the root stock for the following year (Vymazal, 2007). Thus, if removal of P is a priority, harvest timing and frequency is extremely important, with a recommendation that it is done not only prior to senescence, but also during the peak growth period. The P lost in the senescent material re-enters the basin system very rapidly; up to 30% lost through leaching within the first few days of decomposition (Vymazal, 2007).

Although shoot biomass tends to be larger than root biomass (see plants in design section), there is generally more N, P and K stored in the roots than in the shoots, especially when autumn approaches (White & Cousins, 2013; Winston et al., 2013) e.g. Figure 4.

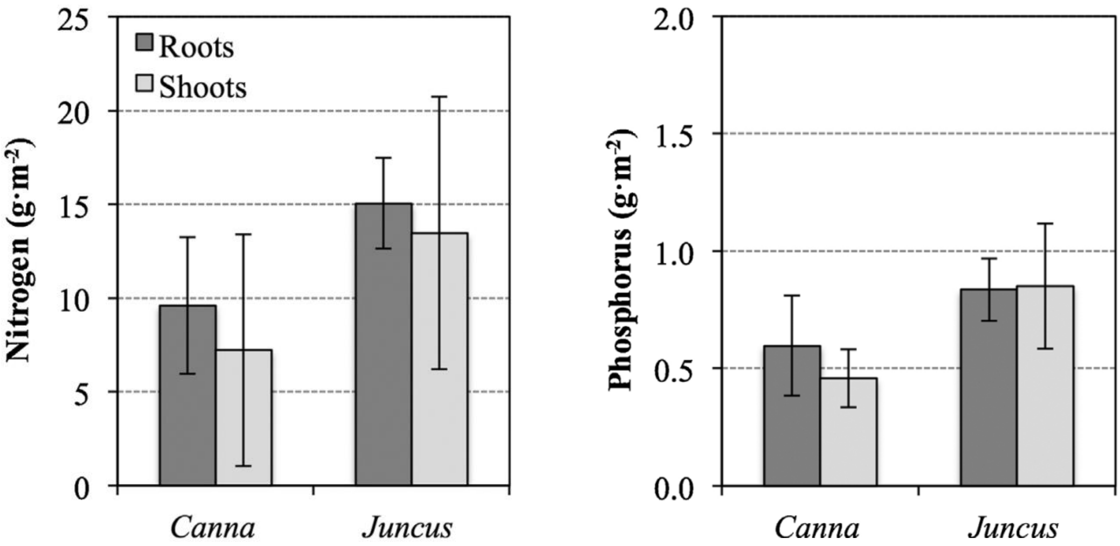

Figure 4. Nitrogen and phosphorus in roots and shoots of *Canna flaccida* and *Juncus effusus* after one summer of growth (harvested 18 September 2008). Nutrients are per m^2 of floating island. Three replicates per bar, with standard error indicated. From White & Cousins (2013).

Storage of metals in plants

Storage of metals tends to show either an even distribution between roots and shoots (e.g. Cu) or predominant storage in the roots (e.g. Zn) (Tanner & Headley, 2011). Table 3 shows uptake of copper and zinc in roots and shoots over a 7 day trial.

Table 3. Uptake of copper and zinc in roots and shoots of four different plant species over 7 days in a FTW, measured as µg/m^2/d. Adapted from: Tanner and Headley (2011).

	Cu		Zn	
Plant species	**roots**	**shoots**	**roots**	**shoots**
Cyprus ustulatus	54	61	3027	282
Carex virgata	54	89	1228	934
Juncus edgariae	38	41	1703	760
Schoenoplectus tabernaemontani	36	24	881	320

Relative importance of different processes

Restricting flow and intercepting particulates on roots is one of the prime benefits of FTWs, consistently removing more BOD and P than open water treatment ponds. However, using synthetic root structure (sisal) Borne et al. (2013) showed that the physical structure alone does not account for most of the benefits of FTWs; water chemistry changes, and to a much lesser extent plant uptake, assist with improving treatment.

3. Treatment Efficiency

Treatment efficiency obtained within a FTW is highly dependent on appropriate design and proper operation, as well as the characteristics of the inflow and the objectives of the treatment. At one end of the scale are FTWs designed for aerobic treatment with mixing or air bubbled into the system, often with low % island coverage and addition of calcium carbonate to aid nitrification reactions. These basins are predominantly to remove ammonium. At the other end of the scale are anaerobic basins with up to 100% island coverage, with addition of carbon in the form of e.g. molasses, to supply the denitrification process. Thus, in aerobic basins, ammonium removal may be high whereas nitrate is produced and may exceed inflow nitrate concentrations. In the latter, denitrification reactions remove nitrate, but ammonium may not be nitrified, resulting in NH_4^+ increasing (due to organic carbon decomposition) such that outflow exceeds inflow. Sometimes floating islands achieve very high rates of removal because of a tightly controlled DO, pH and carbon supply in a hydroponic system. The concentrations of pollutants also affects the removal rate, with higher inflow concentrations often resulting in higher removal rates.

There can be different flow regimes, such as plug flow, where a quantity of effluent is kept in the basin for around 3-7 days, continuous flow, or sporadic flow (such as storm events). Some mesocosm and lab based studies use synthetic effluent with dissolved nutrients, which may exaggerate treatment efficiencies, especially for P and metals which are usually bound to particulates.

Thus, the main considerations when examining performance of a FTW are:

1. Dissolved oxygen: aerobic/anoxic/anaerobic. Natural aeration or artificial aeration through bubblers. With aerobic basins tending to towards nitrification and anaerobic basins tending towards denitrification.
2. Carbon sources: either naturally, through organic carbon, or added artificially, to enhance denitrification rates. Decomposition of organic carbon also results in increased ammonia production within the basin.
3. pH: with alkaline pH increasing nitrification and acidic pH increasing denitrification.
4. Root mass: aiding removal of particulates due to physical filtering and settling processes
5. Mixing: circulation of water to aid the nutrient supply to microbiological processes.
6. Plug flow or continuous flow: affecting residence times and nutrient gradients.
7. Concentrations of inflow pollutants: with higher nutrient supply increasing rates of decomposition/nitrification/denitrification unless limited by another factor.
8. Changes in the FTW chemistry with time. Often pH and redox potential drops due to microbiological processes and restriction of oxygen diffusion from the surface.

Thus, direct comparison between different FTWs has little meaning, and the best comparison is with a relevant control basin. This is often a basin without an island which is receiving the same effluent, however sometimes it is before and after the retrofitting of an island, which doesn't guarantee exactly the same effluent inputs.

New treatment systems can take over a year to stabilise, and even then they can have high variation in treatment efficiency, especially if environmental conditions vary or sinks (such as sediment adsorption) become saturated. However, significantly higher performance of FTWs can be noticed in as little as two days (Van de Moortel et al., 2010), particularly in relation to nitrification/denitrification and other processes which are predominantly dependent on microorganisms, due to their fast response time (Kadlec & Wallace, 2009).

In this section, treatment efficiency from peer-reviewed FTW studies will be summarised separately, with relevant details supplied, and compared to a control where possible. Where complete columns are blank, there was no information.

Abbreviations follow this system: NO_x represents nitrate in the form or either NO_2 or NO_3; TN is Total Nitrogen; N_{org} is organic nitrogen; Cu_{tot} is total copper; Cu_{part} is particulate copper; Cu_{diss} is dissolved copper; DRP is dissolved reactive phosphorus; BOD is biological oxygen demand; COD is chemical oxygen demand; PBP is Particle Bound Phosphorus; TKN is Total Kjeldahl Nitrogen.

Table 4. Removal rates in the study by Van de Moortel *et al.* (2010). Close to 100% coverage of the island resulted in reduced redox potential and anoxic conditions. This resulted in high NO_3 removal rates, but poor NH_4 (thus TN) and P removal rates.

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	2.16	1.77	**18**		1.9	**12**
TN	mg/l	21.8	13.1	**40**	*identical*	19.5	**11**
NH_4	mg/l	16.1	10.8	**33**	*to*	16.5	**-2**
NO_3	mg/l	0.37	0.2	**46**	*control*	0.08	**78**
N_{org}	mg/l	4.31	1.6	**63**	*inflow*	2.87	**33**
TOC	mg/l	27.7	16.4	**41**		23	**17**
COD	mg/l	81.3	46.6	**43**		51.4	**37**
Cu	mg/l	10.0	5.5	**45**		8.4	**16**
Fe	mg/l	454	325	**28**		259	**43**
Mn	mg/l	164	153	**7**		176	**-7**
Ni	mg/l	10.0	6.1	**39**		5.75	**43**
Pb	mg/l	6.10	3.4	**44**		4.58	**25**
Zn	mg/l	57.5	29.7	**48**		47.6	**17**
SO_4^{2-}	mg/l	64.2	49.8	**22**		53.7	**16**
pH	mg/l	7.35	7.08	**4**		7.48	**-2**
cond	µS/cm	1035	1017	**2**		1015	**2**

Table 5. Removal rates in the study by White & Cousins (2013). Troughs of 1.15 m^2 and 3.03 m^2 were used with 100% island coverage and soluble fertiliser added to pond water as the inflow.

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	$mg/m^2/day$				37.2	15.4	**59**
TN	$mg/m^2/day$				320	106	**67**

Table 6. Removal rates in the study by Yang *et al.* (2008). This was a lab based hydroponic study with synthetic effluent (dissolved P and N, but also organic matter used), although the objective was to represent a nursery run-off treatment system, which naturally has few suspended solids. 100% island coverage was used with purposely anaerobic conditions, 3 day batch process, and glucose added to aid denitrification. Thus, high NO_x removal rates were obtained, but NH_3 removal was negative as decomposition of organics was still taking place but with limited or no nitrification.

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l				1.25	1.17	**6**
TN	mg/l				3.76	2.59	**31**
NH3	mg/l				0.93	1.19	**-28**
NO_x	mg/l				1.39	0.12	**91**
COD	mg/l				41.8	34.8	**17**
DO	mg/l				0.01	0	**100**

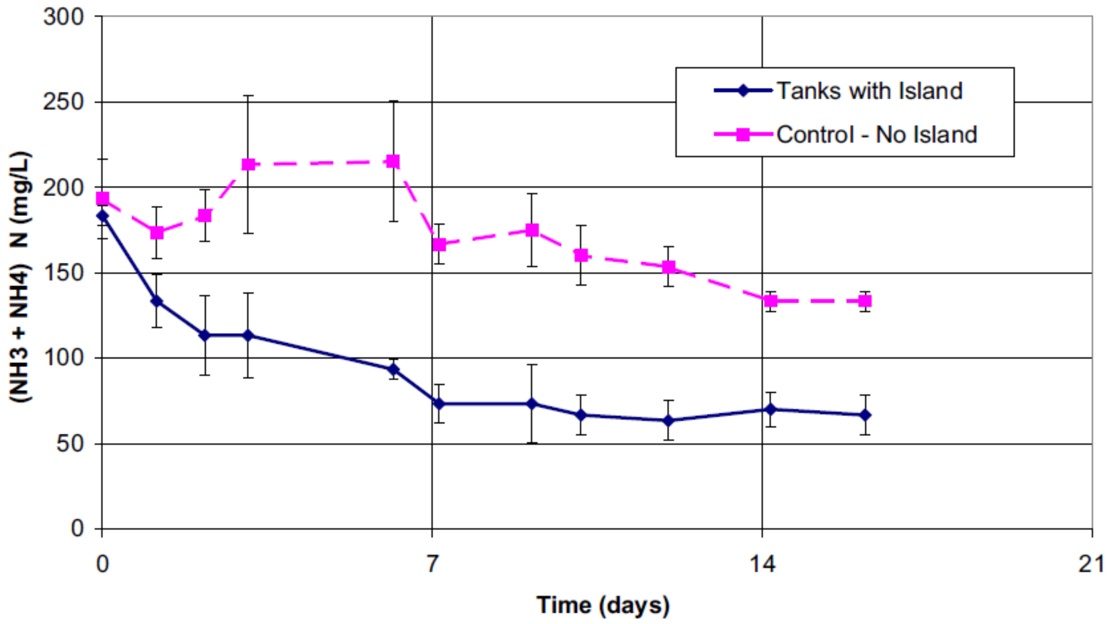

Figure 5. Ammonium removal rates; graph from the study by Stewart *et al.* (2008). An **aerobic** lab experiment with 100% island coverage, calcium carbonate added, and aerated with a bubbler. Synthetic effluent was created using liquid fertiliser (soluble). Conditions were optimised for ammonium removal.

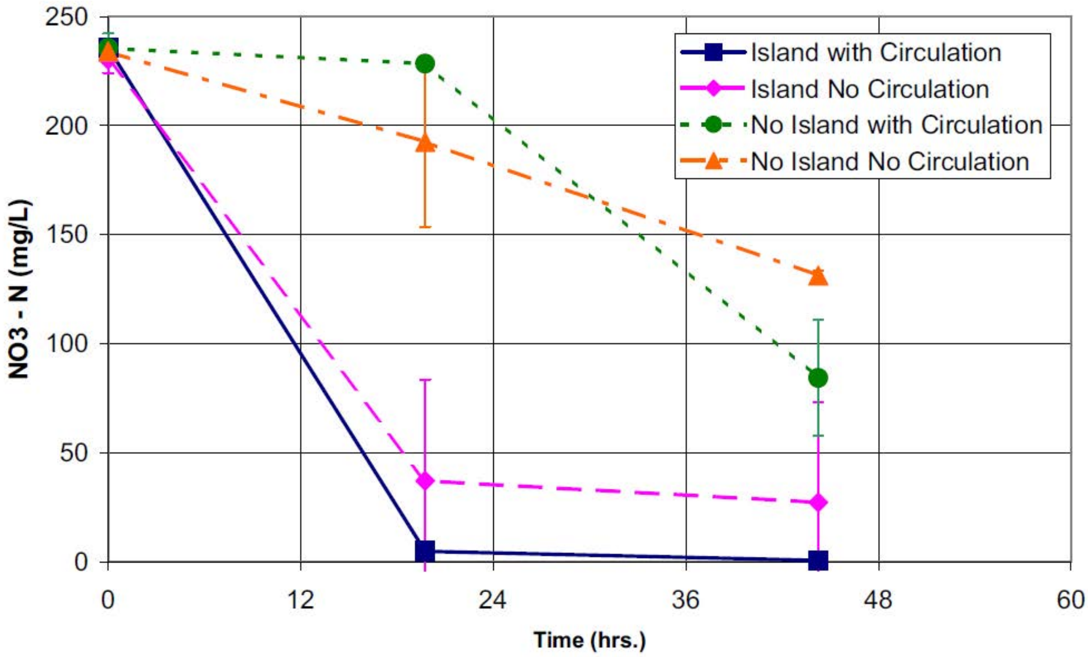

Figure 6. Nitrate removal rates; graph from the study by Stewart *et al.* (2008). An **anaerobic** lab experiment with 100% island coverage and carbon (molasses) added. In some of the replicates water was circulated by a pump. Synthetic effluent was created using liquid fertiliser (soluble). Conditions were optimised for nitrate removal. Redox potential in the control decreased from +200mV to +48mV, but in tanks with islands it decreased to -200mV (much better for denitrification).

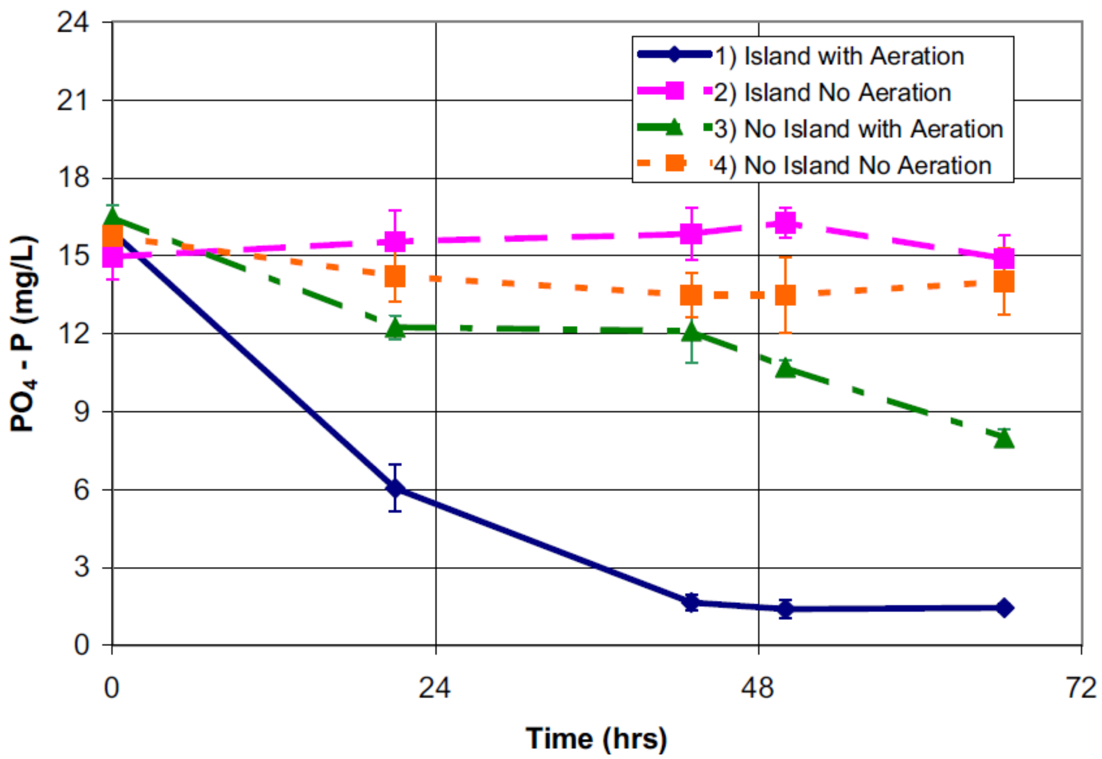

Figure 7. Phosphate removal rates; graph from the study by Stewart *et al.* (2008). Both **anaerobic and aerobic** basins were tested (conditions as in Figures 5 and 6) without islands and with 100% cover of islands. Phosphate removal was best achieved when there was both aeration and floating islands.

Table 7. Removal rates in the study by Borne et al. (2013). A control stormwater retention pond was compared with a retention pond with 50% cover of floating island, receiving the same effluent. Data was retrieved from a graphical presentation of inflow and outflow effluent concentrations.

	control			FTW		
	inflow	outflow	**% removal**	inflow	outflow	**% removal**
TSS	30	24	**20**	*identical*	12	**60**
Cu_{tot}	0.0090	0.0075	**17**	*to*	0.0057	**37**
Cu_{part}	0.0035	0.0030	**14**	*control*	0.0019	**46**
Cu_{diss}	0.0049	0.0044	**10**	*inflow*	0.0038	**22**
Zn_{tot}	0.035	0.022	**37**		0.013	**63**
Zn_{part}	0.027	0.017	**37**		0.010	**63**
Zn_{diss}	0.006	0.005	**17**		0.005	**17**

Table 8. Removal rates in the study by Tanner & Headley (2011). After 7 days batch experiment with 1m x 1m mesocosms and 36% island cover. Artificial stormwater used. FTW results are from the plant species which gave best results (*Cyperus ustulatus*).

	control			FTW		
	inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP			**3**			**58**
DRP			**-5**			**60**
Cu_{tot}			**7**			**57**
Cu_{diss}			**5**			**50**
Zn_{tot}			**-1**			**19**
Zn_{diss}			**1**			**37**
Turbidity - subsurface			**24**			**67**
Turbidity - bottom			**24**			**67**
DO (subsurface)			**8**			**39**
DO (bottom)			**11**			**40**

Table 9. Removal rates in the study by Stefani *et al.* (2011) based on median values. Effluent was from aquaculture, following conventional activated sludge treatment. There was a 19% cover of islands and a continuous flow (0.09 m/s).

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l				0.55	0.19	**65**
SS	mg/l				350	320	**9**
COD	mg/l				15	5	**67**
BOD	mg/l				4.2	2	**52**
pH					7.3	7.2	**1**
cond	µS/cm				645	645	**0**

Table 10. Removal rates in the study by Winston *et al.* (2013). The study examined a stormwater retention pond before (control) and after (FTW) retrofitting an **18%** coverage of floating island. Data is a mean over different storm events.

		control			FTW		
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	0.26	0.11	**58**	0.41	0.05	**88**
PBP	mg/l	0.13	0.04	**69**	0.17	0.03	**82**
OP	mg/l	0.13	0.07	**46**	0.24	0.02	**92**
TN	mg/l	1.01	0.41	**59**	3.49	0.43	**88**
NH_3	mg/l	0.10	0.05	**50**	1.6	0.04	**98**
TKN	mg/l	0.88	0.35	**60**	3.32	0.37	**89**
NO_x	mg/l	0.12	0.06	**50**	0.17	0.06	**65**
N_{org}	mg/l	0.89	0.34	**62**	1.72	0.33	**81**
TSS	mg/l	216	24	**89**	252	13	**95**

Table 11. Removal rates in the study by Winston *et al.* (2013). The study examined a stormwater retention pond before (control) and after (FTW) retrofitting an **9%** coverage of floating island. Data is a mean over different storm events.

			control			**FTW**	
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	0.26	0.17	**35**	0.19	0.12	**37**
PBP	mg/l	0.13	0.05	**62**	0.07	0.05	**29**
OP	mg/l	0.14	0.12	**14**	0.12	0.07	**42**
TN	mg/l	1.64	1.05	**36**	1.17	0.61	**48**
NH_3	mg/l	0.12	0.11	**8**	0.11	0.05	**55**
TKN	mg/l	1.43	0.97	**32**	0.84	0.55	**35**
NO_x	mg/l	0.20	0.08	**60**	0.34	0.06	**82**
N_{org}	mg/l	1.50	0.93	**38**	0.72	0.5	**31**
TSS		354	30	**92**	101	22	**78**

Table 12. Removal rates in the study by Chang *et al.* (2013) **during** storm events in a functioning stormwater retention pond; assessed before (control) and after (FTW) fitting floating islands with 8.7% cover. Nutrient concentrations are given as the means over several different storm events. The pond contained a fountain.

			Control			**FTW**	
		inflow	Outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	0.028	0.027	**4**	0.058	0.050	**14**
OP	mg/l	0.006	0.006	**0**	0.021	0.010	**52**
TN	mg/l	0.300	0.377	**-26**	0.626	0.526	**16**
NH_3	mg/l	0.048	0.052	**-8**	0.102	0.104	**-2**
NO_x	mg/l	0.006	0.017	**-183**	0.062	0.029	**53**

Table 13. Removal rates in the study by Chang *et al.* (2013) **outside** of storm events in a functioning stormwater retention pond; assessed before (control) and after (FTW) fitting floating islands with 8.7% cover. Nutrient concentrations are given as the means over different sampling times. Notice that the treatment rates are much higher than during storm events (probably due to lower flows and thus higher retention times).

			control			**FTW**	
		inflow	outflow	**% removal**	inflow	outflow	**% removal**
TP	mg/l	0.037	0.034	**8**	0.055	0.029	**47**
OP	mg/l	0.003	0.002	**33**	0.020	0.004	**80**
TN	mg/l	0.303	0.349	**-15**	0.655	0.552	**16**
NH_3	mg/l	0.121	0.103	**15**	0.208	0.102	**51**
NO_x	mg/l	0.025	0.022	**12**	0.032	0.025	**22**

Overview

With more cover of floating islands there is a tendency for redox potential to drop due to reduced O_2 diffusion from atmosphere. This leads to denitrification processes dominating in which there is high removal of NO_3 but low removal of NH_4 (Yang et al., 2008; Van de Moortel et al., 2010). Indeed NH_4 removal may be negative due to decomposition of organics to NH_4 without subsequent removal by nitrification (Table 4 and 6). Aeration can prevent this NH_4 accumulation by encouraging nitrification, and it also prevents P release from sediments that can occur at low redox potentials (Figure 7).

Low % island cover had detrimental effects on treatment efficiency, as did lower residence times. For example, TN removal was 48% with 9% island cover in the Winston et al. study (2013), but this increased to 88% TN removal with 18% cover (Tables 10 and 11). Similarly Chang *et al.* (2013) found only 14% TP removal with 9% cover during storms, but outside of storm flows this removal increased to 47% (Tables 12 and 13).

Around 20% cover seems optimal if the basin is to be maintained as an aerobic system without artificial aeration, and still achieve good removal efficiency. Beyond this point it is probably worth using 100% cover, with a choice between a high nitrate removal anaerobic basin, or artificial aeration (bubbling) to produce a high treatment rate aerobic basin. Stewart et al. (2008) illustrates how tightly controlled conditions and addition of calcium carbonate (nitrification) or carbon (denitrification) can be used to optimise treatment rates. Stewart et al. (2008) also showed that nitrification and denitrification processes can be achieved in a single aerobic tank if tightly controlled. Treatment efficiencies noted by Stewart and White & Cousins (2013) are likely to be around the maximum achievable in FTWs due to the use of soluble fertilisers in their experiments and their tightly controlled hydroponic systems. Therefore when assessing potential performance of a new FTW we must decide whether it will be a tightly controlled situation or more of a field based FTW.

Table 14 summarises these studies. The improvements through using Floating Islands, as discussed, vary due to conditions, however we can expect between around 2 and 55% increase in P removal compared to a Free Water Surface wetland, and a 12 to 42% increase in N removal. Metal removal is also considerably higher in FTWs (20-50% higher). Most importantly, if conditions are tailored for denitrification (anaerobic and sufficient carbon supply) NO_3 removal can be up to 100% in FTWs.

Table 14. Summary of % removal rates in different studies for main nutrients and metals. 'Improvement' examines the increase of treatment efficiency of FTWs beyond the control wetlands (FST wetlands). The Van de Moortel study is excluded from the fianl comparison since anaerobic conditions produced P release and is not an example of good FTW management.

	Study	Moortel	White	Yang	Stewart*	Stewart*	Borne*	Tanner	Stefani	Winston	Winston	Chang	Chang	**RANGE**
	% cover	**100**	**100**	**100**	**100**	**100**	**50**	**50**	**19**	**18**	**9**	**8.7**	**8.7**	**8.7-100**
	notes	anaerobic	anaerobic	anaerobic	aerobic	anaerobic						storm	non-storm	
FTW	TP	12	59	6	91			58	65	88	37	14	47	**6-91**
FTW	TN	11	67	31						88	48	16	16	**11-88**
FTW	NH_4	-2		-28	66					98	55	-2	51	**-28-66**
FTW	NO_3/NO_x	78		91		100				65	82	53	22	**22-100**
FTW	Cu_{tot}	16					37	57						**16-57**
FTW	Zn_{tot}	17					63	19						**17-63**
controls	TP	18			53			3		58	35	4	8	**3-58**
controls	TN	40								59	36	-26	-15	**-26-59**
controls	NH_4	33			26					50	8	-8	15	**-8-33**
controls	NO_3/NO_x	46				42				50	60	-183	12	**-183-60**
controls	Cu_{tot}	45					17	7						**7-45**
controls	Zn_{tot}	48					37	-1						**-1-48**
improvement	TP				38			55		30	2	10	39	**2-55**
improvement	TN									29	12	42	31	**12-42**
improvement	NH_4	*study*			40					48	47	6	36	**6-48**
improvement	NO_3/NO_x	*excluded*				58				15	22	236	10	**10-236**
improvement	Cu_{tot}						20	50						**20-50**
improvement	Zn_{tot}						26	20						**20-26**

Notes:
(*) indicates data extracted from graphs.
In Stewart study, PO_4^{3-} was assessed instead of P

3.1 Seasonal Variation

Seasonal variation in FTW is due to (i) temperature variations, which affect plant and especially microbial productivity, (ii) consequent DO variations due to increased oxygen demand when there is increased microbiological activity, and to some extent the solubility of oxygen in water at different temperatures, and (iii) seasonal growth patterns in plants.

The effect of season on treatment efficiency depends on the main processes involved in their removal, particularly how temperature and oxygen variations affect these processes. For example, spring and autumn are peak P uptake periods for vegetation in wetlands (Kadlec & Wallace, 2009) however, the main process of P removal is settling and adsorption, so seasonal P removal was found to vary less than that of nitrogen (Wittgren & Maehlum, 1997).

Studies have shown conflicting results over how variable treatment efficiency is over different seasons, particularly with N removal, but this is likely to be due to differences in limiting factors. As previously mentioned, plant uptake as NO_3 or NH_4^+ tends to be relatively small compared to microbiological processes (Riley et al., 2005). Thus, studies have found N removal to be affected by seasonal temperature variation (Spieles & Mitsch, 2000; Picard et al., 2005). However, Maehlum and Stalnacke (1999) and Mander et al. (2000) found little difference in N removal between warm and cold climates and Van de Moortel et al. (2010) found more variation due to temperature in P than in N. It is likely that these differences are due to other factors that may be limiting, particularly anoxia. For example, in the study by Van de Moortel et al. (2010) there was low NH_4 removal as 100% island coverage produced low DO and reducing conditions, nullifying any further potential N removal increases due to increased temperature. Also, as mentioned previously, in practice decomposition is not found to be highly temperature dependent in wetlands (Akratos & Tsihrintzis, 2007; Kadlec & Wallace, 2009) and therefore ammonia production rates are not likely to change much with temperature. Thus, interactions between season, light, temperature and DO mean that an individual variable is not a good predictor of activity, and the net effect can be counter-intuitive (Stein & Hook, 2005; Kadlec & Wallace, 2009). However, despite these interacting effects very low temperatures (5 °C) certainly restrict microbiological activity and plant growth (Mitsch and Gosselink 1993).

Rainfall

Rainfall can have a large and varied effect on pollutants entering a basin. If the inflow is from a combined sewer system rainfall events can massively increase dilution and flow rates into the wetland. Van de Moortel et al. (2010) found that heavy rainfall caused a significant reduction in inflow conductivity from 1102 µS to 733 µS, and total nitrogen from 23.1 mg TN/l to 16.9 mg TN/l. Then, after rain events although other constitutions remained diluted in the pond, ammonium and nitrate concentrations actually increased (probably due to microbiological activity). With stormwater treatment ponds, the inflow comes from road run-off which has often had an accumulation of metals during the dry period, so initial concentrations during a storm are usually high, as the metals and particulates get washed off the road, but then rapidly decrease as the storm continues and the concentrations become diluted (Barbosa & Dodkins, 2010).

Rainfall and evaporation also have an effect on dilution within the basins (Kadlec & Wallace, 2009). The addition of rainwater can alter the water chemistry (oxygen, pH), rates of microbiological activity, and affect physical processes e.g. increased depth increasing settling. It is also important to consider that when measuring inflow and outflow concentrations, differences may be due to changes in dilution, rather than any removal within the basin, and loading capacities must take rainfall input and evaporation (and drainage) into consideration (Kadlec & Wallace, 2009).

Shading and Temperature
Floating islands can significantly reduce water temperature (by shading) in the warmer months, and also reduce temperature variation if there is sufficient cover (Van de Moortel et al., 2010). However, Winston et al. (2013) with only 18% island cover, found there was little water temperature reduction (preventing them producing conditions for trout to live in the FTW). Van de Moortel et al (2010) also found that although summer tempertures were lower in FTW compared to open water wetlands, winter temperatures were not lower. However, ice still persisted longer in FTWs during the winter due to reduced wind disturbance at the surface in FTWs.

4. Design Considerations

To achieve treatment objectives careful consideration must be taken in design and operation of the wetland. These must be specific to the flow volume, flow variation, the concentrations of pollutant and the required characteristics of outflow. Wetlands can easily be overloaded with sludge so pre-treatment (removal of large material by bar screen or settling of grit and stones) and primary treatment (sedimentation) are essential for domestic effluents prior to entering the wetland. Good design of these initial stages is also extremely important in maximising the treatment efficiency and the cost of running a FTW and to prevent them becoming unnecessarily clogged by high sludge loadings.

4.1 Island Cover

Since floating islands can restrict oxygen diffusion from the air into the water (Smith & Kalin, 2000), island coverage is an extremely important design factor. For example, an almost complete coverage by islands resulted in poor P retention in sediments due to anoxia (Van de Moortel et al., 2010).

The percentage cover of a pond by the island is one of the most important considerations in FTW design. High cover (>50%) can cause anoxia but low cover (9 to 18%) may produce little additional treatment effect (e.g. Winston et al., 2013). The anoxia is not only caused by islands reducing air-water contact, but also because of the high rate of microbiological processes such as nitrification and decomposition. Thus, the optimum size of the island to prevent anoxia is likely to be dependent on the quality of the influent, particularly ammonia/nitrate and organic carbon concentrations. Flow design, mixing and aeration will also be major factors (See section 3. Treatment Efficiency).

Using more island coverage should increase microbiological activity due to the larger root area, however if high rates of aerobic microbiological activity is to be maintained (e.g. nitrification) oxygen consumption will necessarily be high. To maintain high island coverage without depleting oxygen, bubblers can be installed (Stewart et al., 2008). This requires investment and energy costs, and therefore their use depends on a cost-benefit analysis, although energy can be provided by e.g. solar power. Baffles have also been introduced in some FTWs to increase circulation around the roots. Mixing waters to promote aeration should be done with care as disturbance of sediments can liberate trapped P.

4.2 Optimising for N removal

Since P is effectively conservative, but N can be released as gas through correctly managing the microbial environment, strategies for permanently removing N are very different from those for removing P.

Aerobic and Anaerobic basins

Floating islands may increase denitrification by increasing anoxia, although it is preferable to have an oxygenated basin with a high residence time as a first stage to convert most of the ammonia to nitrate in the nitrification process.

Thus, with N the main objective is to convert as much ammonium as possible to nitrate, usually through an aerobic 1st stage, and then to convert as much of this nitrate to N_2 gas, through an anoxic 2nd stage. Oxygen in the aerobic stage can be rapidly depleted with high coverage of FIs and high rates of microbiological activity, so FI cover has to be carefully managed, or artificial aeration has to be included. Sufficient alkalinity must also be available for nitrification, which can be achieved through the addition of $CaCO_3$ (Stewart et al., 2008).

Nitrification reduces DO and pH, though these conditions are ideal for the next (anoxic) denitrification stage. FI cover can be much higher at this stage. Yang et al. (2008) achieved 97% N removal rates with 0% DO in a hydroponic system, though in more natural systems anoxia can cause sulphide toxicity (Lamers et al., 2002) that kill or restrict root growth.

Recycling
Denitrification is predominantly limited by C supply, with a recommended C:N loading ratio of 5:1 (Bishay & Kadlec, 2005). In a two stage system Carbon limitation often occurs because much of the organic C is removed by settling in the earlier aerobic stage (Kadlec & Wallace, 2009), thus releasing nitrate. Additional C can be supplied artificially, e.g. as glucose syrup (Yang et al., 2008) but for most effluent treatment systems it is cheaper and more practical to seed the anoxic basin with raw effluent that has not gone through the anaerobic stage (Kadlec & Wallace, 2009).

Recycling is also used to return anaerobic outflow back to the aerobic stage; denitrification makes the effluent more alkaline, ideal for further nitrification of ammonia (Kadlec & Wallace, 2009). Recycling is now in Danish treatment wetland guidelines (Brix & Schierup, 1990). Figure 5 gives an example of how FTW wetlands could be designed for treatment of domestic effluent, including recycling.

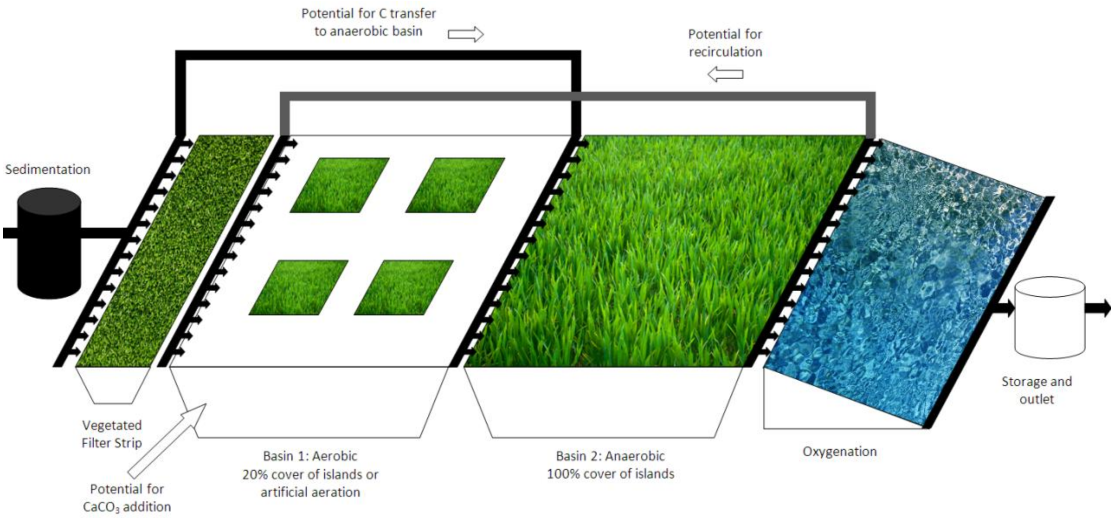

Figure 5. A theoretical design using FTWs for treating low volume domestic effluent (mixture of P, BOD, NH_4^+ and NO_3^- inputs) showing basic design features of combined basins. The vegetated filter strip would have to be adapted/increased/removed depending on solids input from sedimentation.

4.3 Plants
The treatment potential within a FTW depends mostly on the filtering capacity of the roots (root depth and density) and their surface area as a microbiological habitat. Choice of plant species will also affect the rates of nutrient and metal uptake, root/shoot biomass division, growth rates and the way in which the basin water chemistry is altered due to the release of humic acids and protons by plant roots.

Plant dimensions
Tanner & Headley (2011) examined four species growing on floating islands in mesocosms, providing detailed measurements. 90^{th} percentile of root depth averaged between 24 and 48cm, depending on species. The root surface area was between 4.6 and 9.3 m^2/m^2 of floating mat. Above mat biomass was between 834 and 2350 g/m^2 and root biomass of 184-

533 g/m^2 (see Table 3 for more details) with shoot to root ratios of between 3.7 and 4.5. Winston *et al.* (2013) found that *Hibiscus* had shoot:root ratio of 6.3. Indeed, most species have an above mat biomass greater than the below mat biomass, except for Carex spp. such as Carex stricta (Winston et al., 2013), Carex virgata and Cyperus ustulatus (Tanner & Headley, 2011) (Table 3).

Table 5. Mean biomass and shoot:root ratio for FTW plants.

Plant species	**Shoots: Mean above mat biomass (g/m^2)**	**Roots: Mean below mat biomass (g/m^2)**	**Biomass ratio**
[1]*Juncus spp.*	86.3	43.4	2.0
[1]*Carex stricta*	131.4	207.6	0.6
[1]*Spartina pectinata*	121.7	48.1	2.5
[1]*Hibiscus moscheutos*	269	58.9	4.6
[1]*Pontederia cordata*	72	57.7	1.2
[2]*Cyperus ustilatus*	1528	329	4.6
[2]*Carex virgate*	2350	533	4.4
[2]*Juncus edgariae*	1113	299	3.7
[2]*Schoenoplectus tabernaemontani*	834	184	4.5

[1] Winston et al. (2013) in stormwater retention pond
[2] Tanner & Headley (2011) in mesocosm with much more intensive planting

Plant uptake appears to be more associated with total plant biomass rather than root density (Tanner & Headley, 2011), although White & Cousins (2013) found that uptake of N and P by *Juncus effusus* (60.6 N and 3.71 g P/m^2/growing season) was higher than that of *Canna flaccida* (3.71 N and 2.27 g P/m^2/growing season) despite having a similar shoot length, and this was attributed to the much longer roots of *J. effusus*. Nutrient uptake by *J. effusus* was also found to be much higher than that of *Pontederia cordata* in a study by (Chang et al., 2013).

Floating islands are usually allowed 6 months of plant growth to establish before assessing efficiency e.g. (Borne et al., 2013). Once plant growth has reached a maximum (maximum density and shoot biomass) there is no additional net uptake of nutrients by the plants i.e. litter deposition is equal to growth (Kadlec & Wallace, 2009). However, although some of this litter will accumulate on the island, some will sink into the basin with some nutrient release but also with some C and P storage in the sediments.

White & Cousins (2013) found that an increase in nutrient loading increased shoot growth (but not root growth) and suggested that this may be due to a shift in allocation strategy towards shoots when nutrients are plentiful, following Muller, Shmid & Weiner (2000). However, harvesting of the shoots does not appear to affect the root biomass (Borne et al., 2013).

Plant establishment

Vogel (2011) noted that floating island plants have more establishment success and establish quicker, with more cover, when the starting biomass is higher. She recommends planting of as much biomass stock as possible at the start, to aid establishment. The growth rate for some plants may be higher in the first year of establishment, whilst other plants may have higher growth rate in the second year (Svengsouk & Mitsch, 2001).

Buoyancy of islands
Buoyancy of vegetated islands changes seasonally, with mats sinking several centimetres during the spring and summer as the biomass increases (Hogg and Wein 1988 a). Seasonal effects become less pronounced with age, as dead biomass accumulates and decomposition increases (matching biomass accumulation).

4.4 Activated Carbon

Activated carbon is being considered by Frog Environmental Ltd. as a possibility for improving floating island performance prior to the complete establishment of plants by incorporating the carbon within the floating island material. Performance of floating islands usually relates to their ability to increase removal rates for P and N as well as to remove metals, commonly Cu and Zn.

Activated carbon is used in water filters and chemical purification processes. It is highly porous carbon with a high surface area which has been treated by oxygen or sulphuric acid to increase adsorption. It has a surface area of 300-2,000 m^2/g and can adsorb a wide range of pollutants including large organic molecules. Because adsorption works by chemically binding the impurities to the carbon, the active sites in the carbon eventually become filled and adsorption stops. The effectiveness of activated carbon depends on pore size, the carbon source and the manufacturing process.

Typically activated carbon is used to remove metals or organic pollutants rather than nutrients. This is because the surface of activated carbon is negatively charged, attracting positive ions (e.g. Cu^{2+}, Zn^{2+}) rather than negative ions (NO_2^-, NO_3^-). Bhatnagar & Sillanpää (2011) reviewed the adsorption of nitrate on to various carbon substances. Results vary with 1mg/g (Mizuta et al., 2004), 1.7 mg/g (Bhatnagara* et al., 2008) and 4 mg/g (Oztürk & Bektaş, 2004) adsorption of NO_3^-, although these studies are all done in lab conditions and are better than can be expected in the field. Biochar (a form of charcoal) has been tested in field for nitrate removal, though it has tended to have low effectiveness except where nut shells have been the carbon source of biochar (Knowles et al., 2011; Yao et al., 2012).

Nitrate adsorption depends on contact time. Oztürk and Bektaş (2004) achieved complete adsorption within 1 hour at pH<5.0 and 25 °C. Optimal pH for activated carbon adsorption of nitrate occurs at pH2. This is because H+ ions bind to surface and reduce –ve charge, increasing uptake of –ve ions (NO_3^-). Problems with extrapolating these results to the field include (i) the nitrate could be bound to other substances within the water column or sediments, (ii) there would be a diffusion gradient between the site of adsorption (the island) and the bottom of the basin, (iii) the effluent is not being passed through the carbon, so adsorption is passive (iv) optimal pHs for adsorption would not be suitable for a treatment basin, which should be kept around neutral pH.

Ammonia adsorption is around 5.08 mg NH_3/g of carbon at 20 °C increasing to 5.80 mg/g at 60 °C (Long et al., 2008). The temperature of activation of the carbon also affects the adsorption capacity, with higher activation temperatures increasing ammonia adsorption (Ghauri et al., 2012).

P removal in wetlands tends to be predominantly through physical sedimentation processes, which are aided by particle interception by plant roots. When P removal was tested with activated carbon adsorption capacity was 1.11 mg /g at high P concentrations, decreasing with lower P concentrations (Liang et al., 2011). Optimum pH for adsorption ranges between 6 and 10 (Kumarab et al., 2010), which is ideal for FTWs, although again, these studies use data for filtration of nutrients rather than passive adsorption at the surface of the basin.

Adsorption rates of different nutrients are dependent on the nutrient concentration in the effluent, and at low concentrations close to 100% removal is theoretically possible. However, with low circulation and a diffusion gradient within a treatment basin it is unlikely that high percentage removal rates are possible. In addition, we would expect to use around 100 times more carbon (by weight) than the nutrient we are reclaiming, which is likely to be prohibitively expensive.

Activated carbon can provide a carbon source for improving denitrification when C is limiting (Isaacs & Henze 1995; Yang *et al.* 2008). This may be particularly important prior to the establishment of vegetation, which would then provide a source of carbon through decaying organic matter. However, addition between the layers of the floating island may be less useful than simply mixing the powdered activated carbon into the effluent as it enters the basin. Also, a soluble carbon source such as glucose (Yang et al., 2008), acetate or hydrolysate (Isaacs & Henze, 1995) may be better for encouraging denitrification than powdered activated carbon.

5. Conclusions and Recommendations

The main function of floating islands in removing pollutants from effluents is:

- Plant roots assisting in filtering and settling processes for P
- Plant roots acting as a large surface area for micro-organism activity in: decomposition, nitrification, and denitrification (removal of BOD and N).
- Mild acidification of water due to release of humic acids, and a C input from senescent vegetation; assisting denitrification.

P removal is predominantly a physical process. It binds to particulates and removal is assisted by the reduced water movement and the filtering effect of roots on these particulates. This sloughs off to the bottom sediments. Metals are also removed predominantly through binding to particles and sedimentation. Reduced DO in the basin and disturbance of the sediments can result in release of P and metals from the sediments. P is effectively conservative, and if dredging of the sediments is not done (around every 10 years is suggested) the sediment bound P and dissolved P will reach an equilibrium whereby there is no net P removal (and potential for pulses of P in the outflow which are higher than that in the inflow).

N removal is predominantly a microbiological process with NH_4^+ being nitrified to NO_3^- in aerobic basins by nitrifying bacteria, then NO_3^- being denitrified to N_2 gas (and thus released) in anaerobic basins by denitrifying bacteria. FTWs have excellent potential for removing N from effluents. An initial aerobic basin (up to 20% island cover or 100% with aeration) can be used for nitrification and then a second anaerobic basin (100% island cover) can be used for denitrification. Up to 100% N removal is possible, with more tightly controlled conditions increasing the ability to remove N. At the aerobic stage the addition of $CaCO_3^-$ can assist with nitrification (as alkalinity is used up during this process). At the anaerobic stage the addition of C can assist with denitrification (as carbon compounds are used during this process). This C may be added as e.g. glucose or molasses, or as BOD fed from the FTW inlet.

With good management but without hydroponic conditions (i.e. aeration, $CaCO_3$ or artificial C addition) we could expect a FTW wetland to achieve around 60% removal of TP, 75% removal of TN, 50% removal of NH_4^+, 80% removal of NO_3 and 40% removal of metals. All these are expected to be significant improvements (around 20-40% higher) than with basins without islands, depending on specific conditions. More controlled conditions could considerably increase the treatment rates.

Plant uptake only accounts for up to 6% of nutrient (N and P) removal in FTWs. This is also recycled into the system through decomposition unless harvesting is undertaken. Although concentrations of nutrients and some metals (e.g. Zn) are higher in the roots, shoot biomass of plants tends to be higher. Thus shoot harvesting often removes a little over half of the nutrients taken up by the plants. Floating islands also provide access for root harvesting, but harvesting of roots is unlikely to be beneficial as it is more time consuming and also reduces the filtering capacity and microbiological activity associated with the root network: the principal mechanisms of nutrient removal in FTWs. Evidence suggests that removal of shoots does not negatively affect the roots.

FTWs have other advantages over conventional Free Water Surface Wetlands:

- They can adjust to varying water levels

- A higher retention time is possible as they can be made deeper without submerging the vegetation
- Habitat value for birds/amphibians

Recommendations for domestic effluent treatment:
Domestic effluent usually has high BOD, NH_4^+, NO_3^- and P although specific operation and design of FTWs should be tailored to the specific characteristics of the inflow.

1. N removal is the principal benefit of FTWs:
   - An aerobic basin for nitrification is required to convert ammonia to nitrate
   - An anaerobic basin for denitrification is required to convert nitrate to N_2 gas.
   - Although nitrification and denitrification can be achieved in the same basin, separate aerobic/anaerobic basins can be used to more easily control the processes.
2. Depending on cost considerations and inflow water alkalinity, $CaCO_3$ can be added in the aerobic basin to aid nitrification.
3. C can be added in the anaerobic basin to aid denitrification. In smaller treatment systems requiring high water quality outflow a hydroponic system with glucose or molasses addition can be used. For larger treatment systems with greater costs considerations, input of C can come from a controlled input of BOD directly from the FTW inflow.
4. 100% cover of islands, with mixing, is optimal for N reduction in the anaerobic basin.
5. 100% cover of islands with aeration (bubbling) is optimal for aerobic nitrification. If cost considerations prevent aeration, 20% island cover is recommended in the aerobic basin to prevent anoxia occurring.
6. A recycling system from the anaerobic to the aerobic basin, although not always necessary, may be useful when there is excessive NH_4^+ in the outflow i.e. to increase nitrification rates.
7. Aeration is required after the denitrification basin to prevent the release of anoxic waters to the environment.
8. Circum-neutral pH should be maintained in anaerobic and aerobic basins. If pH drops considerably there is a danger of P release.
9. Dredging, particularly of the first (aerobic) basin is recommended every 10 years to remove P trapped in sediments, as well as accumulating metals. Alternative (dormant) treatment basins may be required to be made operational treat effluent as dredging operations are undertaken in the main basin.
10. Plants with high root surface area and high plant biomass are recommended for the floating islands e.g. *Juncus effusus*. Ecological considerations may result in other species being chosen or plant mixtures being used.
11. Harvesting should be done, but only of the shoots.

Use of Activated Carbon
The use of activated carbon between layers of floating island material to assist in pollutant removal will probably have limited effectiveness. This is due to a diffusion gradient between the surface of the basin and the bottom of the basin, and the passive nature of adsorption i.e. the effluent is not being filtered through the medium. At the very most (with high retention times and full adsorption) N and P removal is likely to be about 1g for every 100g of activated carbon used. The proper establishment of plants, a focus on correct basin design, and water chemistry control, is likely to be a much more effective use of resources.

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From: Suplee, Mike
To: Greeley, Carrie
Subject: Nutrient Rules comment for BER-older
Date: Wednesday, April 02, 2014 10:33:29 AM
Attachments: image001.gif

From: Carla Parks, Mayor [mailto:tfallsmayor@blackfoot.net]
Sent: Wednesday, February 19, 2014 2:01 PM
To: Johnson, Elois
Subject: Water Quality Standards Rulemaking

Circular DEQ-12A

I find making rules that are currently impossible to meet dangerous and ludicrous. Ratcheting down the maximum allowable nitrogen and phosphorus as this circular requires is simply not possible. It reminds me of making a law against flatulence. It is impossible to prevent it at this time, but since it is now against the law, we are sure to be able to prevent it in the future. And just trust us, by another rulemaking (separate from the first) we promise you will not be arrested for “infractions”. We all want clean water (and fresh air) and we need to be striving for attainable limits. Really, you make mockery of rules by passing impossible ones. Give us goals we can reach and timelines that are based in reality. And please remember that if it is economically unattainable, it is still unattainable.
Thank you for giving me a chance to give my input on this.
Sincerely,
Carla Parks

Carla M. Parks, Mayor
City of Thompson Falls
PO Box 99
Thompson Falls, MT 59873

Office 406-827-3557
Cell 406-381-6903
e-mail: tfallsmayor@blackfoot.net

From:	Christine Brick
To:	DEQ WQP Admin; Johnson, Elois
Cc:	Greeley, Carrie
Subject:	Nutrient Rule comments
Date:	Tuesday, April 01, 2014 4:49:08 PM

Please accept the attached comments from the Clark Fork coalition on the proposed Numeric Nutrient Standard rule package. Thank you!

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Christine Brick
Science Director
Clark Fork Coalition
P.O. Box 7593
Missoula, MT 59807
ph. 406-542-0539 ext. 202
~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~

CLARK FORK COALITION P.O. Box 7539, Missoula, MT 59807 ph. 406.542.0539

April 1, 2014

Ms. Carrie Greeley & Ms. Elois Johnson
Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

Re: Comments on Numeric Nutrient Standards and Variances

Dear Ms. Greeley and Ms. Johnson,

We submit the following combined comments to DEQ and the BER on the proposed rule package for numeric nutrient standards and variances on behalf of the Clark Fork Coalition. The Clark Fork Coalition (CFC), founded in 1985, is a non-profit organization dedicated to protecting and restoring the 14 million-acre Clark Fork River watershed. We are comprised of 2,700 supporters who are united behind the belief that clean water is integral to the health of our communities.

The CFC has long worked toward reduction of nutrient concentrations in waters of the Clark Fork watershed. We were one of the founding members of the Tri-State Water Quality Council and we were active in development of the Voluntary Nutrient Reduction Program on the Clark Fork that led to the first numeric nutrient standards in Montana. More recently, we've participated in DEQ's Nutrient Working Group. Therefore we are pleased to now see promulgation of a statewide rule package for numeric nutrient standards.

We commend DEQ for your substantial efforts and your patience in developing a rule package that is practical, implementable, and ultimately, we believe, protective of water quality. We appreciate that this is no easy task given the substantial range and variation in natural nitrogen and phosphorus concentrations in Montana, and given the financial challenges of upgrading old and failing wastewater treatment systems in many Montana communities.

Scientific Basis

We are fully supportive of the scientific approach and rationale upon which the standards are based, and we particularly support the emphasis on ecoregion-based dose-response studies in combination with data from reference streams. We appreciate that DEQ submitted their *Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers* for anonymous peer review, and that they responded to that review. We hope that DEQ will continue to refine and update the standards over time as more research is done and more information becomes available.

The process for development of site-specific criteria provides flexibility that should be used to tighten the standards if it becomes apparent that nuisance algae are becoming worse – or not improving – over time. Along with changing the numeric criteria themselves, DEQ should also consider flexibility to change the period of application of the standards beyond the 10 days before or after target dates listed in Table 12A-1 in Department Circular DEQ-12A. As one example, in the upper Clark Fork, low snowpack in some years combined with early irrigation withdrawal often results in severely attenuated spring peak flow. In this reach there are many years (2013 was one) when prolific growth of nuisance algae (Cladophora) becomes apparent in the mainstem between Flint Creek and Rock Creek by late April or early May, well before the period of application would start on July 1. We recognize the inherent difficulty of using a standard (total N or P) that is highly sediment-correlated, but this is one example of an area that will likely not improve without a longer period of nutrient discharge control.

Variances

With respect to the proposed variance procedure, we recognize that variances are necessary based on SB 95 and SB 367, now codified in 75-5-313 MCA. We have the following comments on Department Circular DEQ-12B, *Nutrient Standards Variances*:

With respect to the general variance and end-of-pipe treatment requirements, we note that there are possible low cost alternatives to treatment lagoons, such as fed-batch reactors, that would provide better treatment than lagoons. While there is obviously a cost associated with this, it should be affordable to at least some communities that currently use lagoons. In the triennial reviews, DEQ should carefully consider currently available low-cost technologies that are more effective than lagoons.

With respect to wastewater facility optimization studies conducted as a requirement of a general variance, we question why the study would be done after the variance is issued, instead of as a prior condition of receiving the variance. We believe that a facility should be required to optimize as described in Section 2.1 before the variance is granted. At the very least, optimization analysis should occur concurrently with the variance application.

One of our overall concerns about the variance process is that DEQ will need to spend large amounts of time on variance requests and triennial variance reviews. The triennial reviews are critical for advancing water quality, and must not simply be perfunctory. How does DEQ plan to accommodate this workload with current resources?

Apart from issuing variances, DEQ could take a more proactive approach toward helping municipalities upgrade their systems by providing timely education and information to help them meet standards. As an example scenario, a financially-strapped small municipality is more likely to hire a sub-par consultant to receive sub-par advice on how to upgrade their system. DEQ could help communities avoid this by at least maintaining a current list (with references) of best available technologies for a range of plant sizes and costs, or otherwise facilitate better communication on best practices for plant optimization and upgrade, or on alternative methods of meeting standards such as land application.

Nonsignificance Criteria

We propose the following change to ARM §17.30.715(1). We strongly urge the deletion of the language "inorganic nitrogen, or inorganic phosphorus" from subsection (c) of the proposed ARM §17.30.715(1). While we do not believe this language would have the legal effect suggested by the Department in the December rule notice – i.e. allowing nondegradation review of new nutrient discharges under the old narrative standard – this phrase adds needless confusion to the rule.

As amended by the rule, ARM §17.30.715(1) would read as follows:

17.30.715 CRITERIA FOR DETERMINING NONSIGNIFICANT CHANGES IN WATER QUALITY

(1) The following criteria will be used to determine whether certain activities or classes of activities will result in nonsignificant changes in existing water quality due to their low potential to affect human health or the environment. These criteria consider the quantity and strength of the pollutant, the length of time the changes will occur, and the character of the pollutant. Except as provided in (2) , changes in existing surface or ground water quality resulting from the activities that meet all the criteria listed below are nonsignificant, and are not required to undergo review under 75-5-303 , MCA:

. . .

(c) discharges containing toxic parameters, inorganic nitrogen, or inorganic phosphorous, except as specified in (1) (d) and (e) , which will not cause changes that equal or exceed the trigger values in Department Circular DEQ-7. Whenever the change exceeds the trigger value, the change is not significant if the resulting concentration outside of a mixing zone designated by the department does not exceed 15% of the lowest applicable standard;

(d) changes in the concentration of nitrate in ground water which will not cause degradation of surface water if the sum of the predicted concentrations of nitrate at the boundary of any applicable mixing zone will not exceed the following values:

(i) 7.5 mg/L for nitrate sources other than domestic sewage;

(ii) 5.0 mg/L for domestic sewage effluent discharged from a conventional septic system;

(iii) 7.5 mg/L for domestic sewage effluent discharged from a septic system using level two treatment, as defined in ARM 17.30.702; or

(iv) 7.5 mg/L for domestic sewage effluent discharged from a conventional septic system in areas where the groundwater nitrate level exceeds 5.0 mg/L primarily from sources other than human waste.

For purposes of this subsection (d) , the word "nitrate" means nitrate as nitrogen; and

(e) changes in concentration of total inorganic phosphorus in ground water if water quality protection practices approved by the department have been fully implemented and if an evaluation of the phosphorus adsorptive capacity of the soils in the area of the activity indicates that phosphorus will be removed for a period of 50 years prior to a discharge to any surface waters;

(f) changes in the quality of water for any harmful parameter, including parameters listed in Department Circular DEQ-12, for which water quality standards have been adopted other than carcinogenic, bioconcentrating, or toxic parameters, in either surface or ground water, if the changes outside of a mixing zone designated by the department are less than 10% of the applicable standard and the existing water quality level is less than 40% of the standard;

(g) changes in the quality of water for any parameter for which there are only narrative water quality standards if the changes will not have a measurable effect on any existing or anticipated use or cause measurable changes in aquatic life or ecological integrity.

See December 2013 Rule Notice at 13-14. Under the proposed rule, since new or increased discharges to surface water containing nitrogen and/or phosphorous would potentially affect the eutrophication of those waters, they would be subject to the eutrophication-based nitrogen and phosphorus standards of DEQ-12A. Therefore, nondegradation review would take place under subsection (f), *supra*, and the requisite 10% and 40% thresholds would be calculated based on the numeric nitrogen and phosphorous standards. This is consistent with the Department's statement of the basic purpose of the rule:

The proposed amendments to ARM 17.30.715 will allow the department to calculate nonsignificant changes in water quality for the base numeric nutrient standards in DEQ-12A. If adopted by the board, base numeric nutrient standards will preclude the need to use the narrative standards at ARM 17.30.637(1)(e) to interpret eutrophication-based water quality impacts from nutrients.

Id. It is likewise consistent with the Department's rationale that "The proposed deletion of "or nutrients," in (1)(c), corresponds with the retaining of toxic-level nitrogen compounds in DEQ-7 and the relocation of eutrophication-based nitrogen and phosphorus standards to DEQ-12A."

In the December 2013 rule notice, however, the Department has proposed replacing the term "or nutrients" in subsection (c) with the almost-synonymous phrase "inorganic nitrogen or phosphorous," effectively re-inserting the phrase "or nutrients" into that paragraph. In addition the Department has added subsequent language to its rationale stating that, in direct contradiction to the above statements, it intends to continue to use *narrative* standards as the basis for nondegradation review:

. . . the term "or nutrients" in (1)(c) has been replaced with "or total inorganic phosphorus or total inorganic nitrogen," for the specific purpose of providing a nonsignificance threshold for nondegradation review of new dischargers, which are commonly subdivisions. This change allows the department to continue to carry out these reviews in the same manner as currently practiced, because DEQ-7 provides a trigger value for both of these inorganic compounds. ARM 17.30.715(1)(c) also provides: "Whenever the change exceeds the trigger value, the change is not significant if the resulting concentration outside of a mixing zone designated by the department does not exceed 15% of the lowest applicable standard." When these provisions become applicable, the "lowest applicable standard" would be the narrative standard contained in ARM 17.30.637(1)(e). As a result, the part of the nondegradation rules at ARM 17.30.715(1)(g) that relate to the narrative standards would apply.

This understanding of the legal effect of the proposed rule is incorrect. First, even if the rule deletes the term "or nutrients" from subsection (c) and replaces it with the phrase "total inorganic phosphorus or total inorganic nitrogen," the result will merely be that nutrients are regulated under two separate paragraphs of §715. New or increased discharges of nitrogen and phosphorus would remain subject to the plain language of subsection (f), since they contain "parameters listed in Department Circular DEQ-12, for which water quality standards have been adopted . . . ." Regardless of the outcome of review under subsection (c), review would still have to take place under the 10% and 40% thresholds of subsection (f), applying the numeric standards in DEQ-12.

Second, the Department's contention that under paragraph (c) the narrative standard at ARM 17.30.637(1)(e) would be the "lowest applicable standard" makes no sense. The "lowest applicable standard" for nitrogen and phosphorous would be the numeric standards in DEQ-12. One cannot calculate thresholds of 10% or 40% of a narrative standard. That concept is logically and semantically meaningless.

Third, as already noted, reviewing new or increased discharges under §17.30.637(1)(e) would be directly contrary to the fundamental purpose of the rule, as stated earlier in the same paragraph, to "preclude the need to use the narrative standards at ARM 17.30.637(1)(e) to interpret eutrophication-based water quality impacts from nutrients."

It has been our consistent understanding throughout this rulemaking process, based on numerous discussions at meetings of the Nutrient Working Group and a series of earlier rule notices, that non-significance determinations for new and increased nutrient discharges under the new rules would be made under ARM 17.30.715(f) rather than under the existing narrative standard. Indeed, we agree with the Department that a fundamental purpose of the proposed rule is to preclude the use of narrative standards and replace them with numeric standards, which can be applied more precisely and consistently, and which reflect the considerable scientific understanding the Department has developed regarding the effects of various concentrations of nitrogen and phosphorous on the state's surface waters. Our reading of the proposed rule – which the Department has shared in the past – is that review of new and increased discharges would take place under ARM §17.30.715(f), applying the numeric standards contained in Circular DEQ-12. If this is no longer how the Department intends to apply the proposed rule, we will be forced to reconsider our support for the proposed rule package. Such an application would not provide effective protection of state surface waters against degradation from nutrients, which is a primary purpose of adopting numeric nutrient standards. We request that the Department clarify their intent in this regard.

Economic Impact

Finally, we note that while there may be temporary economic impact in some communities as a result of implementing nutrient standards in the short term, we also stand to lose economically in the long term if nutrient standards are NOT adopted. Montana is known nation-wide for clean, beautiful rivers that support healthy fisheries. Our outstanding rivers are an important quality-of-life reason why companies choose to locate in Montana, and they're a large part of the lure that drew 11 million travelers to spend $3.2 billion dollars that supported 43,000 jobs in Montana in 2013. Yet, one of the most frequent questions we're asked from locals and visitors alike is "what causes that slimy green algae in the river and what are you doing about it?" We sincerely hope that this rule package will make that question a thing of the past.

Best regards,

Karen Knudsen
Executive Director

karen@clarkfork.org

Christine Brick
Science Director

chris@clarkfork.org

From: Laidlaw, Tina
To: Greeley, Carrie
Cc: Mathieus, George; North, John; Suplee, Mike; Perkins, Erin
Subject: EPA"s Comments on New Rule 1 and Circular DEQ-12B
Date: Tuesday, April 01, 2014 4:15:59 PM

Carrie,

Attached are EPA's comments (in pdf) on MDEQ's proposed nutrient rules (New Rule 1, Circular DEQ-12B). The Word document contains editorial suggestions (using track changes) to Montana's Numeric Nutrient Standards Implementation Guidance.

Thank you for the opportunity to comment.

Tina Laidlaw

Tina Laidlaw
U.S. EPA, Montana Office
10 West 15th St., Suite 3200
Helena, MT 59626
phone: (406) 457-5016

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 8
1595 Wynkoop Street
DENVER, CO 80202-1129
Phone 800-227-8917
http://www.epa.gov/region08

APR 1 2014

Ref: EPR-EP

Tracy Stone-Manning, Director
Montana Department of Environmental Quality
P.O. Box 200901
Helena, MT 59620-0901

Re: EPA Comments on Montana's Nutrient Proposals (New Rule 1 Nutrient Standards Variances; Circular DEQ-12B; and Montana's Numeric Nutrient Standards Implementation Guidance)

Dear Ms. Stone-Manning:

This letter provides the comments of the Environmental Protection Agency (EPA) Region 8 on Montana Department of Environmental Quality's (MDEQ) draft nutrient rules contained in: 1) New Rule 1 Nutrient Standards Variances; 2) Circular DEQ-12B; and 3) Montana's Numeric Nutrient Standards Implementation Guidance (Version 1.3).

MDEQ has spent the last decade developing the scientific rationale behind the proposed numeric nutrient criteria for wadeable streams to ensure they are protective of designated uses. MDEQ recognized that meeting the protective criteria could be challenging for dischargers, initiating a stakeholder workgroup to develop implementation tools that would allow dischargers to make incremental progress towards achieving the stringent criteria. As described in the following comments, the Agency is supportive of MDEQ's approach to setting water quality standards for nutrients for the State's rivers and streams, including the adoption of protective numeric nutrient criteria and the accompanying variance regulations. The EPA has worked collaboratively with the State to ensure that not only are MDEQ's criteria protective of applicable designated uses and based on sound scientific rationale, but also that the State's general and individual variance approaches are consistent with the Clean Water Act and the EPA's implementing regulations. As a general matter, the EPA supports the use of variances, as appropriate and consistent with 40 CFR §131.10, to provide time to meet designated uses and associated criteria in certain situations. MDEQ's variance approaches will allow the State and its stakeholders time to implement a phased approach to improve water quality, while retaining the currently applicable designated uses as the long-term goal for the State's rivers and streams. The EPA specifically supports the use of multiple discharger variances[1], similar to MT's general variance provision, by States and authorized tribes that want to find ways to improve the efficiency of both their WQS adoption and the EPA's review and approval process.

Please note that the positions described in our comments, regarding both existing and proposed water quality standards, are preliminary in nature and should not be interpreted as the final EPA decisions under Section 303(c) of the Clean Water Act (CWA).

The EPA looks forward to discussing any outstanding issues or concerns as the rulemaking process continues. We greatly appreciate the years of hard work by MDEQ and its considerable expertise on this topic. Our detailed comments are summarized below.

EPA COMMENTS

1) Limits of Technology-Based Variances. Section 3 of New Rule I Nutrient Standards Variances (New Rule I) authorizes individual variances if attainment of the criteria is "precluded due to economic impacts or limits of technology, or both."

Under the EPA's water quality standards regulation, adoption of variances may be granted if it can be demonstrated based on site-specific facts and circumstances that the otherwise applicable designated use and criterion or criteria are not feasible to attain during a certain temporary time frame. 40 CFR §131.10(g) sets forth the limited factors that may be used to justify variances. While none of the EPA's 131.10(g) factors include the phrase "limits of technology," such technology limits may be relevant to a demonstration provided under 40 C.F.R. §131.10(g) where water quality-based controls would "result in substantial and widespread economic and social impacts" or if it can be demonstrated that "human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place."

With respect to each of the factors MDEQ has proposed, there may be site-specific circumstances in Montana where it would be reasonable for the Department to consider adoption of discharger-specific individual variances provided the demonstration also shows that a 40 CFR §131.10(g) factor has been met. The decision to issue such an individual variance can only be made by completing a rulemaking to revise the WQS for an individual segment based on review of site-specific information. Each individual variance will be a Montana WQS rule change that must be submitted to the EPA for review and approval pursuant to 40 CFR §131.20(c).

---

[1] EPA-820-F-13-012. Discharger-specific variances on a broader scale: Developing credible rationales for variances that apply to multiple dischargers. March 2013.

2) <u>Variance Limits Reflective of the Highest Attainable Condition</u>. Department Circular DEQ-12B (DEQ-12B) establishes the following variance limits that apply through May 31, 2016:

Table 12B-1. General variance end-of-pipe treatment requirements per §MCA 75-5 -313(5)(b), through May 2016.

Discharger Category	Monthly Average Total P (μg/L)	Monthly Average Total N (μg/L)
≥ 1.0 million gallons per day	1,000	10,000
< 1.0 million gallons per day	2,000	15,000
Lagoons not designed to actively remove nutrients	Maintain current performance	Maintain current performance

MDEQ has documented that the limits proposed in Table 12B-1 represent "starting point concentrations" that "may not be the lowest concentrations that could economically be achieved by every discharger today."[2] This perspective is further supported by the nutrient reduction steps outlined in MDEQ's Numeric Nutrient Standards Implementation Guidance that suggest further nutrient reductions are feasible. (Implementation Guidance, page 7).

The EPA's position is that variances should specify the interim use(s) and water quality criteria that reflect the highest attainable effluent conditions that require the point source discharge concentration and load to be minimized to the maximum extent attainable so that the highest degree of protection for use classification is achieved. This approach is consistent with the "wherever attainable" caveat to the CWA §101(a)(2) goal. Where appropriate, compliance schedules to achieve the highest attainable effluent condition as soon as possible can be established in the permit.

The EPA's recently Proposed Water Quality Standard Regulatory Clarifications[3] specify two options for defining the highest attainable effluent condition in a variance:

> "a variance must specify (1) the highest attainable interim use and numeric criterion that will apply during the term of the variance or (2) an interim numeric effluent condition that reflects the highest attainable condition for a specific permittee(s) during the term of the variance."[4]

[2] Letter from Richard Opper, MDEQ Director to Jim Martin, EPA Region 8 Regional Administrator, 9 March 2011.
[3] 78 Fed. Reg. 54518, 54533 (Sept. 4, 2013).
[4] 78 Fed. Reg. 54518, 54533 (Sept. 4, 2013).

In its proposed regulations, MDEQ has included an initial set of "end of pipe treatment requirements" (see above) accompanied by an expiration date for the initial phase within the general variance. This expiration is appropriate given that the State statute authorizing the variance, MCA 75-5-313, sets forth end-of-pipe treatment requirements for only that time frame. As the expiration date approaches for the initial set of end-of-pipe treatment requirements to expire, the EPA fully expects MDEQ to readopt the general variance with the next set of phased end-of-pipe treatment requirements, reflecting the highest attainable effluent condition at that time. The EPA is committed to working collaboratively with the State during the general variance readoption process to ensure that at no time are eligible permittees left without coverage under the general variance. The EPA understands that MT's intention is to continue the general variance, as appropriate, until the State's waters attain the numeric nutrient criteria, for up to 20 years from initial adoption. The EPA is supportive of that approach.

3) Variances for New Dischargers. In the Implementation Guidance (middle of page 6), MDEQ defines the scope of the implementation provisions as:

> "The provisions for general, individual, and alternative variances in section 75-5-313, MCA, are available to all discharge permit holders and are not limited to dischargers under permit on the effective dates of MDEQ Circular DEQ-12A or MDEQ Circular DEQ-12B." [underline added]

The EPA's long-standing policy is that variances are authorized only where one of the factors for removing a designated use in 40 CFR §131.10(g) are met. Importantly, all six of the removal criteria are subject to the caveat that only a designated use that is not an existing use may be removed. 40 CFR §131.10(g) specifies that "states may remove a designated use which is not an *existing* use." 40 CFR §131.3(e) defines existing uses as "those uses actually attained in the water body on or after November 28, 1975, whether or not they are included in the water quality standards."

Variances are not authorized in situations where the site-specific facts indicate that existing uses would be impacted. However, the EPA recognizes that there may be situations where it would be possible for a discharger to demonstrate that the variance protects the existing use while providing temporary relief from meeting the underlying water quality standard. In these cases, a variance may be justified.

4) Nutrient Reduction Steps. Section 2 (page 7) of the Implementation Guidance establishes a set of nutrient reduction steps for the three categories of dischargers. The guidance states that:

> "the Department will only supersede the reduction steps defined here if substantial cost reductions for existing technology have occurred, or technological innovations have

> allowed for nutrient reductions well beyond the defined steps and those technologies can be readily implemented on wastewater facilities in Montana". [underline added]

The EPA's position is that variance limits reflect the highest degree of pollutant removal attainable. Because those limits have not yet been determined for the three categories of dischargers, we recommend MDEQ strike this sentence from the final Implementation Guidance. In addition, because plant performance may vary greatly and to allow maximum flexibility to achieve the final limits, MDEQ may want to consider simply establishing the final interim variance limit that would apply for each category of discharger instead of outlining specific nutrient reduction steps facilities would be required to meet each permit cycle. The duration of compliance schedules to meet the final limits can be customized based on discharger-specific information.

5) Economic Analysis Exemption for Limits of Technology-Based Variances. MDEQ's Implementation Guidance exempts dischargers applying for an individual variance based on limits of technology from preparing an economic analysis to demonstrate economic hardship. This language is found on page 8 and repeated on page 14:

> "Permittees applying for an individual variance based on discharging at the limits of technology do not have to prepare the economic analysis presented below in **Section 3.1.1**. Rather, they should demonstrate to the Department that the waste treatment system they are proposing can achieve, at a minimum, the nitrogen and phosphorus concentrations shown in **Section 1.2** of this document, and that achieving those concentrations still will not enable them to attain the base numeric nutrient standards at a 14Q5 flow." (middle of page 8)

Because each individual variance will be a Montana WQS rule change that must be submitted to the EPA for review and approval pursuant to 40 CFR §131.20(c), the variance application will need to demonstrate consistency with 40 CFR §131.10(g). As noted in Comment #1, although none of the EPA's 40 CFR §131.10(g) factors include the phrase "limits of technology," such technology limits may be relevant to a demonstration provided under 40 CFR §131.10(g)(6) where water quality based controls would "result in substantial and widespread economic and social impacts."

Dischargers should use the most appropriate 40 CFR §131.10(g) factor to demonstrate they meet the requirements to be eligible for a variance. The guidance language exempting permittees from the federal requirement to provide this demonstration, even in situations where the most appropriate factor is 40 CFR §131.10(g)(6), could result in variances that may not comply with the EPA's regulations.

To address this concern, we recommend MDEQ consider the following modification to the language found on pages 8 and 14.

EPA-Recommended Language:

"Permittees applying for an individual variance based on discharging at the limits of technology ~~do~~ may not have to prepare the economic analysis. Permittees must demonstrate, based on one of the factors at 40 CFR§131.10(g) that it is infeasible to meet its water quality-based effluent limits based on the applicable designated use and associated criteria."

6) Alternative Variances. MCA 75-5-313(10)(a) and (b) authorize MDEQ to issue an "alternative" variance in situations where the discharger is an "insignificant" source of the nutrient load. MDEQ's Implementation Guidance provides additional detail (pages 16-17) on approaches (e.g., modeling) that can be used to evaluate whether the discharger nutrient contribution is "insignificant" and eligible for an alternative variance.

As noted in the EPA's 2011 letter to MDEQ[5], none of the 40 CFR §131.10(g) factors authorize variances based on *de minimus* considerations; therefore, a variance based on a *de minimus* demonstration would not comply with the EPA's regulations. Instead, *de minimus* situations may be addressed through the development of total maximum daily load (TMDL) allocations pursuant to CWA §303(d). This approach is described in New Rule Section 8 and addresses situations where a TMDL has been approved and the discharger meets the waste load allocation.

7) Detailed Comments on the Implementation Guidance: In addition to the comments summarized in this letter, the EPA has provided a number of edits and formatting changes to the Implementation Guidance using track changes. These comments are intended to help clarify the information in the document or improve readability. The EPA considers these revisions to be non-substantive and intended simply as editorial suggestions.

[5] Letter from Jim Martin, EPA Region 8 Regional Administrator to Richard Opper, MDEQ Director, 16 March 2011.

Conclusion

We hope our comments are helpful to MDEQ and the parties to this rulemaking. We appreciate MDEQ's efforts to address issues of concern to the EPA. If there are questions concerning our comments, please contact Tina Laidlaw (406-457-5016). We look forward to working with the parties to address these issues.

Sincerely,

Sandra Spence, Chief
Water Quality Unit

DRAFT 1.3
*Base Numeric Nutrient Standards Implementation Guidance**

***Retitled from earlier drafts to reflect the more encompassing nature of this document. Previous draft number was 7.3**

December 2013

Prepared by:
Water Quality Planning Bureau, Water Quality Standards Section
Montana Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

WQPBWQSTR-002

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ACRONYMS

Acronym	Definition
DEQ	Department of Environmental Quality (Montana)
EPA	Environmental Protection Agency (US)
LMI	Low to Moderate Income
MCA	Montana Code Annotated
MHI	Median Household Income

1.0 INTRODUCTION

This document was developed through the collective efforts of the Nutrient Work Group and the Department. It provides guidance pertaining to the implementation of Montana's base numeric nutrient standards and variances from those standards. The remaining sections address the following topics:

Section 2.0: For permittees operating under a general nutrient standards variance, this section provides the defined effluent limits (i.e., nutrient reduction steps) to be met over several permit cycles of the general variance.

Section 3.0: Provides guidance for the development of Iindividual Nnutrient Sstandards Vvariances for public- and private-sector entities, based on economic factors and the limits of technology;.

Section 4.0: Provides guidance for the development of Individual alternative Nnutrient Sstandards Vvariances for public- and private-sector entities, based on economic factors;.

Section 5.0: Outlines a streamlined approach for developing site-specific numeric nutrient criteria for streams or rivers where full biological support is demonstrated but where the existing nutrient concentrations exceed applicable base numeric nutrient standards; .and

Section 6.0: Provides a detailed, data-intensive modeling approach for developing site-specific numeric nutrient criteria. This approach lends itself to the development of individual variances for dischargers.

1.1 SCOPE

The provisions for general, individual, and alternative variances in section 75-5-313, MCA, are available to all discharge permit holders and are not limited to dischargers under permit on the effective dates of DEQ Circular DEQ-12A or DEQ Circular DEQ-12B.

1.1 2 DEFINITIONS

1. **Limits of technology** means wastewater treatment processes for the removal of nitrogen and phosphorus compounds from wastewater that meets the more stringent of the following: (a) ability tocan consistently achieve a concentration of 70 µg TP/L and 4,000 µg TN/L, or (b) the best demonstrated control technology, processes, or operating methods available at the time the Department evaluates a permittee's application for a limits of technology variance.
2. **Pollution control project** means an upgrade to a wastewater treatment facility and all directly relevant infrastructure.

2.0 DEFINED NUTRIENT-REDUCTION STEPS FOR PERMITTEES OPERATING UNDER A GENERAL NUTRIENT STANDARDS VARIANCE

The Department and the Nutrient Work Group developed a series of defined nutrient-reduction steps to be taken over time and that are specific to recipients of general nutrient standards variances. Per §75-5-

313 [8], MCA, general nutrient standards variance may be established for no more than 20 years. The intent of establishing nutrient reduction steps upfront for most of the 20 year period is to provide permittees regulatory certainty well out into the future. This in turn allows for better facility planning and financing. State law still requires the Department to review triennially the general variance concentrations, and to lower them conforming with technological advancements and improvements in cost (§75-5-313 [7][b], MCA). However, the Department will only supersede the reduction steps defined here if substantial cost reductions for existing technology have occurred, or technological innovations have allowed for nutrient reductions well beyond the defined steps and those technologies can be readily implemented on wastewater facilities in Montana.

For the purposes of permit development, the values provided below apply to recipients of general nutrient standards variances and the concentrations should be viewed as ~~long-term~~monthly averages applicable during the time period the base numeric nutrient standards are in effect.

1. For facilities > 1 million gallons per day:

A. By 2016 (or first receipt of general nutrient standards variance): 10 mg TN/L, 1.0 mg TP/L
B. Next permit cycle (5 year later): 8 mg TN/L, 0.8 mg TP/L
C. Next permit cycle (5 years later): 8 mg TN/L, 0.~~3~~ 5 mg TP/L
D. Next permit cycle (5 years later): ~~5 mg TN/L, 0.15 mg TP/L~~Under Development

2. For facilities < 1 million gallons per day:

A. By 2016 (or first receipt of general nutrient standards variance): 15 mg TN/L, 2.0 mg TP/L
B. Next permit cycle (5 year later): 12 mg TN/L, 2.0 mg TP/L
C. Next permit cycle (5 years later): 10 mg TN/L, 1.0 mg TP/L
D. Next permit cycle (5 years later): 8 mg TN/L, 0.8 mg TP/L

3. For lagoons not designed to actively remove nutrients:

A. By 2016 (or first receipt of general nutrient standards variance): Maintain current lagoon performance and commence nutrient monitoring in the effluent
B. Next permit cycles (5 years later): Implement BMPs identified during optimization study

3.0 GUIDANCE PERTAINING TO ~~T~~THE EVALUATION PROCESS FOR INDIVIDUAL VARIANCES~~: PUBLIC-SECTOR PERMITTEES~~

The following sections provide guidance on applying for an individual variance. **Section 3.1** applies to the public sector, while **Section 3.2** applies to the private sector.

3.1 PUBLIC-SECTOR PERMITTEES

Montana law allows for the granting of nutrient standards variances based on the specific economic and financial conditions of a permittee (§75-5-313 [1], MCA). These variances, referred to as individual nutrient standards variances ("individual variances"), may be granted on a case-by-case basis because the attainment of the base numeric nutrient standards is precluded due to economic impacts, limits of technology, or both. Individual variances may only be granted to a permittee after the permittee has made a demonstration to the Department that adverse, significant economic impacts would occur, the limits of technology have been reached, or both, and that there are no reasonable alternatives to discharging into state waters. The processes by which the demonstration is made are provided here, and were developed in conjunction with Montana Nutrient Work Group.

Methods outlined below in **Section 3.1.1** are Montana's modifications to methods presented in U.S. Environmental Protection Agency (1995) and pertain to the economic impacts rationale for an individual variance. If adverse, substantial and widespread economic impacts to a community trying to comply with base numeric nutrient standards are can be demonstrated, the facility interim effluent limit will be determined via a sliding scale as proposed by EPA in its September 10, 2010 memo "EPA Guidance on Variances".

Permittees applying for an individual variance based on discharging at the limits of technology do not have to prepare the economic analysis presented below in **Section 3.1.1**. Rather, they should demonstrate to the Department that the waste treatment system they are proposing can achieve, at a minimum, the nitrogen and phosphorus concentrations shown in **Section 1.2** of this document, and that achieving those concentrations still will not enable them to attain the base numeric nutrient standards at a 14Q5 flow. Various factors will have a bearing on the final effluent concentrations approved by the Department for individual variances discussed in this paragraph.

3.1.1 Substantial and Widespread Economic Impacts: Process Overview

In taking this approach, theThe Department has assumed that most permittees who cannot comply with the base numeric nutrient standards (DEQ-12, Part A) would pursue a general variance (DEQ-12, Part B). Therefore, individual variances discussed here are generally for permittees for whom significant economic impacts would occur even at the general variance treatment levels. As noted, above, the Department will assess economic impact using a modified version of EPA's economic-impact guidance. For communities with secondary scores (discussed further below) of 1.5 or lower, the cost cap for the upgrade would be set at 1.0% or lower of the median household income (MHI) for a town, including existing wastewater fees. If the cost cap were below existing wastewater rates, then no further action would be required. Higher Secondary scores would lead to a higher MHI cost cap. A small flow chart of the overall process is as follows:

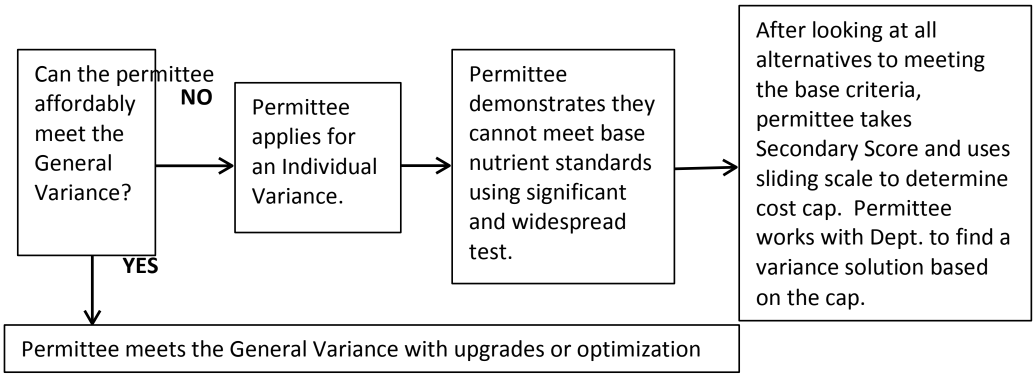
Deleted: upgrade cost-cap

Comment [TL1]: This is a EPA mailcode, not a reference number.

Deleted: reference No. 8EPR-EP

Comment [TL2]: Not relevant since the approach relies on the sliding scale.

Deleted: The Nutrient Work Group has indicated that 1.0% of MHI is an acceptable cost cap for a community to expend on wastewater treatment where economic hardship due to meeting base numeric nutrient standards has been demonstrated.



~~3.1 Substantial and Widespread Economic Impacts: Process Overview~~

The following is an overview of the steps required to carry out a substantial and widespread economic analysis for a public-sector permittee. **The evaluation can be undertaken directly in an Excel spreadsheet template which contains instructions. The template is called "PublicEntity_Worksheet_EPACostModel_2013.xlsx" and is available from the Department.**

Step 1: Verify project costs that would occur from meeting the base numeric nutrient ~~criteria~~ standards and calculate the annual cost of the new pollution control project.

Step 2: Calculate total annualized pollution control cost per household including existing wastewater fees and the new pollution control project (manifested as an increase in the household wastewater bill).

Steps 3-5: The Substantial Test

Step 3: Calculate and evaluate the Municipal Preliminary Screener score based on the new wastewater fees and the town's Median Household Income. This step identifies communities that can readily pay for the pollution control project vs. those that cannot.

Note: If the public entity passes a significant portion of the pollution control costs along to private facilities or firms, then the review procedures outlined in Chapter 3 of EPA (1995) for 'Private Entities' should also be consulted to determine the impact on the private entities.

Step 4: Calculate the Secondary Test to get a secondary score. This measurement incorporates a characterization of the socio-economic and financial well-being of households in the community where the wastewater plant is located. It comprises five evaluation parameters which are then compared against state averages for a score. The scores of the five parameters are averaged to provide the secondary test score for a given community. A secondary score can range from 1.0 to 3.0. 3.0 is a strong score and 1.0 is a weak score.

Note: The Secondary Score is based on the assumption that the ability of a community to finance a project may be dependent upon existing household financial conditions within that community.

Step 5: Assess where the community falls in the substantial impacts matrix. This matrix evaluates whether or not a given community is expected to incur substantial economic impacts due to the implementation of the pollution control costs. If the applicant can demonstrate substantial impacts, then the applicant moves on to the widespread test. If the applicant cannot demonstrate substantial impacts, then they will not perform the widespread test; they will be required to meet the base numeric nutrient standards.

Note: The evaluation of substantial impacts resulting from compliance with base numeric nutrient standards includes two elements; (1) financial impacts to the public entity as measured in **Step 3** (reflected in increased household wastewater fees), and (2) current socio-economic conditions of the community as measured in **Step 4**. Governments have the authority to levy taxes and distribute pollution control costs among households and businesses according to the tax base. Similarly, sewage authorities charge for services, and thus can recover pollution control costs through user's fees. In both cases, a substantial impact will usually affect the wider community. Whether or not the community faces substantial impacts depends on both the cost of the pollution control and the general financial and economic health of the community.

Step 6: The Widespread Test

Step 6: If impacts of meeting base numeric nutrient ~~criteria~~ standards are expected to be substantial, then the applicant goes on to demonstrate whether or not the impacts are expected to be widespread. The Widespread test consists of questions that ask the permittee about current economic, social and population trends in the affected area (usually the committee and possibly outlying areas tied to the community). The permittee is then asked to estimate the effects of higher wastewater costs on each of these trends. Further optional questions are asked about the effects of higher wastewater costs on things like city debt limits, improved water quality, future development patterns, and other factors that the applicant may want to add.

Note: Estimated changes in socio-economic indicators of the community and other geographical areas tied to the community as a result of pollution control costs ~~and~~ will be used to determine whether widespread impacts would occur.

Step 7: Final Determination of Substantial and Widespread Economic Impacts

Step 7: If widespread impacts are also demonstrated, then a permittee is eligible for an individual variance after having demonstrated to the Department that they considered alternatives to discharging (including but not limited to trading, land application, and permit compliance schedules). If widespread impacts have not been demonstrated, then the permittee is not eligible for an individual variance).

3.1.2 Completing the Substantial and Widespread Assessment Spreadsheet

Deleted: or may request a general variance if they can discharge at the general variance concentrations defined in Department Circular DEQ-12, ~~Part~~ B.

Deleted: (however, the permittee may still receive a general variance if they can comply with the end-of-pipe treatment requirements thereof

Comment [T3]: This section seems out of place here the remedy section hasn't yet been covered. I'm wondering if this should be an appendix? Or go at the back of the entire section.?

Detailed steps for completing the substantial and widespread cost assessment are found in the spreadsheet template "PublicEntity_Worksheet_EPACostModel_2013.xlsx" available from the Department and on the Nutrient Workgroup website. Readers should refer to that spreadsheet, as it is self-explanatory and instructions are found throughout. Below are a few additional details which may help clarify some of the steps:

1. Start at the far left tab of the spreadsheet ("Instructions [Steps to be Taken]") and review the instructions. They are the same steps outlined in **Section ~~2~~3.1.1** above, but in more detail. Proceed to subsequent tabs to the right, making sure not to skip any of worksheets A through F.
2. Summarize the project on Worksheet A.
3. Detail the costs of the project on Worksheet B.
4. Calculated the annual cost per household of existing and expected new water treatment costs on Worksheet C.
5. On Worksheet D, carefully read the text in blue and compare it to the results from the MHI test and the community's Low to Moderate Income (LMI) level. Based on this screener, the evaluation will either terminate (i.e., it has been shown that the water pollution control is clearly affordable), or will continue to the secondary tests on the next tab which is Worksheet E[1].
6. On Worksheet E, note the linkages to websites and phone numbers where the information requested can be obtained. Then use this information to fill in Worksheet F where a secondary score is calculated.
7. The next tab, 'Substantial Impacts Matrix', shows if the community has demonstrated substantial impacts (or not). Those that have clearly demonstrated substantial impacts as well as those that are 'borderline' move on to the widespread tests.
8. On the 'DEQ Widespread Criteria' tab, complete the four descriptive questions. Then, complete the six primary questions and determine the outcome as to whether impacts are widespread. If still unclear, complete the additional secondary questions and again evaluate.
9. In order to be eligible for an individual variance, both substantial and widespread tests must be satisfied.
10. If substantial and widespread impacts are demonstrated, then the permittee moves on to the next tab, Worksheet I, Remedy. In this step, the permittee examines and reports whether there are "reasonable alternatives" to the individual variance that "preclude" the need for an individual variance. If not, then then the cost the permittee will need to expend towards the pollution control project will be based on the sliding scale (see below). The cost cap is determined as a percentage of the community's MHI, and the key driver of the required cost cap is the Secondary Score.

The difference between the cost cap MHI from the sliding scale and what is currently being paid (also in MHI) is the additional money that can go towards the pollution control project. Once the amount of money available is determined, ~~DEQ~~ the Department and the applicant will look at both capital and O&M investments that could be used to meet an individual variance, given what money is available. Refer to **Section 3.1.3** below for more details on the remedy process.

[1] The Department appended the LMI test to EPA's Municipal Preliminary Screener at this step in the process. This was done in order to address communities in which the income distribution is skewed such that there is a large proportion of high- and low-income individuals, but less in the middle near the median household income. As modified, the test should assure that such communities will move on to the more detailed secondary tests.

3.1.3 The Remedy: Determining the Target Cost of the Pollution Control Project

If a permittee has demonstrated that substantial and widespread economic impacts would occur if they were to comply with the base numeric nutrient standards, and there are no reasonable alternatives to discharging (including trading, permit compliance schedules, general variances, alternative variances, or alternative effluent management loading reduction methods such as reuse, recharge, or land application), then the cost the permittee will need to expend towards the pollution control project will be based on a sliding scale (**Figure 3-1**). The cost cap is determined as a percentage of the community's MHI, and the key driver of the cost cap is the secondary test (secondary score) calculated in **step 4** of **Section 3.1.1.**

For example, a community has demonstrated that substantial and widespread economic impacts would occur from trying to comply with the base numeric nutrient standards, and there were no reasonable alternatives to discharging. If the permittee's average secondary score from the secondary tests was 1.5, then the annual cost cap for the pollution control project (including current wastewater fees) would be the dollar value approximately equal to 1.0% of the community's MHI at the time that the analysis was undertaken (see blue line, **Figure 3-1**). This 1.0% would include existing wastewater costs plus the new, hypothetical upgrades.

If this community was already paying ≥ 1.0% of community MHI for its wastewater bill, then no additional monies would be spent on capital or O&M costs (and no additional upgrades would occur). Still, additional improvements may still be expected. The facility's current discharge nutrient concentrations might become the basis of the community's individual variance but the community must first look at optimization options such as operator training and use all tools available within their cost cap to improve water quality. Once those are considered, the individual variance can be developed.

The difference between the cost cap MHI from the sliding scale and what is currently being paid in MHI is the additional money that can go towards the pollution control project. This amount could be zero in some cases, as in the example just given. This additional money is calculated for the whole town over 20 years (assumed life of the pollution control project) in order to see what the total amount of money available would be. The cost cap, which is given as a percentage of a community's MHI and determined by the 'sliding scale' in **Figure 3-1**, would translate to the final wastewater bill that the community would pay after the upgrade.

For example, a community with 10,000 households has a MHI of $40,000/year. The community's secondary score is 1.5 and therefore the sliding scale indicates that 1.0% MHI needs to be expended on the pollution control project. To receive the individual variance, the per-household wastewater bill for the community would need to become, on average, $400 per year ($33.33 per month), because $400 is 1.0% of MHI in that community. If the average household in this community currently has a wastewater bill that is $300 per year ($25.00 per month), then a bill increase of $100 per year per household on average would be warranted to reach $400 per year or 1% MHI. Multiplying $100/year in an increased wastewater bill by the number of households on the system (10,000) provides the total annual dollar value available to be expended towards construction, operations, and maintenance of the wastewater upgrade. In this hypothetical case, that amounts to $1 million (10,000 X $100) that could be spent per year on an upgrade project. The upgrade itself may be significantly more than $1 million in initial capital costs, but the annualized payback of capital costs plus O&M costs of the upgrade could not be more than $1 million per year. Annualizing $1 million per year over several years could allow for a substantial

Comment [T4]: Might be helpful to present this information as a table (i.e., similar to Shari's tables) to help people understand the concept).

upgrade of several million dollars. Again, if the current wastewater bill of this town was already $400 or higher, then no additional significant capital or O&M cost upgrade would be expected (i.e., no further significant system upgrade would be required).

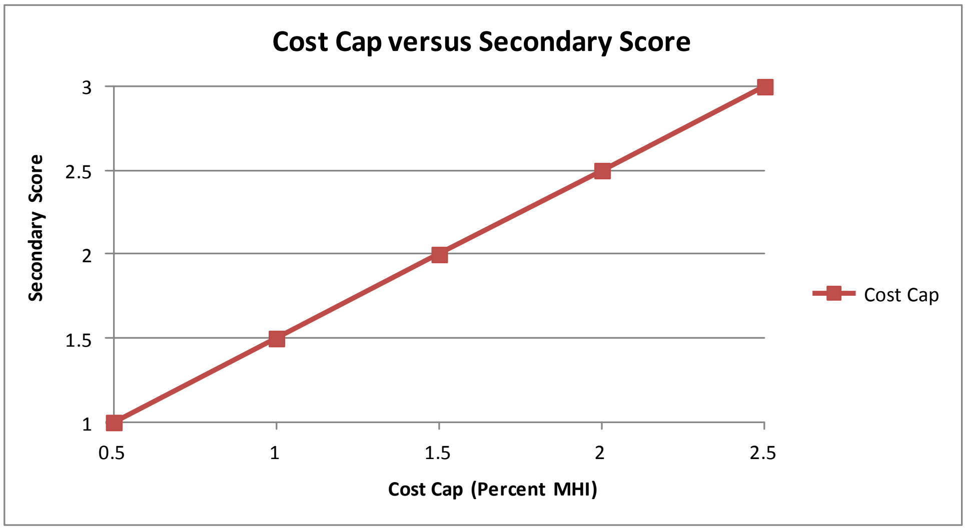


Figure 3-1. Sliding scale for determining cost cap based on a community's secondary score. The horizontal axis represents percentages of a community's median household income (MHI) that the community would be expected to expend towards the pollution control project as a function of the secondary score shown on the vertical axis.

Add a step here about "selecting/determining the remedy"

~~DEQ~~ The Department looks at the town's current treatment level (TN and TP) and current treatment technology, which informs (along with the additional money amount) what the next level of treatment should be. Once the amount of money available is determined, ~~DEQ~~ the Department and the applicant look at both capital and O&M investments that could be used to meet an individual variance, given what money is available. Staff from the Department will review the application and the remedy. The staff will generally include the Department's economist, an engineer from the Technical and Financial Assistance Bureau, staff from the Water Quality Standards Section, and staff from the Water Protection Bureau (i.e., permitting).

Comment [T5]: Perhaps this section should either be a subsection titled, "Selecting the remedy" or these 4 paragraphs could be moved to the beginning of the Remedy section and followed by examples. Might help clarify the examples and how they were established.

The WWTP applicant must propose a level of water treatment greater than what they are currently meeting. If a town is already at the cost cap, then they still must look at optimization options such as operator training and use all tools available within their cost cap which could lead to water quality improvement. The variance must be established as close to the underlying numeric criteria (or general variance) as possible to show both that the highest attainable use is being realized and that further incremental progress towards the underlying standard is occurring. ~~DEQ~~ The Department and the applicant will evaluate options and select the alternative that would result in the highest effluent condition that does not trigger substantial and widespread economic impacts. The decision process should include engineering costs, design, treatment effectiveness, etc. The decision regarding the pollution control project may also account for facility upgrades that do not directly improve water quality. For example, if $4 million is available over 20 years for a given community, but $2 million is needed for replacing delivery system piping over that 20 years, it may be the case that only $2 million are available to directly reduce nutrient concentrations in the effluent.

Finally, the final cost of the engineering project may not exactly match the dollar value associated with the percent MHI determined via **Figure 3-1** (i.e., the actual project cost could be somewhat lower or somewhat higher than the dollar value equivalent for the percent MHI of the community in question). Engineers should view the dollar value equivalent of the MHI derived from **Figure 3-1** as a target, to help select the most appropriate water pollution control solution for the community. In order to accommodate actual engineering costs for the project, the Department will provide flexibility around the dollar value arrived at via **Figure 3-1**, subject to final Department approval.

When the level of treatment required has been established and accepted by the Department, it will be adopted by the Department following the Department's formal rule making process and documented in Circular DEQ-12~~, Part~~ B.

~~4~~3.~~0~~ 2 ~~THE EVALUATION PROCESS FOR INDIVIDUAL VARIANCES:~~ PRIVATE-SECTOR PERMITTEES

Individual nutrient standards variances ("individual variances") may be granted to permit holders in the private sector, on a case-by-case basis, because (1) the attainment of the base numeric nutrient standards is precluded due to economic impacts, (2) treatment to the limits of technology still does not enable the permittee to attain the base numeric nutrient standards, or (3) both reasons (§75-5-313 [2], MCA). Individual variances may only be granted to a permittee after the permittee has made a demonstration to the Department that adverse, significant economic impacts would occur, limits of technology have been reached, or both, and that there are no reasonable alternatives to discharging into state waters.

Methods outlined below in **Section 3.2.1** pertain to the economic-impact rationale (the first case in the paragraph above) and are almost identical to those presented in U.S. Environmental Protection Agency (1995). If adverse substantial and widespread economic impacts to a private entity trying to comply with nutrient standards are demonstrated, the facility upgrade (cost cap) will be determined via approaches discussed in **Section ~~3~~3.2.3.**

Permittees applying for an individual variance based on discharging at the limits of technology do not have to prepare the economic analysis presented below in **Section 3.2.1**. Rather, they should demonstrate to the Department that the waste treatment system they are proposing can achieve, at a minimum, the nitrogen and phosphorus concentrations shown in **Section 1.2** of this document, and that achieving those concentrations still does not enable them to attain the base numeric nutrient standards at a 14Q5 flow. Various factors will have a bearing on the final effluent concentrations approved by the Department for individual variances discussed in this paragraph.

~~4~~3.2.1 Substantial and Widespread Economic Impacts: Process Overview

The following is an overview of the steps required to carry out a substantial and widespread economic analysis for a private-sector permittee. The evaluation can be undertaken directly in an Excel spreadsheet template which contains instructions~~ (see **Section 3.2**)~~. The template is called "PrivateEntity_Worksheet_EPACostModel_2012.xlsx" and is available from the Department.

Step 1: Verify Project Costs and Calculate the Annual Cost of the Pollution control project to the private entity.

Step 2: Substantial Test. Run a financial impact analysis on the private entity to assess the extent to which existing or planned activities and/or employment will be reduced as a result of meeting the water quality standards. The primary measure of whether substantial impact will occur to the private entity is profitability. The secondary measures include indicators of liquidity, solvency, and leverage.

Step 3: Widespread Test. If impacts on the private entity are expected to be substantial, then the applicant goes on to demonstrate whether they are also expected to be **widespread** to the defined study area.

Note: Estimated changes in socio-economic indicators in a defined area as a result of the additional pollution costs will be used to determine whether widespread impacts would occur.

Step 4: Final Determination of Substantial and Widespread Economic Impacts. If both substantial and widespread impacts are demonstrated, then a permittee is eligible for an individual variance after having demonstrated to the Department that they considered alternatives to discharging (including but not limited to trading, land application, and permit compliance schedules). If widespread impacts have not been demonstrated, then the permittee is not eligible for an individual variance (however, the permittee may still receive a general variance if they can comply with the end-of-pipe treatment requirements thereof).

43.2.2 Completing the Substantial and Widespread Assessment Spreadsheet

Detailed steps for completing the substantial and widespread cost assessment are found in the spreadsheet template "PrivateEntity_Worksheet_EPACostModel_2012.xlsx" (available from the Department). Readers should refer to that spreadsheet, as it is self--explanatory and instructions are found throughout. Detailed steps for private sector entities are also found in Chapter 3 of U.S. Environmental Protection Agency (1995). Below are a few additional details which may help clarify some of the steps:

1. Start at the far left tab of the spreadsheet ("Instructions [Steps to Take]") and review the instructions. They are the same steps outlined in **Section 3.2.1** above. Proceed to subsequent tabs to the right, making sure not to skip any of the worksheets.
2. Summarize the project on Worksheet A.
3. There are no worksheets B through F on the private test.
4. The next worksheet is G where one details the costs of the project.
5. In the next tab, carefully read the 'Substantial Impact Instructions'.
6. In worksheets H through L, the four main substantial tests are presented. For these tests, profit and solvency ratios are calculated with and without the additional compliance costs (taking into consideration the entity's ability to increase its prices to cover part or all of the costs). Comparing these ratios to each other and to industry benchmarks provides a measure of the impact on the entity of additional wastewater costs. For profit and solvency, the main question is how these will be affected by additional pollution control costs. The Liquidity and leverage

measures look at how a firm is doing right now financially, and how much additional financial burden they could take on.

7. In the Tab entitled "Substan.Impacts_Determined", instruction is given as to how to interpret the results from the 'Substantial' tests in worksheets H through L.
8. If a 'Substantial ' finding is made, then proceed on to the next tab. If it is not made, then a the variance based on evaluations in this sub-section will not be given.
9. On the 'DEQ Widespread Criteria' tab, complete the descriptive questions. Then, complete the primary questions and determine the outcome as to whether impacts are widespread. If still unclear, complete the secondary questions and again evaluate.
10. In order to be eligible for an individual variance, both substantial and widespread tests must be satisfied.
11. If both substantial and widespread impacts are demonstrated from additional pollution control costs, see **Section 3.2.3** below.

43.32.3 Cost-cap (or other solution) for Private Entities

Comment [T6]: Recommend using the same language used for public sector and call it "remedy" not cost cap.

U.S. Environmental Protection Agency (1995) provides very little guidance as to what financial expenditure should be made towards water pollution control when a private firm has demonstrated substantial and widespread impacts would occur if they complied with the standards. U.S. Environmental Protection Agency (1995) only states that "…if substantial and widespread economic and social impacts have been demonstrated, then the discharger will not have to meet the water quality standards. The discharger will, however, be expected to undertake some additional pollution control."

In cases where substantial and widespread economic impact has been demonstrated per methods outlined here in **Section 3.02**, the Department expects that in most cases the discharger (and their engineers) will propose to the Department some level of effluent improvement beyond that which they are currently doing, but less stringent that the general variances concentrations (which are now in statute at §75-5-313, MCA, and which will later be adopted as Department rules in 2016). A likely scenario would be that the discharger could implement a treatment technology one level less sophisticated than that required to meet the general variance concentrations. Basic definitions for different treatment levels are found in Falk et al. (2011); for example, through 2016 the general variance requirement for dischargers > 1 MGD corresponds to treatment level 2 in Falk et al. (2011). When the discharger and the Department have come to agreement on the level of treatment required, the treatment levels will be adopted by the Department following the Department's formal rule making process, and documented in Circular DEQ-12, Part B.

4.0 GUIDANCE PERTAINING TO ALTERNATIVE NUTRIENT STANDARDS VARIANCES

Statute provides for alternative nutrient standards variances ("alternative variances") in addition to general and individual variances. A permittee may request an alternative variance if the permittee demonstrates to the Department that achieving the nutrient concentrations established for an individual or general nutrient standards variance would not result in a significant reduction of instream nutrient loading (§75-5-313[10][a], MCA). The idea behind the alternative variance is that the permittee is a very small proportion of the watershed's nutrient load. For example the permittee's discharge may be extremely small compared to the volume of the waterbody, and/or the waterbody may be highly dominated by non-point nutrient sources. Either way, an alternative variance is an option when the permitee can demonstrate that meeting general variance concentrations at §75-5-313[5][b], MCA (or future Department updates) would not result in an environmentally significant improvement in water quality and material progress towards attainment and maintenance of the waterbody's base numeric nutrient standards. Alternative variances are evaluated by the Department on case-by-case basis. Permittees may apply for an alternative variance for nitrogen, phosphorus, or both.

In many circumstances the need for an alternative variance will be precluded because the non-significance of the permittee's nutrient load to the waterbody in question will have already been accounted for in the development of the waterbody's Total Maximum Daily Load (TMDL), consistent with NEW RULE I(8). In such cases, the wasteload allocation in the TMDL becomes the basis for the discharge permit and no variance of any kind is needed. Put differently, the concentration of nutrients in the permittee's discharge may be higher than the general variance concentrations in statute (or future Department updates), but it would not be sensible— from a practical or economic perspective— to require the permittee to reduce those concentrations because their contribution to the overall watershed nutrient load is insignificant. Therefore, the permittee's existing discharge concentrations become the basis of the TMDL and the permit limit; no variance is needed.

In the absence of a completed TMDL, a permittee may apply for an alternative variance if it can be reasonably demonstrated to the Department that the discharger's nutrient load is non-significant. Watershed models are useful for this purpose and **Section 6.0** of this document addresses some modeling techniques. The Department will consider other modeling approach as well. The alternative variance derived via modeling can operate as an interim effluent limit until the time that the TMDL is completed.

Whether a point source is or is not a significant load in a watershed is not likely to be a static situation, and will probably change over time. Therefore, a permittee granted an alternative variance must demonstrate throughout the variance period that the facility's discharge has remained insignificant (per §75-5-313[10][b], MCA). This is necessary because if, for example, nonpoint source cleanups were substantial, the facility's nutrient load may have become significant in the watershed over time and may be preventing the waterbody from attaining the base numeric nutrient standards. Permittees granted an alternative variance should work with the Department regarding the frequency of monitoring needed to carry out the demonstration discussed in this paragraph.

5.0 STREAMLINED METHODS FOR DEVELOPING SITE-SPECIFIC NUMERIC NUTRIENT CRITERIA

5.1 BACKGROUND AND RATIONALE

Numeric nutrient criteria have been proposed for all major and several minor ecoregions in Montana (Suplee and Watson, 2013). Suplee and Watson (2013) also include a limited number of site specific criteria, and it has been acknowledged that the Department will need to develop other site-specific nutrient criteria going forward. A criteria development approach using empirical or process-based models (e.g., QUAL2K) is provided in **Section 6.0** of this document. That process is, however, data intensive. There will likely be streams which warrant site-specific numeric nutrient criteria but for which

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Comment [TL7]: I would recommend re-organizing this section to follow the flow of how SSC may be derived. For example,
Step1. Confirm the biological health of a segment (Section 5.3),

a smaller dataset and less rigorous analysis can be used; this paper outlines a simplified, streamlined approach for doing this. Criteria developed via this streamlined process may be adopted as site-specific standards under the Board of Environmental Review's rulemaking authority in §75-5-301(2), MCA.

This simplified approach was motivated by observations stemming from the application of the Department's methodology for assessing stream eutrophication (Suplee and Sada de Suplee, 2011). Using those methods, some streams have been found to support a healthy stream ecology and are in compliance with the biologically-based assessment parameters (e.g., levels of benthic chlorophyll *a*, macroinvertebrate HBI metric), but show exceedences of one or both of the nutrients (N, P) recommended as criteria. Site-specific numeric nutrient criteria are likely to be appropriate in these situations.

Section 5.0 is organized as follows:

Section 5.2: The basic concept and approach is presented;
Section 5.3: Assessment of biological health and minimum dataset requirements are provided; and
Section 5.4: A case study example is given.

5.2 SITE-SPECIFIC METHODS

This section outlines the streamlined approach for deriving site-specific nutrient criteria for streams and small rivers.

5.2.1 Principal Site-specific Methods

Nutrient concentration data from reference sites have been compiled for each ecoregion (Suplee and Watson, 2013). Data from dose-response studies (nutrient concentration as dose, impact to beneficial use as response) applicable to each ecoregion have also been compiled. Each of these data types provide concentration ranges within which this streamlined site-specific criteria method can operate. In applying this method, two scenarios will be encountered.

Ecoregion	Period when Criteria Apply	Derivation Approach	95% value	
Northern Rockies	X	Reference or dose-response	X	

Scenario 1: **Figure 5-1** illustrates how information from ecoregionally-applicable reference sites can be used. It is assumed here that a stream assessment (per Suplee and Sada de Suplee, 2011) has already been carried out and has shown that a particular stream's biological condition supports all uses, i.e., no detrimental eutrophication effects have been observed. In **Figure 5-1**, the Department's recommended criterion (black dot with X) falls within the reference distribution of the ecoregion's reference-site data (median dataset[2]; Suplee and Watson, 2013). This occurs in a number of ecoregions, for example for TP in the Middle Rockies, due to the fact that dose-response studies were the primary consideration in setting the criteria. What the data show us is that there are reference sites which routinely manifest nutrient concentrations higher than the regional criterion; therefore, there is a range of concentrations beyond the recommended nutrient criterion that may still be protective within the ecoregion.
In scenario 1, If an *assessed* stream meets the Department's biological expectations and manifests a nutrient concentration falling between the Department criterion and the 95th percentile of the

Comment [TL8]: Somewhere you might want to indicate that these methods can be used to establish more stringent and less stringent criteria on a site-specific basis.

Comment [TL9]: I would recommend adding a generalized flow diagram to demonstrate the process. It might also help connect all the various options for considering SSC development or the individual variance route. The suite of tools got confusing to me so I would imagine others would find it even more challenging.

The graphics are nice visuals; however, I think adding a table would be helpful. The table could mimic the criteria table in DEQ-12A and show whether that set of ecoregional criteria were based on reference values or dose-response. Then the table could present the 95% values for comparison to ambient data. The method for calculating the criteria is simply using the 80% of the ambient data for the segment, regardless of the original basis for the criteria.

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[2] The median dataset must be used for this analysis and is available from the Department. In the median dataset, within any given ecoregion, nutrient concentrations from each site were first reduced to a median, and then descriptive statistics were calculated for the population of site medians. For an example, see Table 3-1B in Suplee and Watson (2013).

ecoregional reference dataset (within the dashed arrow, **Figure 5-1**), then the assessed stream is eligible for a site-specific criterion. The stream's new criterion should be established at the 80th percentile of the stream's nutrient dataset[3]. This criterion can then be recommended to the Board of Environmental Review for adoption as a site-specific nutrient standard during a subsequent triennial review.

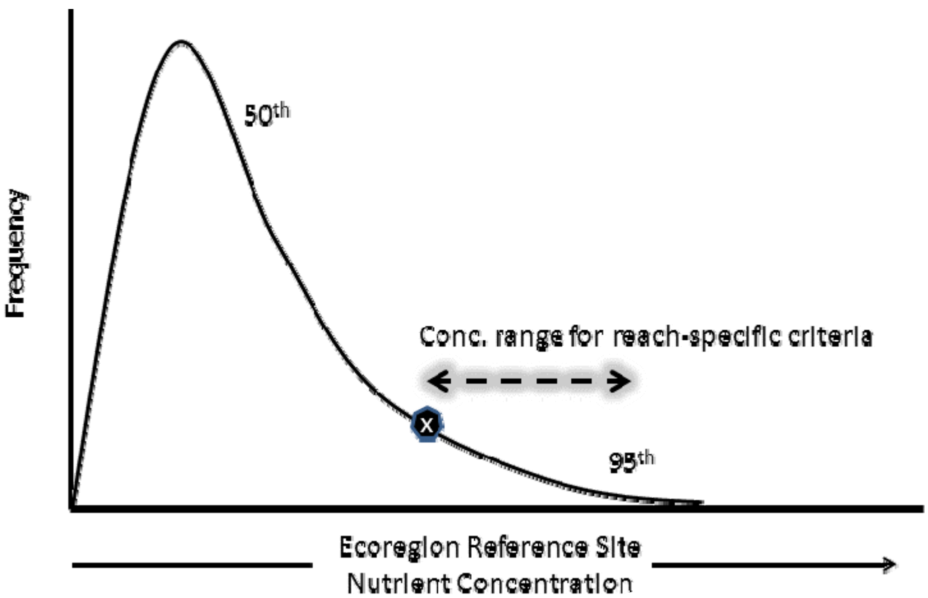

Figure 5-1. Scenario 1. Candidate site-specific nutrient criteria may fall between the ecoregional criterion recommended by the Department (black dot with X) and the 95th percentile of the applicable reference distribution (dashed arrow). The reference distribution used must be the median dataset from Suplee and Watson (2013) or its equivalent update. This method only applies to streams that demonstrate good biological health and full support of beneficial uses using assessment methods in Suplee and Sada de Suplee (2011).

__Scenario 2__: In other cases, the criteria recommended by the Department are very near to or beyond the 95th percentile of the ecoregional reference distribution. In these cases, the approach shown in **Figure 5-1** will not work and an alternative approach is illustrated in **Figure 5-2**. For each level III ecoregion,

[3] Assuming the assessment methodology in Suplee and Sada de Suplee (2011) remains the same, the stream in question would, in the future, be assessed using the binomial test for streams considered compliant with the nutrient criteria (i.e., null hypothesis is "stream compliant with nutrient criteria"). Due to the allowable exceedence rate (20%) and the gray zone (15%) established in the binomial test, a site-specific nutrient criterion set at the 80th percentile of the site's existing dataset will consistently PASS the binomial in the future (assuming the stream's nutrient conditions are unchanged). The T-test would also be PASS.

Suplee and Watson (2013) have provided in each concluding paragraph a range of concentrations from the dose-response studies they reviewed. The dose-response studies most applicable to the ecoregion in question (not the broader range of generally-applicable studies) will provide the concentration range within which site-specific criteria can be identified.

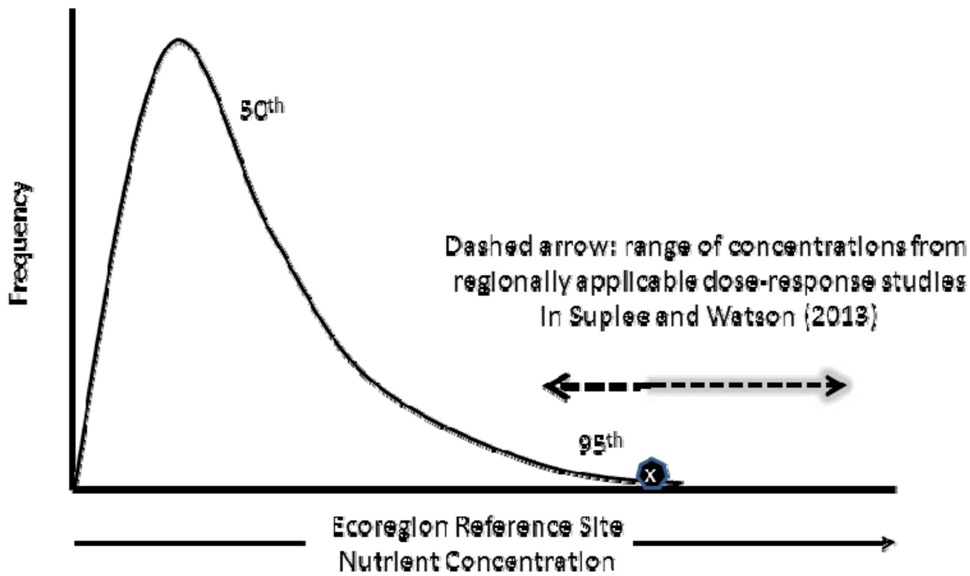

Figure 5-2. Scenario 2. Site-specific criteria derivation method for cases where a Department-recommended criterion is near or above the 95th percentile of the ecoregional reference distribution. Candidate site-specific nutrient criteria fall between the criterion recommended by the Department (black dot with X) and the upper range of the values from the dose-response studies specifically applicable to the ecoregion in question (dashed arrow with gray fringe). The dose-response studies must be from Suplee and Watson (2013) or equivalent updates.

If an *assessed* stream meets the Department's biological expectations but manifests a nutrient concentration above the Department's criterion, and that criterion is near or above the 95th percentile of the ecoregional reference dataset, then the range of concentrations from the applicable dose-response studies can be reviewed. If the assessed stream's nutrient concentration at the 80th percentile falls within the range of the regionally-applicable dose-response studies, then that concentration can be used as a site-specific criterion. This criterion can then be recommended to the Board of Environmental Review to be adopted as a site-specific nutrient standard.

In general, streams whose nutrient concentrations fall outside of the defined ranges in **Figures 5-1** and **5-2** are not eligible for this streamlined approach. Rather, methods outlined in ~~Appendix A of the Department's draft guidance~~ **Section 6.0** of this document ~~"*Nutrient Standards Implementation Guidance*"~~ should be used. There may also be cases where an upstream level IV ecoregion with naturally high nutrient concentrations is influencing the stream in question, and the reach-specific methods in Section 4.0 of Suplee and Watson (2013) may be applicable.

Moved (insertion) [1]

5.2.2 Other Methods

Recent work in the scientific literature provides a means to develop site-specific criteria on a stream-by-stream basis; the method was specifically developed for western regions of the United States (Olson and Hawkins, 2013). This method uses a geospatially-driven model that considers major environmental factors within a watershed that influence nutrient concentrations in streams (geology, precipitation, soil bulk density, etc.). The Department is using this method to help derive nutrient criteria for an area of the state with few or no reference sites and what appears to be naturally-elevated phosphorusconcentrations. It should be pointed out that the method is not for use in the plains region of Montana
(Olson and Hawkins, 2013).

The Department will consider results provided by others that have used the Olson and Hawkins (2013) method. (Again, this is predicated on the assumption that full biological support is shown in the stream.) However, results from this model will need to be reviewed by the Department on a case-by-case basis. If approved, they can be recommended to the Board of Environmental Review for adoption as site-specific standards.

5.3 Confirmation of Biological Health, and Minimum Dataset

This section addresses the minimum requirements needed to assert that the biological health of the stream fully supports beneficial uses.

5.3.1 Assessment of the Biological Health of the Stream

Assessment methods outlined in Suplee and Sada de Suplee (2011) will be used. That assessment methodology is designed to provide a minimum dataset by which eutrophication-based impacts to beneficial stream uses can be assessed. Data types include:

1. A minimum nutrient dataset (usually 12-13 independent samples)
2. Benthic chlorophyll *a* samples
3. Periphyton samples for taxonomic identification and biological metrics
4. Aquatic insect (macroinvertebrate) samples for taxonomic identification and biological metrics

Data are to be collected during the defined growing season for the ecoregion in question. Given that the minimum data requirements have been met for all data types (nutrients and biological), a stream assessment may come to a scenario that lends itself to site-specific nutrient criteria. **Table 5-1** shows cases where site-specific criteria are likely valid; the table shows just two of the many potential outcomes in the final status determination of a stream assessment (Suplee and Sada de Suplee, 2011).

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Section Break (Next Page)
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Comment [TL10]: May want to remove this section since it didn't seem to work at a localized scale and isn't currently being used by DEQ.?

Moved up [1]: <object>In general, streams whose nutrient concentrations fall outside of the defined ranges in **Figures 5-1** and¶
5-2 are not eligible for this streamlined approach. Rather, methods outlined in ~~Appendix A of the Department's draft guidance~~ **Section 6.0** of this document ~~*"Nutrient Standards Implementation Guidance"*~~ should be used. There may also be cases where an upstream level IV ecoregion with naturally high nutrient concentrations is influencing the stream in question, and the reach-specific methods in Section 4.0 of Suplee and Watson (2013) may be applicable.¶
¶

Comment [TL11]: Recommend moving this section to the beginning of the SSC section

Table 5-1. Data assessment outcomes which lend themselves to site-specific nutrient criteria.

Scenario(s)	Scenario subclass	Nutrient Binomial Test	Nutrient T-test	Benthic Algae	Diatom Increaser Taxa-Probability of Impairment	Macroinvertebrate HBI Score
7,8	7/8a	FAIL	FAIL	≤125 mg Chla/m2 or ≤35 g AFDW/m2	≤51%	>4
7,8	7/8b	FAIL	FAIL	≤125 mg Chla/m2 or ≤35 g AFDW/m2	≤51%	≤4

Comment [TL12]: DEQ staff are familiar with this table; however, stakeholders may find it confusing. Suggest removing the table and rely on the narrative description so stakeholders understand the context. Also, what if someone wants to pursue SSC but hasn't collected diatom or macro data but have chl-a data that shows full support. That may trigger a SSC study.

In **Table 5-1**, which applies to western Montana streams, it has been found that an assessed stream's nutrients are elevated and fail both statistical tests (the binomial, which looks as the proportion of observations above the criterion, and the t-test, which addresses the dataset average and the presence of high outliers). Note however that the biological signals are all or nearly all acceptable; benthic algal biomass is below the threshold, diatom metrics (where applicable) show a low probability of nutrient impairment, and the macroinvertebrate-based HBI metric is acceptable since it is < 4 (at least for scenario subclass 7/8b), meaning water quality is very good (Hilsenhoff, 1987). Of the two cases shown, subclass 7/8a is less clear due to the elevated HBI score and additional data collection would be warranted before site-specific criteria are developed. For prairie streams, see scenarios 5 and 7, part 2 (Suplee and Sada de Suplee, 2011) as they are equivalent to those in **Table 5-1**.

5.3.2 Dataset Minimum

Comment [TL13]: Recommend adding a table summarizing the minimum number of samples required per indicator.

All data collection ~~must~~ should follow Department SOPs (e.g., DEQ, 2011a; DEQ, 2011b; DEQ, 2012). Dataset minimums for a stream assessment are defined in Suplee and Sada de Suplee (2011). For the purposes of developing site-specific nutrient criteria via this process, the dataset needs to have been collected for three years (though not necessarily contiguously) for all of the data types required in Suplee and Sada de Suplee (2011). For western Montana streams, this would be nutrients, benthic chlorophyll *a*, diatoms (where applicable), and macroinvertebrates. If the dataset minimums to complete a stream assessment were achieved after just two years of data collection (which is common), a complete third year of data must be collected as well. For prairie streams, data types should include nutrients, measurement of dissolved oxygen (5 continuous days at a minimum, during summer), diatoms, and visual assessment of aquatic plant densities (DEQ 2011a), for a minimum of three years.

The complete, three-year dataset ~~must be~~is taken through the assessment data matrix. In some cases the additional year may change the initial outcome and it may result that the stream no longer comes to the scenarios shown in **Table 5-1** and site-specific criteria are not warranted. However if the assessed stream again arrives to the scenarios in **Table 5-1**, site-specific nutrient criteria are likely warranted and the approaches outlined in **Section 5.2** may be applied.

5.3.3 Consideration of the Other Nutrient

Where a site-specific criterion is warranted for a nutrient elevated above the Department's ecoregion-based criteria, consideration must be given to the other nutrient in the stream (N vs. P, and vise-versa). For example, a stream manifesting good biological health but elevated P concentrations may very likely be N limited, and should be maintained so. If N limitation were alleviated, there is a high likelihood that the biological health of the stream would be impacted. The Redfield ratio (Redfield, 1958) will be used as a general guide for establishing which nutrient limits (ratio < 6, N limits; ratio > 10, P limits) and for establishing the final concentration of the other nutrient.

What the updated criterion for the non-elevated nutrient should be needs to be determined on a case-by-case basis in conjunction with the Department. A first-cut approximation would be roughly 75% of the established ecoregional criterion concentration.

In some cases, *both* N and P will be elevated above the Department's recommended criteria. In such cases each nutrient should be evaluated per methods in **Section 2.0** and it may result that site-specific criteria for both N and P will be higher than the Department's values. In such cases factors other than nutrients are likely limiting nutrient effects in the stream.

5.4 CASE-STUDY EXAMPLE

The following is a case which lends itself to site-specific nutrient criteria.

5.4.1 Data Summary for Stream X (in Middle Rockies Ecoregion)

Years of data: 3 (2004, 2011, 2012)
Number of Nutrient Samples: 12-14 (meets minimum)
Average Total Phosphorus (TP) Concentration: 35 µg/L
Average Total Nitrogen (TN) Concentration: 40 µg/L
Benthic Chlorophyll a Samples: 3 (each comprised of 11 sub-replicates) (meets minimum)
Diatom Metric Samples: Not applicable (Department has no validated diatom-based metrics for the Middle Rockies ecoregion at this time)
Macroinvertebrates Samples: 3 (meets minimum)

5.4.2 The Assessment of Stream X

The applicable criteria for the Middle Rockies are 30 µg TP/L and 300 µg TN/L (Suplee and Watson, 2013). Data for stream X were evaluated and TN was found to be quite low (average = 40 µg/L), well below the recommended ecoregional criterion of 300 µg/L. However TP averaged 35 µg/L and was above the ecoregional criterion of 30 µg/L. All biological indicators were found to be acceptable; the data fit scenario subclass7/8b in **Table 5-1**. In additional, other aspects of the data were considered. The macroinvertebrate O/E scores were reviewed to see if they were above 1.0[4] (none were). The benthic

[4] O/E scores decline from an ideal score of 1.0 due to impacts from a variety of stressors (excess sediment, heavy metals, elevated temperatures, etc.). However it is not uncommon to see scores > 1.0. These indicate the stream has more species of macroinvertebrates than the model is expecting to see for the region. Essentially, slightly elevated nutrient levels have led to a less austere environment and more species can exist than is normally seen. For this reason O/E scores > 1.0 can be indicative of nutrient enrichment above reference. When nutrient enrichment becomes excessive, O/E scores again drop below 1 due to species loss.

chlorophyll *a* concentrations were not only below the threshold they were very low (<< 50 mg Chl*a*/m^2), as was algal AFDM. Nitrate concentrations were also evaluated, and all concentrations were very low.

5.4.3 Site-specific Criteria Derivation for Stream X using the Streamlined Approach

The Department's recommended criterion for the Middle Rockies ecoregion (where stream X is located) is 30 µg TP/L; this value matches the 82^{nd} percentile of the Middle Rockies' reference data (median dataset; Suplee and Watson, 2013). The TP concentration at the 80^{th} percentile of stream X's dataset is 42 µg TP/L, a concentration equal to the 89^{th} percentile in the Middle Rockies reference dataset. Therefore, stream X fits scenario 1 (**Figure 5-1**) because its site-specific TP value (42 µg/L) falls between the Department's recommended criterion and the 95^{th} percentile of the Middle Rockies reference dataset. Stream X's new criterion (42 µg TP/L) is not too far above the Department's criterion, so a large reduction in the stream's TN criterion is not warranted. But it is prudent to set the TN lower than 300, to 250 µg TN/L (which is at the 97^{th} percentile of the Middle Rockies reference distribution). This maintains a Redfield ratio of < 6 which should help maintain N limitation. **The site specific criteria would be 42 µg TP/L and 250 µg TN/L, applicable during the growing season for the Middle Rockies (July1-Sept 30).**

6.0 GUIDELINES FOR DEVELOPING SITE-SPECIFIC NUMERIC NUTRIENT CRITERIA VIA WATER QUALITY MODELING, AND THE RELATION OF THESE CRITERIA TO INDIVIDUAL NUTRIENT STANDARDS VARIANCES

Comment [TL14]: After re-reading this document, I would recommend moving this entire section on individual variance up in the document so it follows the discussion of variances (section 4). Then Section 5 can follow and focus on SSC development.

Circumstances may arise where, for a specific discharger, it may not make sense to move to the new, lower general variance concentrations at the time the Department updates them during a triennial standards review. Similarly, it may not make sense for a discharger to upgrade to one of the nutrient reduction steps (see **Section 2.0** of this document) that have been defined for the 3 permit cycles subsequent to the initial treatment requirements (e.g., 1 mg TP/L and 10 mg TN/L) defined in statute at §75-5-313 (5)(b), MCA.

In some cases a permittee may be able to demonstrate, using water quality modeling and reach-specific data, that greater emphasis on reducing one nutrient (the target nutrient) will achieve the same desired water-quality conditions as can be achieved by emphasizing reduction of both nutrients. Requiring a point source discharger to immediately install sophisticated nutrient-removal technologies to reduce the non-target nutrient to levels more stringent than what is in statute at §75-5-313(5)(b), MCA may not be the most prudent nutrient control expenditure, and would cause the discharger to incur unnecessary economic expense. Since this can be interpreted as a form of economic impact, *sensu* §75-5-313(1), MCA, these situations are appropriately addressed by individual variances.

If such a case can be demonstrated to the satisfaction of the Department, then a permittee can apply for an individual variance which will include discharger-specific limits reflecting the highest attainable condition for the receiving water rather than limits based on ~~any new~~a general variance concentration. ~~The demonstration must consider effects on the downstream waterbody including effects from the non-target nutrient; if the downstream waterbody will be impacted by the facility, some additional level of reduction on the target and/or non-target nutrient (beyond that required to protect beneficial uses in the receiving waterbody)will be necessary or the individual variance may not be granted. In addition,~~

tThe permittee is will be required to provide monitoring water-quality data that can be used to determine if the justification for less stringent effluent limits continues to hold true (i.e., status monitoring is required), consistent with New Rule I(4)(a). . This is Bbecause status can change, for example due to substantive nonpoint source cleanups upstream of the discharger, status monitoring by the discharger is required.

The purpose of **Section 6.0** is to provide guidelines for the types of information the Department would need to evaluate in order to grant an individual variance that allows a permit a discharger to (1) remain at treatment levels less stringent than any general variance requirements as defined in statute at §75-5-313 (5)(b), MCA (or Department updates), or (2) remain at levels less stringent than the reduction steps in **Section 2.0** of this guidance document. Individual variances approved by the Department become effective and may be incorporated into a permit only after a public hearing and adoption by the Department (§75-5-313[4], MCA).

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6.1 Mechanistic and Empirical Modeling Approaches for Establishing Individual Variances and (Potentially) Reach-specific Nutrient Standards and Individual Variances (If Necessary)

Two general approaches may be used to establish reach-specific nutrient standards: to establish that upgrading a wastewater facility to updated general variance levels would not result in material progress towards attaining defined water-quality endpoints and beneficial use support:

1. Simulations based on mechanistic computer models

2. Demonstration of use support based on empirical data

Comment [TL15]: Same paragraph repeated in Section 6.0. Delete here?

Deleted: In some cases a permittee may demonstrate, using water quality modeling and reach-specific data, that greater emphasis on reducing one nutrient (target nutrient) will achieve similar water-quality and biological conditions in the receiving water as can be achieved by emphasizing the reduction of both nutrients (i.e., both nitrogen and phosphorus).

Comment [TL16]: Isn't this the same as SSC development?

Comment [TL17]: 3 years is required for SSC, why is this different?

Whichever approach is selected—and in fact both approaches can be pursued simultaneously—the Department will require would like a 2-year biological characterization of the reach in question. A solid understanding of the biological status existing under the current level of water quality is required. Factors (both natural and human-caused) independent of nutrient concentrations can influence biological integrity and need to be understood. The biological characterization will change from case to case, but will normally involve collection of diatoms, macroinvertebrates, benthic and phytoplankton algae density, and critical physical and chemical parameters that influence these. See Section 2.0 of **Appendix A** for an example of the types of biological data and the rationale for each. The nutrient concentrations identified via this modeling may be adopted as site-specific standards under the Board of Environmental Review's rulemaking authority in §75-5-301(2), MCA, but would require an analysis of their downstream effects prior to adoption (downstream effects are discussed further in **Section 6.2**).

The following provides further detail on the two modeling approaches bulleted above.

Simulation Based on Mechanistic Computer Models. The Department will consider mechanistic model results that demonstrate that the lowering of one nutrient (e.g., TP) without the lowering (or with less lowering) of the other would achieve essentially the same water quality endpoint (i.e., equivalent movement towards the similar water quality and biological goals), subject to Department approval of the model and the model's parameterization. Modeled endpoints may include changes in water quality (pH, dissolved oxygen, etc.), and benthic and phytoplankton algae density. Mechanistic models ~~must~~ should be supported by data from a Department-approved study design that includes characterization of the chemical, biological, and hydrological conditions of the study reach during a lower-than-average baseflow condition. Data collection should follow Department SOPs.

The Department encourages the use of the QUAL2K model (Chapra et al., 2010) but may consider results from other water quality models as well. Assuming the point source is a major contributor to the nutrients in the receiving stream, ~~M~~modeled nutrient reduction scenarios from the facility can vary, but scenarios based on the five treatment levels described in Falk et al. (2011)—which represent steps in biological nutrient removal technologies—are encouraged by the Department. The Department will consider nitrogen and phosphorus independently in this analysis.

The state of the art in computer water quality/algal growth modeling is such that nutrient co-limitation and community interaction of river flora is poorly simulated (or is not simulated at all). Models usually treat algal growth dynamics in streams and rivers as though the algae were a monoculture (which is not the case). Because of the uncertainties in model simulations, the Department will require monitoring (per NEW RULE I [4][a]) for dischargers that are permitted to depart from general variance concentration requirements (via an individual variance) based on a mechanistic model. The intent of the monitoring is to corroborate (or refute) the computer simulated results. At a minimum, growing season benthic-algae sampling will be required for a reach of the river downstream of the permittee's mixing zone, to be established in coordination with the Department. If the base numeric nutrient standard for the river in question was developed based on another water quality endpoint (for example, pH), then data collection ~~must~~ should also include that parameter. If the collected data and the computer modeling results corroborate one another, then a reach-specific base numeric nutrient standard may be in order. Any reach-specific nutrient standards so determined may be adopted by the Board of Environmental Review under its rulemaking authority in §75-5-301(2), MCA, but would require an analysis of their downstream effects prior to adoption.

Demonstration of Use Support Based on Empirical Data. Permittees may begin at any time to collect nutrient concentration, benthic and phytoplankton algae, and other biological and water quality data in the receiving waterbody downstream of their mixing zone. In cases where the Department's base numeric nutrient standards for the waterbody were developed using a specific water quality endpoint (for example, pH), data collection must include that parameter. Data collection ~~shall~~ should follow Department SOPs. Permittees are strongly encouraged to coordinate with the Department on study design and data collection protocols upfront, to assure that the data will be acceptable to the Department when the time comes for evaluating the outcomes. For example, it has been shown that chlorination of effluent can, in some cases, mute the effects of nutrients for some distance downstream (Gammons et al., 2010); this would need to be accounted for in any study design. Subject to Department approval, these data may be used ~~to demonstrate that remaining at the previous general-variance treatment level (assumed here to have been achieved by the permittee) was adequate to support beneficial uses of the waterbody~~to develop an individual variance. If the collected data conclusively indicate that beneficial uses of the waterbody are fully supported, then reach-specific base numeric nutrient standards may be ~~in order~~appropriate. Any reach-specific nutrient standards so determined may be adopted by the Board of Environmental Review under its rulemaking authority in §75-5-301(2), MCA, but would require an analysis of their downstream effects prior to adoption. An example of an empirical approach to developing reach-specific nutrient criteria is provided in Section 2.0 of **Appendix A**.

6.2 PROTECTION OF DOWNSTREAM BENEFICIAL USES

Any reach-specific criteria developed for a receiving stream using a mechanistic or empirical model will also need to protect downstream beneficial uses. This is a basic requirement of a water quality standard under the Federal Clean Water Act and EPA Regulations found in 40 CFR Part 131. "How far downstream" is a consideration which will vary from case- to-case; an example is provided in Sections 2.7 and 4.0 of **Appendix A**. Mechanistic models have very clear advantages over empirical models for running hypothetical scenarios and assessing potential downstream impacts, however a mechanistic model will normally be more expensive to complete. A budget estimate for a mechanistic and an empirical model is provided in Section 6.0 of **Appendix A.** If it results that modeling (of either type) has shown that beneficial uses of the assessed reach can be protected with site-specific criteria, but a downstream reach will be negatively impacted by the higher concentrations of one (or both) nutrients, then the Department ~~will~~ would require treatment levels which ~~will~~would support the uses in the downstream waterbody, or it ~~will not grant the individual variance.~~ would have to recommend against the site-specific standards.

6.3. UNWARRANTED COST AND ECONOMIC IMPACT

In order to satisfy the economic impact component of an individual variance (§75-5-313[2], MCA) which may be developed as a result of the modeling methods described above, permittees ~~must~~ should provide the Department approximate estimates of the capital costs, and operations and maintenance costs, which would have been expended in order to upgrade the facility to the new general variance concentrations. The intent is to demonstrate that there were substantial savings in capital costs, materials, fuel, and energy by opting *not* to upgrade the facility. The permittee can compare the cost saved to the MHI of the community, similar to what is done for determining substantial and widespread economic impacts (see steps 1 through 5, **Section 2.2**); however, the Department wants to make clear that no specific percent of MHI needs to be realized in order for this aspect of the analysis to be satisfied. Permittees are encouraged to work with the Department's economist when carrying out this analysis (Jeff Blend or his successor). Capital costs saved would not include design-related work and overhead. Operations and maintenance cost saved should be estimates of fuel and/or electrical consumption, and other materials (e.g., chemicals). Permittees are not required to carry out a complex analysis comparing the relative economic or social value of protecting one resource (the stream or river) vs. another (e.g., air quality) and then trying to quantify the relative savings. Rather, the Department wants a straight-forward quantification of cost savings associated with the key factors of concern (capitol costs, fuel and electrical consumption, and routine materials used such as chemical additions).

6.4 DEPARTMENT PERIODIC REVIEW OF THE INDIVIDUAL VARIANCE

Comment [TL19]: This section seemed very similar to the approach for establishing SSC described in Section 5.2?

If it's not the same, I would recommend clarifying the different between the use of empirical data in this approach versus SSC would be helpful.

Comment [TL20]: Recommend moving this to the end of the SSC section.

Comment [TL21]: This section seems more relevant to the Individual variance discussion section earlier in the document (Pages 13- X). Recommend moving.

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<object>The ultimate endpoint of the modeling work is site-specific nutrient standards adopted by the Board that are demonstrably protective of downstream beneficial uses. In some cases where site-specific criteria have been developed ~~N~~an individual variance will be necessary, as the site-specific criteria may not be immediately achievable because (for example) the new criteria are still below the limits of technology and the point source is a major proportion of the stream flow. ~~utrient concentrations in the draft individual variance would be based on the results of modeling and the assessment of downstream use protection as described above~~

Moved up [2]: Individual variances approved by the Department become effective and may be incorporated into a permit only after a public hearing and adoption by the Department¶
(§75-5-313[4], MCA).

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Status monitoring of the receiving stream and the affected downstream waterbody will be used to evaluate the individual variance justification going forward. For example: model results have shown that a large reduction of phosphorus by the permittee would render the receiving stream P-limited and in full support of beneficial uses, without a major reduction in nitrogen. At the same time, nonpoint contributions of nitrogen to the downstream part of the waterbody of concern are presently large enough that a substantial reduction of nitrogen load ~~by~~ at the permittee's facility would have had little or no beneficial effect on the waterbody's uses. As a result, the permittee's individual variance reflects a low TP concentration and a TN concentration of, say, ~~10~~9 mg/L. If in the next ten years (of the twenty year variance period) nonpoint sources cleanup sufficiently that the facility's ~~10~~ 9 mg TN/L concentration has become a sizeable proportion of the downstream nitrogen load and reduction of that load would benefit the stream, then the justification for the ~~10~~ 9 mg TN/L will have changed. Any updated individual variance would reflect a lower TN concentration. As before, modeling could be used to help derive the updated TN concentration.

7.0 References

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APPENDIX A: RECOMMENDATIONS FOR SAMPLING AND MODELING THE EAST GALLATIN RIVER TO ACCOMPLISH MULTIPLE OBJECTIVES

1.0 Background

The Department indicated in its draft numeric nutrient standards rule package that a person may collect and analyze water quality and biological data along a reach of stream or river to determine if reach-specific numeric nutrient criteria different from those of the Department are warranted. A draft proposal of this type was provided to the Department in July 2012 for the East Gallatin River (HDR Engineering, 2012)[5]. The Sampling and Analysis Plan (SAP) provided to the Department in July 2012 (HDR Engineering, 2012) is based on sites that were sampled in 2009-2010 for the purpose of determining flow-stage relationships in the East Gallatin River. Building on those sites, the following are recommendations for an optimized study design which can be used to develop reach-specific nitrogen and phosphorus criteria for the East Gallatin River. It is hoped that this document may also serve as a blueprint for similar work that may be carried out on other Montana rivers or streams.
The Department already has a public-reviewed and finalized assessment methodology for determining when a stream reach is impaired by excess nitrogen and phosphorus (Suplee and Sada de Suplee, 2011). However, that assessment methodology was designed to be a minimum data method and was not intended to be sufficient for deriving reach-specific criteria. Therefore, the reader will find that methods recommended below are more data intensive than those needed to complete an assessment via the assessment methodology.

1.1 Design and Possible Outcomes of the Investigation

[5] It should be noted that the Department has developed reach-specific criteria for the East Gallatin River using approaches somewhat different than those provided here. See Section 4.0 in Suplee and Watson (2012).

The East Gallatin River is an excellent case study in which to explore several variations on the development of reach-specific criteria. These variations include:
1. The case where a stream reach may have natural factors (e.g. high turbidity, cold temperature, etc.) that suppress benthic algae growth, and therefore reach-specific criteria are appropriate;
2. The case where benthic algae is found to be above nuisance levels, but modeling shows the algae problem can be addressed by focusing on the reduction of one nutrient more than the other; or
3. The case where reach-specific numeric nutrient criteria for a reach of the East Gallatin River are appropriate, but consideration of downstream beneficial uses precludes their application.

Figure 1-1 below forms the basis for the recommendations in the rest of this document.

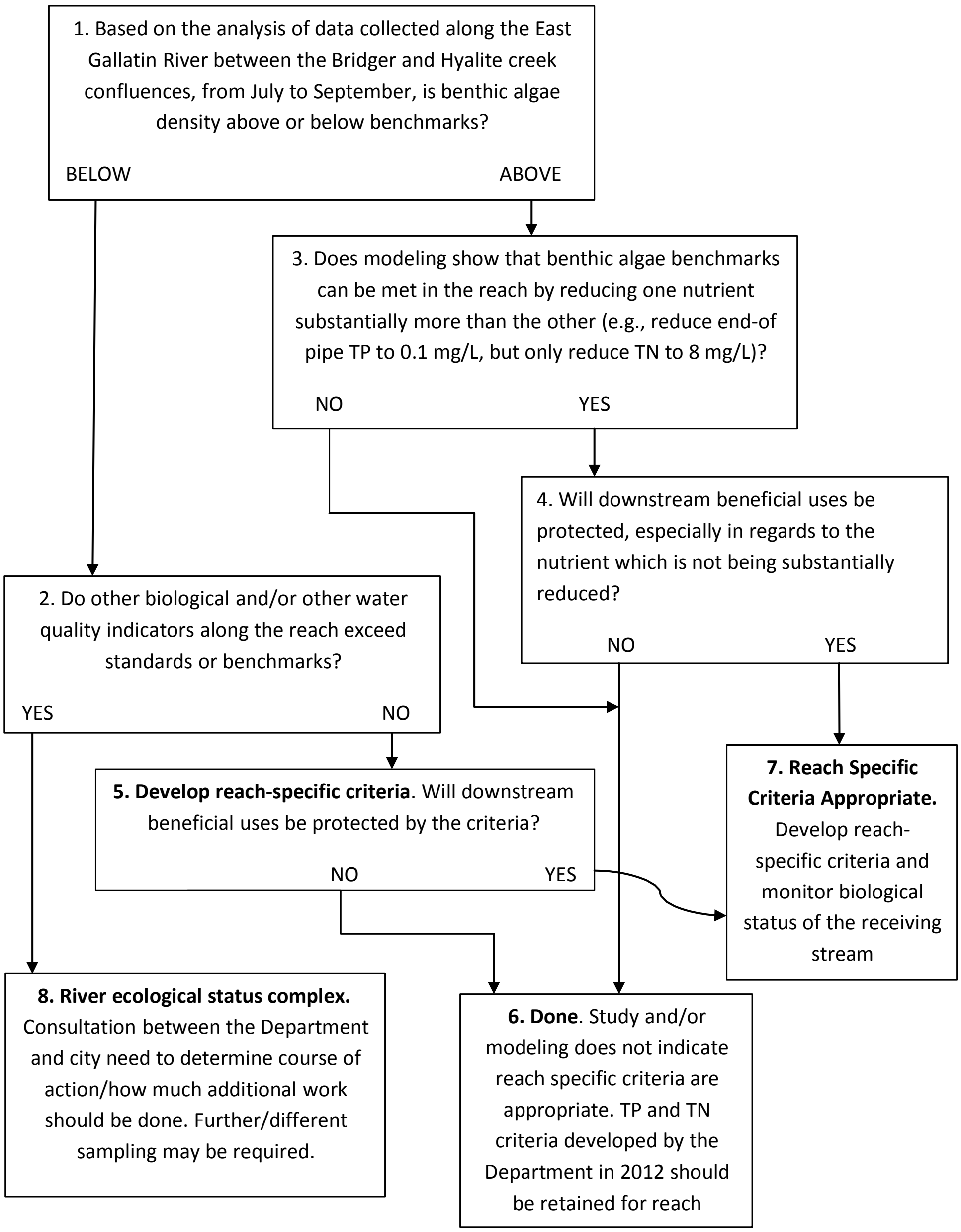

Figure 1-1. Flowchart outlining various outcomes from the analysis of reach-specific data and the development of reach-specific criteria.

Figure 1-1 provides for an empirical approach to developing reach-specific criteria and assessing downstream effects of these criteria. It provides a mechanistic model approach (starting in Box 3), as well as an approach where either option can be pursued (starting in Box 5). Regardless of which approach is taken, as shown in **Figure 1-1,** proper biological characterization of the mainstem East Gallatin River needs to be undertaken. Both criteria derivation approaches require robust field data and an understanding of the impairment status of the river in relation to nuisance algae and/or other aquatic life.

Please note that "other water quality indicators" (Box 2) in **Figure 1-1** does not include a comparison of measured nutrient concentrations to currently recommended criteria for the reach. (That would be circular.) It does, however, include things such as pH, DO, and DO delta; i.e., effect variables. It is a foregone conclusion (based on existing data) that much or all of the reach below the Bozeman water reclamation facility (WRF) outfall will manifest nutrient concentrations in excess of the Department's recommended criteria.

Figure 1-1 does not provide closure in all circumstances. There is a pathway by which one can arrive to Box 8 "River ecological status complex". If the study findings lead to this outcome, it is not clear at this point what the path forward would be. It may require substantially more sampling and analysis. The assumption here is that the Department and the city would want to discuss what (if any) further work would be carried out, and what the endpoints might look like.

1.2 Summary of the Basic Approaches to Reach-specific Criteria

Two broadly defined modeling approaches to developing criteria (empirical and mechanistic) are detailed in the following sections. Briefly, the basic characteristics and strengths and weaknesses of each are given below.

Empirical Approach. Fewer overall sites to sample compared to mechanistic modeling and, as a result, lower overall cost. Samples can be collected most years during baseflow. Samples need to be collected for at least three years, however two of those three years are already needed for the basic biological characterization of the reach and the same sites can be used for both. Robustness of the empirical statistical relationships are difficult to know in advance and could require additional data beyond three years. The ability to run "what if" scenarios or extrapolate predictions outside of the range of data from which the relationship is developed is much more limited compared to that of the mechanistic model.

Mechanistic Approach. This method requires more overall sites and more complex data collection compared to the empirical approach, with concomitantly higher cost. The mechanistic model still requires a two-year biological characterization, only some sites of which will overlap with the sampling sites for the model. The model will also require collection of DO, pH, etc. with deployed water-quality sondes. As you can imagine, these factors increase the cost and complexity of this approach. Data for calibration and validation of the model can be collected during one field season, provided that both collections are done near to peak growth and approximately a month apart. ~~Perhaps t~~Two separate low-flow years of data is probably a better corroboration of the model. Preferably, data collection should occur during a low baseflow (i.e., near the seasonal 14Q5 or, optionally, when baseflow is below the long-term seasonal average). This ensures that physical and biogeochemical conditions are consistent with that of the targeted low-flow period. Once the model is corroborated (i.e., validated) it can readily be used to run "what if" scenarios which can assess downstream uses, different nutrient reduction strategies at the Bozeman WRF and their effects, etc.

2.0 Biological Characterization of the East Gallatin River, and the Empirical Model Approach to Deriving Reach-specific Criteria

Objective 1: Determine the current biological condition of the reach of the East Gallatin River between the Bridger Creek and West Gallatin River confluences during the growing season (summer and early fall) and compare the results to standards and benchmarks used to assess stream eutrophication.

2.1 Detailed Consideration of the Objective 1

The following questions are designed to address objective 1 given above:

In the wadeable regions of the East Gallatin River between the Bridger Creek and West Gallatin River confluences, during the July 20 to September 30 period, what:

(a) are the average benthic algae densities (quantified as chlorophyll a and ash free dry mass, per m^2)?

(b) is the areal coverage and thickness of benthic algae and macrophytes (based on standardized visual assessment methods)?

(c) is the range and central tendency of specified macroinvertebrate metric scores (MT Hilsenhoff Biotic Index, O/E, and EPT taxa richness)?

(d) is the range and central tendency of specified diatom metric scores (WEMAP MVI and WEMAP WA TN)?

(e) are the dissolved oxygen concentrations and pH compared to state standards, and what is the dissolved oxygen delta (daily maximum minus the daily minimum)?

(f) are the concentrations of nitrogen and phosphorus (total and soluble) and total suspended solids?

(g) is the stream temperature, and incoming light intensity(in PAR units, e.g., µmol quanta/$m^2 \cdot s$)? (h) are the concentrations of herbicides which are frequently used in the watershed?

Note in the question at the start of **Section 2.1** the dates during which data collection should occur (July 20 to the end of September). These dates were based on the Middle Rockies growing season (Suplee et al., 2007), and the fact that in the East Gallatin River the first three weeks of July have considerably higher flows compared to August and September (shown in dark gray, **Table 2-1**). Commencing July sampling after July 20th will generally exclude the higher flows and lead to data collection during base flow conditions more consistent with August and September. Sampling could extend into the first two weeks of October, if temperatures remain moderate and base flow conditions remain reasonably stable (Suplee and Sada de Suplee, 2011).

Table 2-1. Discharge, ft^3/sec for USGS Station 06048700 "East Gallatin River at Bozeman, Mont.". Mean of daily values for 10 years of record (calculation period 2001-10-01 to 2011-09-30).

Day of month	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	42	47	45	118	283	433	164	52	43	40	55	47
2	44	43	44	128	267	441	155	51	42	41	55	47
3	44	42	46	124	268	453	147	53	39	42	57	47
4	41	43	48	112	297	433	142	53	37	44	56	47
5	43	44	47	121	295	418	141	51	39	48	55	47
6	43	47	46	148	328	425	130	52	42	50	53	47
7	41	44	46	139	364	479	124	51	43	51	55	46
8	46	44	52	140	379	461	118	52	41	51	62	43
9	44	42	54	149	376	440	108	54	43	52	60	43
10	42	42	56	157	380	443	102	52	50	52	56	44
11	41	42	58	155	373	513	101	49	45	52	56	46
12	42	42	70	164	373	501	97	46	41	53	56	46
13	43	42	88	182	377	465	94	45	42	52	57	45
14	44	42	88	218	404	436	90	45	42	52	56	45
15	43	41	80	232	439	420	84	47	43	55	52	45
16	42	41	80	212	442	404	81	44	42	59	55	43
17	44	41	81	229	464	390	78	44	44	61	54	42
18	46	41	86	239	484	359	75	47	45	59	53	41
19	51	42	89	235	509	335	73	46	44	59	53	43
20	48	40	88	231	528	310	68	42	44	66	52	44
21	47	41	93	254	523	299	66	41	46	63	49	45
22	44	41	94	279	505	277	66	41	47	58	47	44
23	44	41	94	324	495	264	67	45	48	56	48	46
24	44	41	90	315	500	247	62	43	49	56	46	44
25	43	41	89	290	615	237	63	41	46	57	48	45
26	43	42	95	293	540	228	64	41	43	55	50	46
27	47	43	93	270	502	209	63	39	42	55	48	44
28	46	43	95	266	475	195	61	39	42	55	47	44
29	44	41	91	274	490	183	55	41	42	57	46	46
30	45		97	295	466	175	51	41	44	57	47	44
31	43		104		444		50	43		56		43

To further address the questions posed at the start of **Section 2.1**, it will be necessary to measure a number of physico-chemical parameters; the rationale for measuring each of these is described below. Biological parameters specified in the questions above were selected because they are known to be directly influenced by or significantly correlate with lotic nutrient concentrations. The Department has established benchmarks for most of the physic-chemical and biological variables, and East Gallatin River data can be compared against these (DEQ-7, 2012; Suplee and Sada de Suplee, 2010).

Benthic algae densities (chlorophyll *a* [Chl*a*] and ash free dry mass [AFDM] per m^2). Based on work in the Clark Fork River, statewide public opinion surveys, and a whole-stream dose-response study, the Department is using average Chl*a* levels of 125 to 150 mg/m^2 and 35 g $AFDM/m^2$ as harm-to-use thresholds for western Montana rivers and streams (Dodds et al., 1997; Suplee et al., 2009; Suplee and Sada de Suplee, 2011). Algae densities above these levels impact the recreation and aquatic life uses. The Department also has standard visual assessment methods to asses algal and macrophyte density at a coarser scale (WQPBWQM-011, 2011). The general composition, amount, color, and condition of aquatic plants are visually assessed in the field using the Aquatic Plant Visual Assessment Form. This information helps describe the health and productivity of the aquatic ecosystem, records nuisance aquatic plant problems, documents changes in the plant community over time, and can be used to help corroborate the quantitative Chl*a* results.

Macroinvertebrate metrics. The Hilsenhoff Biotic Index (HBI) is included as part of the Department's current eutrophication assessment methodology (see Suplee and Sada de Suplee, 2011). The HBI index was designed to assess biological impacts caused by organic enrichment and eutrophication (Hilsenhoff, 1987). The Department considers HBI scores in the Middle Rockies > 4.0 to indicate an impact to aquatic life (Suplee and Sada de Suplee, 2011). Two other metrics, O/E and EPT richness, were considered during the development of the eutrophication assessment methodology since both metrics correlated significantly to nutrient concentrations (Tetra Tech, 2010); however, for simplicity, only the HBI was retained in that methodology. Nevertheless, it would be of value to include these metrics in this study. The O/E metric evaluates the taxa diversity that was actually **O**bserved compared to an **E**xpected taxa diversity for the location where the sample was collected. The Department uses an O/E ratio of 1.0 to 0.9 as un-impacted; ≤ 0.9 is the harm threshold (i.e., loss of 10% of species). Modest stream nutrient enrichment can actually cause the metric to be > 1.0. A Bray-Curtis Index should be calculated to accompany the O/E to help interpret counterintuitive O/E scores (WQPBWQM-009, 2012). The EPT richness metric was part of older DEQ protocols and has application to intermountain valley and foothill streams. EPT richness values > 14 are considered healthy and this value will decline with water quality impacts (Bukantis, 1998).

Diatom metrics. The Department currently addresses nutrient impacts using increaser diatom taxa metrics which were developed using discriminant function analysis (Bahls et al., 2008, Teply, 2010a and 2010b; Suplee and Sada de Suplee, 2011). Currently there is no calibrated and validated model for the ecoregion in which the East Gallatin River resides (the Department hopes to have such a metric in a year or so). Therefore, two diatom metrics are recommended (one for TN, one for TP) which were developed by others and which correlate closely with stream nutrient concentrations in Montana (Tetra Tech, 2010). The metrics are WEMAP WA TN (for TN) and WMAP MVI (for TP); each was developed from work in the Western Environmental Monitoring and Assessment Program (EMAP) of the early 2000s. Results that differ largely from the regression line shown in Tetra Tech (2010) might suggest a stream with characteristics different from the Middle Rockies norm; for example, a WEMAP MVI diatom score of 1.5 associated with a TP concentration of 0.25 mg/L would be well outside the expected pattern (one would expect a score closer to 3)(Tetra Tech, 2010).

Dissolved oxygen, pH. Standards for dissolved oxygen (DO) and pH for a B-1 waterbody are established in state law (DEQ-7 October, 2012). DO and pH have been linked to elevated nutrient concentrations (Stevenson et al., 2012), making them good parameters to measure. But the Department has frequently observed that DO minima are not found to be out of compliance in heavily eutrophied streams, at least during summer, due to stream re-aeration. However, punctuated DO problems can occur in fall when the built-up algae senesce *en masse* (Suplee and Sada de Suplee, 2011). Therefore, in addition to state-adopted DO standards, the Department uses DO delta (daily maximum minus the daily minimum) of 5.3 as a benchmark for excessive plant productivity and respiration in streams (see Appendix C.2, Suplee and Sada de Suplee, 2011). Others have found DO delta to be valuable in assessing eutrophication in northern rivers, and recommend a benchmark of 5.0 (Minnesota Pollution Control Agency, 2010).

Concentration of nitrogen and phosphorus (total and soluble), total suspended solids, temperature, incoming light intensity, and herbicide concentrations. These water quality parameters are critical for the development of empirical relationships between algae density and nutrient concentrations. Variables that influence light levels are particularly important for algal growth rates. Light measurements can include PAR near the stream bottom, or (as a possible surrogate) measurements of canopy density above the water's surface. Temperature alters the growth rates of stream algae. In addition, stream samples for herbicides which have historically been used in the basin should be collected as these, if present in sufficient concentration, could suppress algal growth. Previous work has shown herbicides to be present in Montana rivers and streams, with atrazine, metolachlor, and triallate being among the most commonly detected (USGS, 2004). Algae (as well as macrophytes) are sensitive to these herbicides and growth can be suppressed at fairly low concentrations (see work by the USGS and EPA at: http://www.epa.gov/oppefed1/ecorisk_ders/aquatic_life_benchmark.htm#benchmarks, **and** http://www.cerc.usgs.gov/clearinghouse/data/usgs_brd_cerc_d_cerc008.html . The Department would not consider suppression of algal growth in the East Gallatin River due to herbicides as a viable rationale for reach-specific nutrient criteria because (a) it is not a naturally occurring environmental variable and (b) future application of BMPs might reduce the amount of herbicides reaching the river and this change could remove the algae-suppressing effect.

2.2 Data Collection Methods

The Department has Standard Operating Procedures (SOPs) for the collection of benthic and phytoplankton algae (both quantitative and qualitative methods)(WQPBWQM-011, 2011), diatoms (WQPBWQM-010, 2011), macroinvertebrates (WQPBWQM-009, 2012), and water quality (WQBWQM-020, 2012), and recommended methods for measuring DO, pH, and DO delta when assessing eutrophication (Suplee and Sada de Suplee, 2011). The Department's 3rd iteration of the Field Procedures Manual (WQBWQM-020, 2012) also summarizes parts of the SOPs most pertinent to field sampling. I recommend these methods be adhered to for all sampling in the East Gallatin River. These documents can be found at: http://deq.mt.gov/wqinfo/qaprogram/sops.mcpx.

A common trait of all the biological sampling methods is the necessity of laying out a short sampling reach, which the Department usually refers to as a 'site'. These short reaches are typically 150 to 300 m in length in wadeable streams, and are delineated at the time of sampling as 40X the wetted width of the stream or a minimum of 150 m. Sample collection at locations where there is a large proportion of the river that is unwadeable requires special consideration and these situations are also addressed in the SOPs.

Collection of DO, temperature, pH, and DO delta are best measured with deployed data sondes (e.g., YSI 6600s). Continuous collection of data via sondes is not needed at all stations but 1 or 2 along the East Gallatin River study reach is recommended for biological characterization. These instruments can be rented seasonally from commercial suppliers.

Details on data collection will need to be elaborated upon in the final Sampling and Analysis Plan (SAP) developed to implement this general study design.

2.3 Recommended Sampling Sites along the East Gallatin River

To address objective 1 and its associated questions, ten sampling sites have been identified along the East Gallatin River between the Bridger Creek and West Gallatin River confluences (**Figure 2-1**). These ten sites are key to the implementation of the empirical approach outlined in **Section 1.2**. Seven sites (A to G; **Figure 2-2**) are intended for more intense chemical and biological sampling, while three (H to J) may be less intensively sampled and are the foundation of the downstream use assessment.

Site A (~0.7 miles downstream of the Bridger Creek confluence, at 45.71516, -111.0358): Establishes water quality and biological conditions near the head of the study reach. Suplee and Watson (2012) indicate that the East Gallatin River upstream of the Bridger Creek confluence should have a higher TP criterion (to account for the natural influence of the Absaroka-Gallatin Volcanic Mountains ecoregion). However, the elevated TP has been diluted out once Bridger Creek joins the river, and the recommended criteria are then the same as for the Middle Rockies as a whole. The site is the natural

starting point for the work. This site also corresponds to site 1 of the mechanistic model (i.e., the QUAL2K model).
Site B (~0.3 stream miles upstream of Bozeman WRF outfall, at 45.72568, -111.06469): Provides a second site to characterize the upper extent of the study reach. It is also not far upstream from the major point source on the river and so can provide a nearby point of reference for any changes occurring downstream of the facility. See also, **Figure 2-3.**

Site C (~0.9 stream mile downstream of the Bozeman WRF outfall, at 45.7284, -111.072): First site downstream of the city of Bozeman WRF discharge. A study shows that the facility's effluent is completely mixed within about 400 ft (0.08 miles) of the discharge (USGS, 1999), although flows at the time of the study were nearly double that of average conditions and nearly 3X the 7Q10. This site—located about 0.9 miles downstream of the discharge— should capture changes in the river due to the effluent, post-mixing. See also, **Figure 2-3.**

Site D (~0.3 stream miles downstream of the Riverside Water & Sewer District ponds, at 45.7363, -111.07105): Conversations with Department staff indicate that the Riverside Water & Sewer District ponds are a likely source of nutrients to the East Gallatin River. By establishing this site (and the one upstream, site C) it should be possible to discern differences in river biology and water quality due to the Bozeman WWTP effluent vs. any subsequent changes due to the ponds. See also, **Figure 2-3**. This site also corresponds to QUAL2K model site 2.

Site E (~0.6 stream miles downstream of the Buster Gulch irrigation diversion, at 45.74765, -111.08195): Site is established below a major water withdrawal to Buster Gulch. The site is established in order to determine if lower water volume is having a measureable effect on water quality or biology of the reach below the withdrawal.

Site F (Lower third of reach at 45.76698, -111.0968): Site will provide data representative of the reach between site E upstream and site G downstream. There are few notable characteristics in this reach of the river (e.g., point sources, tributaries, etc.) and this site will help ascertain the degree to which upstream loads extend their influence downstream.

Site G (upstream of confluence with Hyalite Creek, at 45.7888, -111.1195 [same as site EGRF2]): Establishes water quality and biological conditions near the end of the reach prior to the Hyalite Creek confluence. This site corresponds to a site established in an earlier study on the river (PBS&J, 2011). Any earlier data can be compared to that collected for this study. This site also corresponds to QUAL2K model site 3.

Site H (just upstream of the Dry Creek Irrigation withdrawal, at 45.83059, -111.14617): Nutrient criteria recommended for Hyalite Creek are higher for TP (due to natural geologic sources) and slightly lower for TN (to maintain N limitation) than the reach of the East Gallatin River into which Hyalite flows (Suplee and Watson, 2012). As such, Hyalite Creek is an important water quality change point. This site is intended to discern changes resulting from Hyalite Creek and to characterize the East Gallatin just prior to the Dry Creek irrigation withdrawal. This location is the first site intended for the assessment of downstream uses. This site also corresponds to QUAL2K model site 4.

Site I (just upstream of the Dry Creek Irrigation System return flow, at 45.88921, -111.26408): The Dry Creek Irrigation system is one of, if not the largest, irrigation withdrawals on the East Gallatin River. Irrigation return flows can be a significant source of nutrients and turbidity. The intent of this site is to

characterize the East Gallatin River just prior to the addition of irrigation return flow to the river. The site is part of the assessment of downstream uses, and also corresponds to QUAL2K model site 5.
Site J (just upstream of the confluence with the West Gallatin River, at 45.8923, -111.3286 [same as site EGRF1]): This site is located just upstream of the confluence with the West Gallatin River, and should reflect effects from the Dry Creek irrigation return. The site corresponds to an earlier study site (EGRF1; PBS&J, 2011) and so flow-stage relationships established there can be used; it also is the end of the study reach. The site is part of the assessment of downstream uses, and also corresponds to QUAL2K model site 6.

If resources are a constraint, objective 1 can be addressed with a scaled-down version of this plan. At a very minimum, the Department recommends that sites B, C (or as alternate to C, D), F, G, H, I and J be sampled.

2.4 Sampling Frequency and Duration of Study

Each site should be sampled synoptically at least once during the months of July, August, and September. This will provide good characterization of the sites during baseflow. Two years of data should be collected for the basic biological characterization. This will provide enough information to have some confidence in the biological status of the river during baseflow. If it is intended that the empirical criteria-derivation approach is taken, at least one more year (three total) of baseflow data should be collected at the sites. (Requirements associated with the mechanistic model approach are addressed in **Section 3.0**.) However, if a particular year has unusual high flows ≥ 165% of the long-term average August and September flows, data should not be collected until flows have declined to below this volume. At the USGS gage station at Bozeman on the East Gallatin River (gage No. 06048700), the long-term average flow in August and September is 45 ft^3/sec; thus, until summer and fall flows fall below 74 ft^3/sec, sampling should not occur.

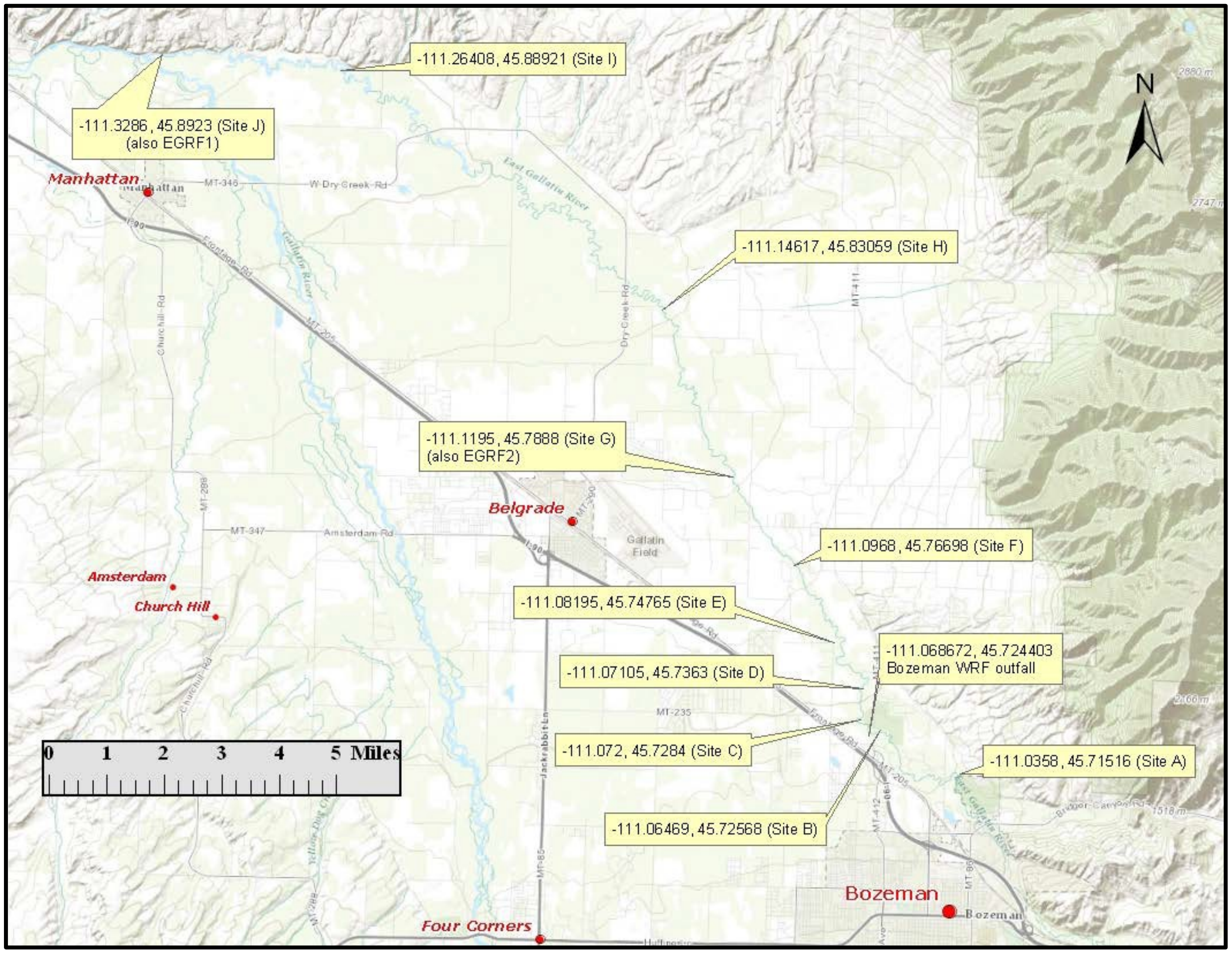

Figure 2-1. Ten biological and water quality sampling sites along the East Gallatin River. Sites A to G are for biological characterization of the East Gallatin River in the reach below the WRF. Sites H to J are for biological characterization <u>and</u> for assessing downstream use protection.

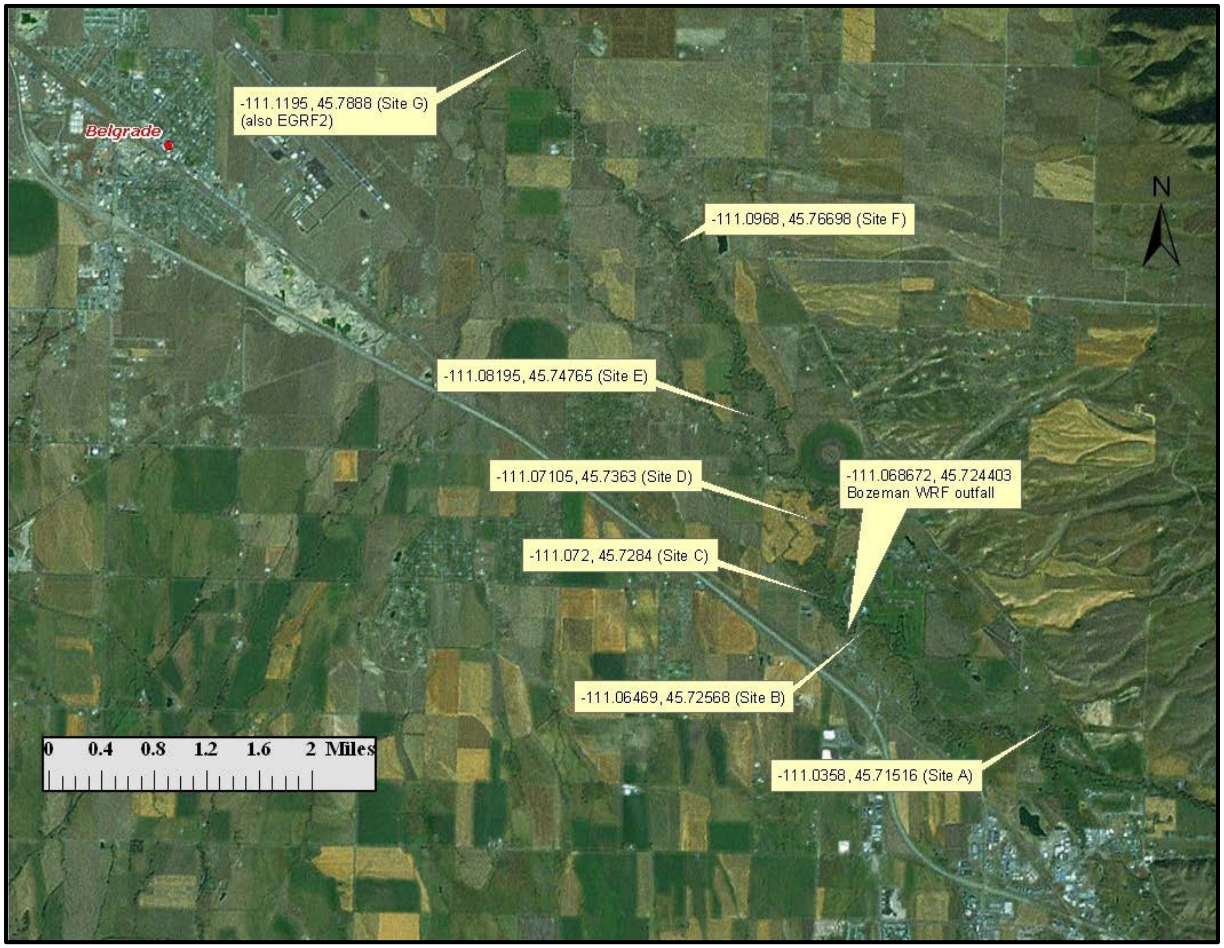

Figure 2-2. Sampling sites A to G along the East Gallatin River between the Bridger and Hyalite creek confluences.

Figure 2-3.Close-up of the three sampling sites around the city of Bozeman WRF discharge.Green dot is USGS gage 06048700.

2.5 Data Analysis and Interpretation

Due to the number of variables measured (e.g. benthic algae density, macroinvertebrates, diatoms), many different data combinations and outcomes are possible. The Department does not believe that establishing a rigid analysis structure upfront—that is, laying out the exact statistical tests, data aggregation methods, etc.—would be beneficial at this point. There are still a number of unknowns going forward and we must allow ourselves some flexibility in how the data will be interpreted. When statistical tests are, ultimately, carried out, a balance should be sought between type I and II error rates, as has been instituted in other Department stream-assessment procedures (Suplee and Sada de Suplee, 2011). This will seek a balance between error that imposes unneeded cost on the regulated community, and error that leads to degradation of (or lack of improvement to) the river environment (Mapstone, 1995).

2.6 Reach Specific Criteria—Empirical Approach

If it appears that natural environmental factors are keeping benthic algae density below nuisance levels in spite of elevated nutrient concentrations, then it may be possible to develop a reach-specific multiple regression equation involving nitrogen, phosphorus, and the additional environmental variable(s) of relevance, as has been done by others (e.g., Dodds et al., 1997; Biggs, 2000). Whether there will be enough data to develop significant relationships is hard to predict in advance, especially if the reduced-sites approach is selected; but it is safe to say the dataset will be relatively small and will require the assumption that all (or most) sites are independent from one another and samples collected a month apart are temporally independent. The Department has been able to substantiate similar assumptions in other cases (see Appendix A.3, Suplee and Sada de Suplee, 2011).
The multiple regression might take on the following form (Neter et al., 1989):

$$Y = \beta_o + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 + \beta_n X_n$$

where Y is the dependent (or response) variable, what is being predicted or explained; β_o is a constant or Y-intercept; β_1 is the slope (beta coefficient) for X_1; X_1 is the first independent variable that is explaining the variance in Y; β_2 is the slope for X_2; X_2 is the second independent variable that is explaining the variance in Y; β_3 is the slope for X_3 and X_3 is the third independent variable that is explaining the variance in Y, and on so on for the total number of slope·variables used ($\beta_n X_n$). For purposes of this work, Y equals benthic algae density (mg Chl*a*/m^2, g AFDM/m^2). Likely explanatory variables (βs) would be TN concentration, TP concentrations, TSS concentration, and stream-bottom PAR. This same approach could be used to explain relationships between other response and causal variables (e.g., macroinvertebrate HBI score as the response [Y], TN, TP, and TSS as causal variables [βs]).

2.7 Protection of Downstream Uses

The next step in the process is to determine if downstream uses will be protected by the reach-specific criteria (Box 5, **Figure 1-1**). Nutrients are assimilated longitudinally in streams and elevated concentrations will eventually decline due to biological uptake and adsorption to the sediments. Thus, assessing protection of downstream uses amounts to an evaluation of whether or not the higher nutrient concentrations being allowed upstream will have a deleterious effect downstream.
It is unlikely that any reach-specific criteria in the East Gallatin River would affect the Missouri River. The confluence of the three forks of the Missouri River results in orders-of-magnitude greater summer flows

than the East Gallatin River. For example, mean August flow in the Missouri River ~24 miles downstream of the three forks is around 2,747 ft^3/sec, whereas in the Gallatin River at Logan it is 490 ft^3/sec, and near the mouth of the East Gallatin River it is about 250 ft^3/sec (USGS, 2002; PBS&J, 2011). The most likely impacts from reach-specific nutrient criteria would be in the reach of the East Gallatin River downstream of the Hyalite Creek confluence. The nitrogen criterion recommended for the East Gallatin River between Hyalite Creek and the confluence with the West Gallatin River is 290 µg TN/L, lower than the 300 µg TN/L for the Middle Rockies (Suplee and Watson, 2012). Data suggest that the stream is nitrogen limited (since TP is naturally elevated) and is the reason why a lower TN criterion has been recommended there. A relaxation of the nitrogen criterion upstream of Hyalite Creek could very well lead to use impacts if the nitrogen limitation is, consequently, alleviated. Two approaches (which tie to Box 5 in **Figure 1-1**) can be taken to address downstream effects:

An empirical approach. If the sites along the East Gallatin River downstream from Hyalite Creek (sites H, I, and J) show a general immunity to elevated nutrients (and the reach upstream of Hyalite Creek does as well) due to some natural factor like elevated turbidity, then reach specific criteria in the East Gallatin River could be extended all the way from the Bridger Creek confluence to the confluence with the West Gallatin River, or even beyond, to the confluence with the Missouri River. However if the reach of the East Gallatin River downstream of the Hyalite Creek confluence shows biological impacts/nuisance algae above targets, then reach specific criteria that may be appropriate for the East Gallatin River further upstream will not protect downstream uses, and should not be put in place.

A mechanistic modeling approach using QUAL2K. This approach links to **Section 3.0**. The model would extend the full length of the East Gallatin River, between the Bridger Creek and West Gallatin River confluences to ascertain whether nutrients at a certain concentration, moving downstream from the point where Hyalite Creek confluences with the East Gallatin, would impact the beneficial uses further downstream. Beneficial uses addressed by the model include DO delta, pH delta, and benthic algae density. **Please note that the mechanistic model requires additional types of sampling and sampling sites (tributaries, irrigation withdrawals and returns) than the empirical approach; see Section 3.0.** The next section discusses approaches that can be used to develop a mechanistic model.

3.0 Developing Reach Specific Criteria via the Mechanistic Modeling Approach

Objective: Collect enough data along the East Gallatin River between the Bridger Creek confluence and the West Gallatin River confluence during a low-flow condition to be able to calibrate and confirm a mechanistic QUAL2K model of the study reach.

This objective still requires adequate biological characterization of the reach, as outlined in **Sections 2.1 through 2.5.** Many sites described in **Section 2.0** overlap with model sites described below; this was done in order to optimize sampling. To assure the reach is long enough to be able to judge the validity of the rate coefficients used in the model, the longitudinal distance must be sufficient to observe during calibration the decline in soluble nutrients, conversions to organic from algal death and recycling, etc. It is the Department's judgment that the East Gallatin River can be effectively modeled if the reach from above the Bozeman WRF to the West Gallatin River confluence (**Figure 3-1**) is considered, a distance of approximately 25 stream miles.

Mechanistic models for criteria derivation require a robust set of field observations including streamflow and water-quality data, measurements from continuously deployed sondes (including, at a minimum, dissolved oxygen, pH, temperature, conductivity, and turbidity), and biogeochemical kinetic observations (if possible). The Department has a detailed Quality Assurance Project Plan (Suplee et al., 2006) and a technical report (Flynn and Suplee, 2011) on the use of the QUAL2K model for developing reach-specific nutrient criteria; the reader is referred to those documents for greater detail. Selected sites are best sampled during one low-flow summer and fall (i.e., a year with flows near the seasonal 14Q5 of the East Gallatin River [McCarthy, 2005] or, alternatively, sequential low-flow summers during the peak of the growing period. Consecutive years with base flows that are below average is preferred but may not always be possible. **If, during the initial biological and water-quality characterization (Sections 2.1 through 2.5), it is found that herbicides are high enough to suppress algal growth, the model will be severely compromised. Therefore, herbicide data are best collected and then assessed in advance of the decision to complete the mechanistic model detailed below.**

3.1 Sites Requiring Water Quality Sonde Deployment

For the QUAL2K model, six sites are recommended (**Figure 3-1**). Sondes could be deployed continuously, or for a week to ten days in middle to late August and then again for another week to ten days in middle to late September, during period of relatively stable flow (or in two sequential Augusts if each has lower-than-average baseflow).

Water quality samples for key model drivers (nutrient concentrations—which include total nitrogen, nitrate+nitrite, ammonia, total phosphorus, and soluble reactive phosphorus; TSS and ISS; alkalinity; hardness; $CBOD_{20}$; Total Organic Carbon [TOC]; and benthic and phytoplankton algae) need to be collected at the six sites, at least once in August and once in September (or in sequential low flow years). These data collections could potentially be synchronized with the data collection in **Section 2.1**.

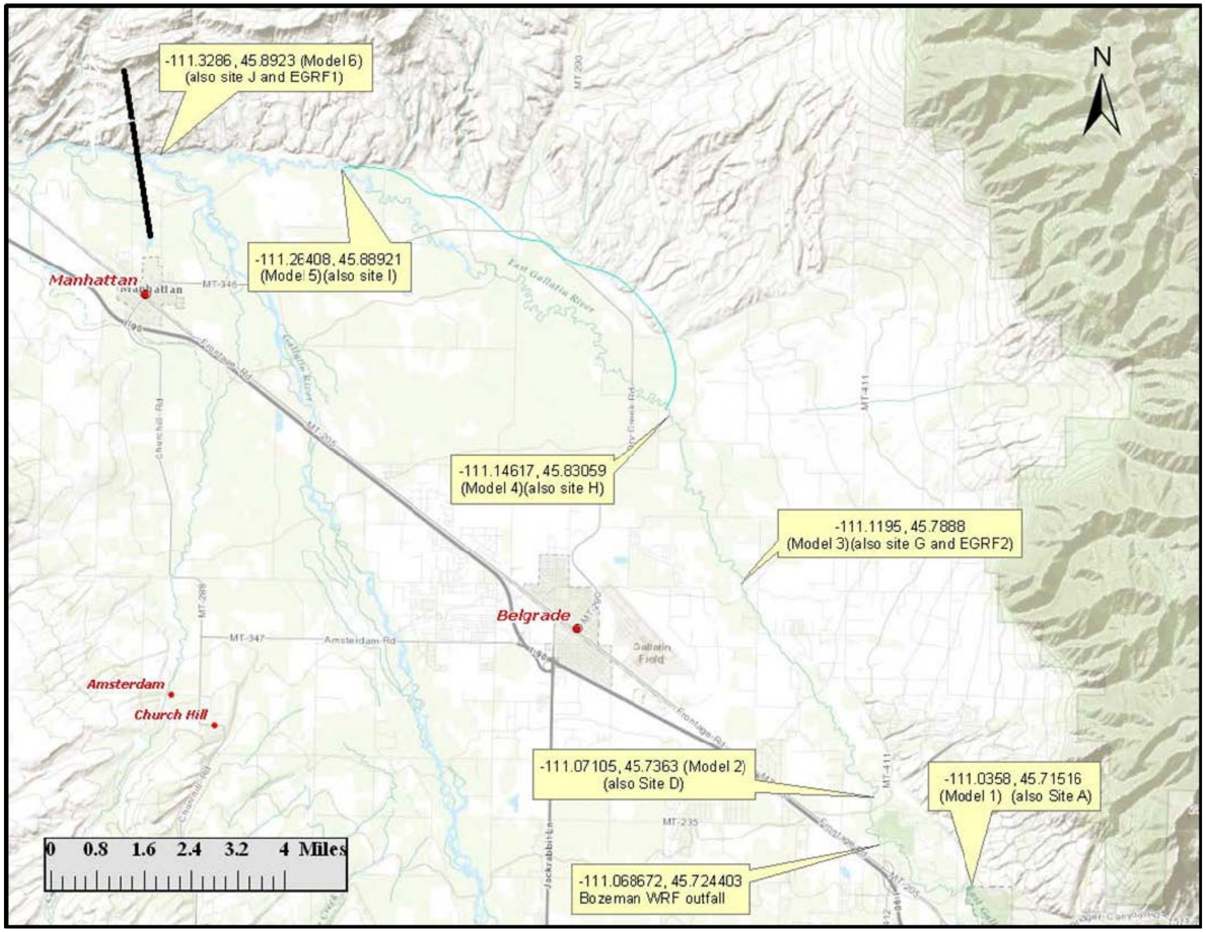

Figure 3-1. Map showing the six main sites along the East Gallatin River needed for the development of the QUAL2K model. Twelve other sampling sites (tributaries, irrigation canal withdrawals, etc.) are needed to develop the model but are not shown on this map.

The sites are:
Model Site 1 (~0.7 miles downstream of the Bridger Creek confluence, at 45.71516, -111.0358; same as Site A): Establishes water quality boundary conditions near the upper-most point of interest on the East Gallatin River based on reasons provided previously (page 9).

Model Site 2 (~0.3 stream miles downstream of the Riverside Water & Sewer District ponds, at 45.7363, -111.07105; same as Site D): For the purposes of the model, this site is intended to represent conditions in the East Gallatin River after the full mixing of Bozeman's WRF effluent discharge and any effects that may be coming from the Riverside Water & Sewer District ponds (see **Figure 2-3**).

Model Site 3 (upstream of confluence with Hyalite Creek, at 45.7888, -111.1195 [same as site G and site EGRF2]): Establishes water quality conditions in the East Gallatin River just before the confluence of Hyalite Creek, which naturally has differing nutrient concentrations (Suplee and Watson, 2012). This site corresponds to a site established in an earlier study (PBS&J, 2011). Any earlier data and flow-stage relationships can be compared to that collected for this study.

Model Site 4 (just upstream of the Dry Creek Irrigation withdrawal, at 45.83059, -111.14617, same as site H): Nutrient criteria recommended for Hyalite Creek are higher for TP (due to natural geologic sources) and slightly lower for TN (to maintain N limitation) than the reach of the East Gallatin River into which Hyalite flows (Suplee and Watson, 2012). As such, Hyalite Creek is an important water quality change point. Model Site 4 is intended to discern changes resulting from Hyalite Creek, and characterize the East Gallatin just prior to the Dry Creek irrigation withdrawal.

Model Site5 (just upstream of the Dry Creek Irrigation System return flow, at 45.88921, -111.26408, same as site I): The Dry Creek Irrigation system is one of if not the largest irrigation withdrawals on the East Gallatin River. Irrigation return flows can be a significant source of nutrients and turbidity. The intent of this site is to characterize the East Gallatin River just prior to the addition of irrigation return flow to the river. Changes in water quality as a result of this inflow will be captured by the next site downstream, model site 6.

Model Site 6 (just upstream of the confluence with the West Gallatin River, at 45.8923, -111.3286 [same as site J and site EGRF1]): This site is located just upstream of the confluence with the West Gallatin River, and should reflect any effects from the Dry Creek irrigation return. The site corresponds to an earlier study site (EGRF1; PBS&J, 2011) and flow-stage relationships established there can be used; it also is the end of the modeled reach.

3.2 Additional Sites Requiring Flow and Water Quality Data

Proper quantification of the water balance, associated mass fluxes, and water quality changes resulting from inputs and outputs to the East Gallatin River are key to a successful modeling strategy. As a result, there are a number of large and small tributaries inflows, irrigation withdrawals and return flows, and point source contributions that need to be quantified. These should be sampled for concentrations of nutrients (total nitrogen, nitrate+nitrite, ammonia, total phosphorus, and soluble reactive phosphorus), TOC, alkalinity, TSS and ISS, hardness, and $CBOD_{20}$ along with instantaneous measurement of temperature, DO, conductivity, pH, and flow.

A list of important hydrologic features that the Department believes should be characterized is shown below. Other tributaries and canals may be included if greater model detail is desired:

1. Bozeman WRF effluent
2. Withdrawal to Buster Gulch irrigation diversion, located ~0.6 upstream of Site E (see **Figure 2-1**); flow only
3. Mouth of Hyalite Creek
4. Withdrawal to Dry Creek irrigation diversion, just downstream of model site 4 (flow only)
5. Mouth of Smith Creek
6. Mouth of Dry Creek
7. Mouth of Ben Hart Creek
8. Mouth of Story Creek
9. Mouth of Cowen Creek
10 Mouth of Gibson Creek
11. Return flow from Dry Creek irrigation diversion (just downstream of model site 5)
12. Mouth of Thompson Creek
13. Mouth of Bull Run Creek

It should be noted that prior to the field assessment, diurnal variation of the discharge of the wastewater from the Bozeman WRF should be considered. If flows from the WRF are significantly variable such that they alter the diurnal flow characteristics of the East Gallatin River itself, further discussions with the Department should be commenced about using a time-variable flow model necessary to represent these changes and their associated effect on water quality.

3.3 Other Data

In addition to the boundary conditions identified previously, forcing functions of air temperature, dewpoint, windspeed, and cloud cover are required to develop incoming PAR estimates and associated heat balances with QUAL2K. The Department has not taken the time to investigate whether suitable information is available from Gallatin Field (or other stations), but it is recommended that such information be assessed to determine availability as well as whether it is appropriate for the East Gallatin River corridor. If suitable information is not available, it is recommended that a meteorological station be placed nearby to measure these inputs for the model.

3.4 Numeric Nutrient Criteria Derivation Process via QUAL2K

A properly calibrated and validated QUAL2K model is necessary for nutrient criteria derivation. Basic criteria for determining when the model is calibrated and validated can be found in Suplee et al. (2006) and are further elaborated upon in Flynn and Suplee (2011). Numeric nutrient criteria can be ascertained by simulating incremental nutrient additions, or more likely in this case nutrient reductions, to the point where water quality standards (e.g., DO, pH), benchmarks (benthic algae density), or other ecological indicators are in compliance /achieved. Detailed discussions of this process are found in Section 13 of Flynn and Suplee (2011).

4.0 Can Beneficial Uses be Supported by Applying Greater Emphasis on Reducing One Nutrient?

The model described in **Section 3.0** can be used to answer certain questions regardless of whether or not the East Gallatin River is found to have nuisance algae levels or other undesirable water quality characteristics. If it is established that algae density is above benchmarks, the model can be used to explore “what if” scenarios, including “what if the city of Bozeman greatly reduced its TP load to the East Gallatin but only reduced its TN load somewhat?”

Figure 4-1 helps illustrates the concept. Taken from Flynn and Suplee (2011), **Figure 4-1** shows growth limitation factors (0-1 scaling factor) from nitrogen, phosphorus, or light at any given point along the river. The horizontal line nearest to the X-axis is the most-limiting factor.

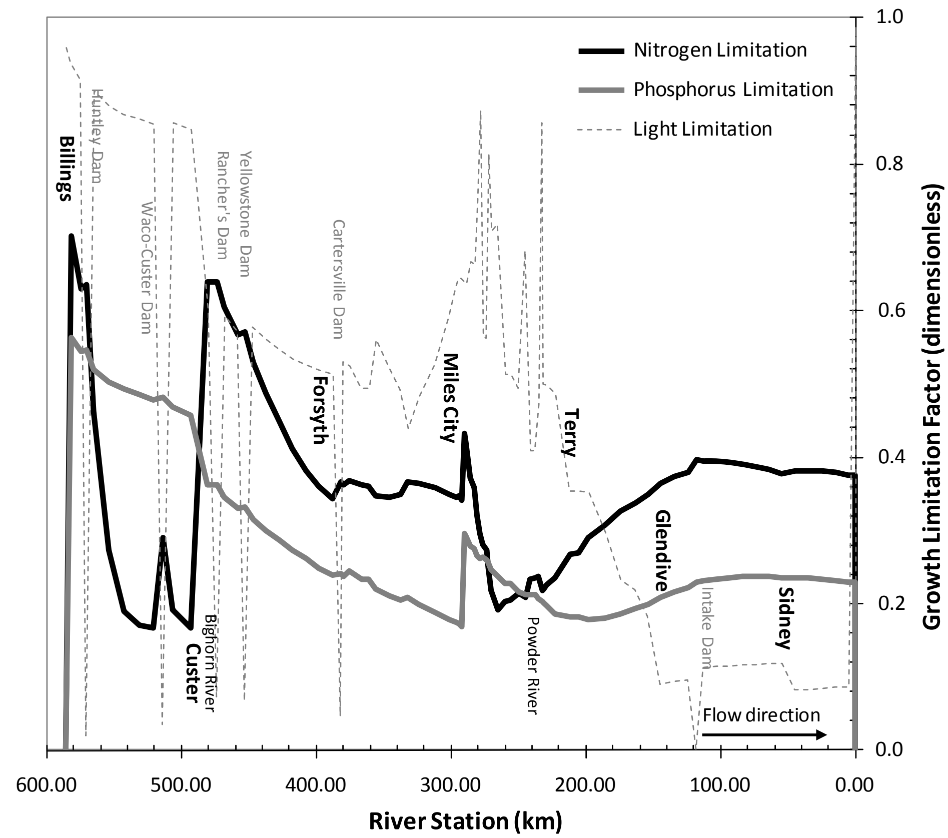

Figure 4-1. QUAL2K model results for nitrogen, phosphorus, and light limitation of benthic algae in the Yellowstone River. From Flynn and Suplee (2011).

What can be ascertained from **Figure 4-1** is that in the case of point-source inputs, the nutrient limitation term can greatly change. In this example, nitrogen limitation is strong downstream of the city of Billings for some distance due to phosphorus load additions from the Billings WWTP (note: the nitrogen load is also large, but the phosphorus load evidently has a much stronger effect because it leads to river phosphorus concentrations far above saturation levels for benthic algae). But the nitrogen-limitation status then changes due to external conditions. So within a model, questions can be posed such as: (1) "What if the Billings TP load were to be greatly reduced such that phosphorus could be made limiting (or co-limiting) with nitrogen?", (2) "What effect would this have on benthic algae levels in the immediate vicinity of the wastewater discharge?", and (3) "What would be the effect further downstream?".

In the case the East Gallatin River, such an exercise would greatly help us understand if a greater reduction in WRF phosphorus (the less expensive nutrient to eliminate) would achieve benthic algae targets by pushing the East Gallatin to P limitation. The model could also be used to see the downstream effects. We know that Hyalite Creek introduces naturally-elevated TP concentrations; in all probability, any TP limitation achieved further upstream would there be lost. The model could also show how changes to WRF treatment systems affect benthic algae. Model results may possibly indicate that a substantial reduction in TN from the WRF is necessary so that nitrogen limitation (and beneficial uses) can be maintained below the Hyalite Creek confluence. Again, the main point is that with the QUAL2K model “what if” scenarios can be evaluated.

5.0 Status Monitoring

If reach specific criteria are developed and it appears that downstream uses will be protected, and those criteria are moving towards adoption by the Board of Environmental Review, the last step in the process is status monitoring. The state-of-the-art in both mechanistic and empirical models is such that they inherently have noise, and confirmation of use-support of the reach-specific criteria is needed to assure stream protection. It is recommended that model sites 1 through 6 be used for this purpose regardless of the method used (mechanistic model or empirical model) to develop the criteria. Data collection should focus on the endpoints of concern (benthic algae density, macroinverebrate metrics, diatom metrics), and (if QUAL2K modeling was used) other endpoints (like pH) that were used in developing the criteria. Presuming that the criteria can be met by changes to the WRF alone, then, after upgrades occur, five years continuous monitoring is recommended at a minimum, to be carried out by the city or its consultants. Five years will also allow enough time to apply robust non-parametric trend statistics to the dataset (Helsel and Hirsch, 2002). Models developed via the methods outlined in **Sections 2.6 and 3.0** may show that, due to nonpoint source contributions, an upgrade to the WRF cannot in and of itself achieve the reach-specific criteria. In this case, the Department and the city should discuss how to proceed with status monitoring. TMDLs for nonpoint source cleanups or application of BMPs generally recognize that implementation will take years (5+), and this should play an important role in determining the monitoring status timeline.

6.0 Budget Estimates

An estimate was made for the cost to complete the data collection and analysis for each of the three major aspects discussed: (1) the biological characterization, followed by either (2) empirical statistical modeling or (3) QUAL2K modeling. Estimates shown are total, that is, the grand total to complete each task including development, calibration, and validation of the models, and any criteria developed thereof. Status monitoring, which would occur afterwards, is not included. Cost estimates were based on 2012 analytical laboratory price sheets, costs for purchasing small equipment or rental of large equipment, etc. They should be viewed as estimates only, as best professional judgment was needed to estimate hours of labor for field data collection, professional data analysis and modeling, etc. See **Appendix A-1** for details.

1. Biological characterization: $75,220

The following are additional costs to be added to that above in order to complete the task:

A. Empirical Model Approach: $30,900

B. QUAL2K Model Approach: $113,635

If the empirical approach is taken, the grand total (biological characterization plus the empirical statistical model) is $106,120. If the minimized study (sites B, D, F, G, H, I and J only) is selected for the empirical approach, which again includes the biological characterization, the grand total drops to $75,853. If the mechanistic model approach using QUAL2K is taken, the grand total (biological characterization plus the calibrated and validated model) is $188,855. If the minimized study (sites B, D, F, G, H, I and J only) is selected for the biological characterization, the grand total for the QUAL2K model approach drops to $168,500.

7.0 Next Steps

This document has outlined the basic conceptual framework for (a) characterizing the biological and water-quality status of the East Gallatin River (**Section 2.0**), (b) using empirical methods to derive the criteria (**Sections 2.6**), (c) using mechanistic modeling approaches to derive the criteria (**Section 3.0**), (d) consideration of downstream effects (**Sections 2.7 and Section 4.0**), and (e) biological status monitoring (**Section 5.0**). This document provides several pathways and options to study and model the East Gallatin River.

If work outlined in this document is to be undertaken, the next logical step would be to develop a detailed SAP. Potentially, a Quality Assurance Project Plan (QAPP) may need to be developed, but that document may be optional so long as Department SOPs are closely adhered to and the SAP provides sufficient detail on topics that are not specifically covered in DEQ SOPs. Further discussion with the Departments Quality Control Officer (Mindy McCarthy; MMcCarthy3@mt.gov) should clarify if a QAPP is needed to further support field sampling. If reach-specific criteria are found to be needed and the QUAL2K model is going to be used, it would be worth further consultation with the Department on a QAPP specific to the model as well as discussions with Department staff during model development.

8.0 References

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Appendix A-1

1. Biological Characterization (2-year study, up to three months per summer). This work is undertaken regardless of preferred modeling approach.

	Benthic Algae (Chl*a*)		Benthic Algae (AFDM)		Macroinvertebrates		Diatoms		WQ (nutrients, TSS)*		Herbicides**	
SITE	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample
A	6	$1,170	6	$300	4	$980	2	$500	6	$960.00	5	$750
B	6	$1,170	6	$300	4	$980	2	$500	6	$960.00	5	$750
C	6	$1,170	6	$300	4	$980	2	$500	6	$960.00	5	$750
D	6	$1,170	6	$300	4	$980	2	$500	6	$960.00	5	$750
E	6	$1,170	6	$300	4	$980	2	$500	6	$960.00	5	$750
F	6	$1,170	6	$300	4	$980	2	$500	6	$960.00	5	$750
G	6	$1,170	6	$300	4	$980	2	$500	6	$960.00	5	$750
H	6	$1,170	6	$300	2	$490	1	$250	6	$960.00	5	$750
I	6	$1,170	6	$300	2	$490	1	$250	6	$960.00	5	$750
J	6	$1,170	6	$300	2	$490	1	$250	6	$960.00	5	$750
Totals:		$11,700		$3,000		$8,330		$4,250		$9,600		$7,500

Subtotals, analytical costs:	$44,380			
YSI 6600 Sonde Rental:	$2,240	Assume 2 sondes, deployed for 1 week each summer for two summers ($560 X 2 X 2).	* TSS	$20.00
Purchase YSI 85	$1,350	For instantaneous DO, temperature, and conductivity. Separate low-cost pH meter can be purchased.	TN	$40.00
Labor in field:	$14,250	Assume a field team of 2 people, 10 sites, 3 hrs/site, average of 4.75 trips per site (for both years), assume $50/hr.	TP	$30.00
Data analysi:s	$10,000	Assume 1 person, contracted, professional environmental consulting firm	SRP	$30.00
Misc. supplies:	$3,000	macroinvertebrate nets, filters, filter apperatus, vehicle gasoline, etc.	nitrate + nitrite	$25.00
GRAND TOTAL, Biological Characterization:	**$75,220**		total ammonia	$15.00
				$160.00

Analytical (min sites)	Field labor (min sites)		
$28,300	$9,975	GRAND TOTAL, min. sites (B, C, F, G, H, I, J):	$54,865

**N, P, and S containing pesticides (Method E507 modified).

2. Statistical Empirical Model (One additional year of data in additional to the biological characterization).

	Benthic Algae (Chl*a*)		Benthic Algae (AFDM)		Macroinvertebrates		Diatoms		WQ (nutrients, TSS)*		Herbicides	
SITE	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample
A	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
B	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
C	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
D	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
E	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
F	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
G	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
H	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
I	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
J	3	$585	3	$150	2	$490	1	$250	3	$480.00	2	$300
Totals:		$5,850		$1,500		$4,900		$2,500		$4,800		$3,000

Subtotals, analytical costs:	$22,550	
YSI 6600 Sonde Rental:	$560	Assume 1sondes, deployed for 1 week for 1 summers ($560 X 1 X 1).
Labor in field:	$6,990	Assume a field team of 2 people, 10 sites, 3 hrs/site, average of 2.333 trips per site, assume $50/hr.
Data analysi:s	$15,000	Assume 1 person, contracted, professional environmental consulting firm. This would be final report and emperical model development
Misc. supplies:	$800	macroinvertebrate nets, filters, filter apperatus, vehicle gasoline, etc.
Year 3 Total:	$30,900	
Emperical Model, TOTAL ‡ **:**	**$106,120**	

Analytical (min sites)	Field labor (min sites)		
$14,735	$4,893	Year 3 Total, min. sites (B, C, F, G H, I, J):	$20,988
		Emperical Model, TOTAL, min sites (B, C, F, G, H, I, J) ‡ **:**	$75,853

3A. QUAL2K Model main sites (data in addition to data from the biological characterization). **Assumes a single year sampling in Aug and Sept.**

	Benthic Algae (Chl*a*)		Benthic Algae (AFDM)		Phytoplankton Chl*a*		Nutrients*		TSS, ISS, Alk, Hardness, TOC†		$CBOD_{20}$	
SITE	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample
1 (same as A)	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
2 (same as D)	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
3 (same as G)	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
4 (same as H)	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
5 (same as I)	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
6 (same as J)	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Totals:		$2,340		$600		$780		$1,260		$720		$720
							*TN	$40.00		†TSS	$20	
							TP	$30.00		ISS	$20	
							SRP	$30.00		alkalinity	$10	
							nitrate + nitrite	$25.00		hardness	$20	
							total ammonia	$15.00		TOC	$35	
							total nutrients:	$140.00		*total WQ:*	$105.00	

3B. QUAL2K Model, Additional Sites. Assumes a single year sampling in Aug and Sept.

	Benthic Algae (Chl*a*)		Benthic Algae (AFDM)		Phytoplankton Chl*a*		Nutrients*		TSS, ISS, Alk, Hardness, TOC†		$CBOD_{20}$	
Additional Sites	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample	Frequency	Cost/sample
(two flow sites)												
Bozeman WRF	0	$0	0	$0	0	$0	3	$420.00	3	$315	3	$180
Hyalite Cr mouth	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Smith Cr mouth	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Dry Creek mouth	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Ben Hart Cr mouth	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Story Cr mouth	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Cowen Cr mouth	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Gibson Cr moutn	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Dry Creek Irrig. return	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Thompson Cr mouth	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Bull Run Cr mouth	2	$390	2	$100	2	$130	2	$280.00	2	$210	2	$120
Totals:		$3,900		$1,000		$1,300		$3,220		$2,415		$1,380

Subtotals, analytical costs:	$19,635	
YSI 6600 Sonde Rental:	$10,800	Assume 6 sondes, deployed for 2 weeks in Aug and 2 weeks in Sept ($1800/month X 6).
Labor in field:	$12,000	Assume a field team of 2 people, 16 sites, 3 hrs/site, average of 2.5 trips per site (for both months), assume $50/hr. Assume flow meter provided by consultant.
Hobo Weather Station:	$1,200	
Data analysi:s	$65,000	To build calibrated and validated model, professional environmental consulting firm with expertise in QUAL2K modeling
Misc. supplies:	$5,000	vehicle gasoline, filters, syringes, Aquarods, etc., contingencies
QUAL2K Model, TOTAL:	**$113,635**	

From: Suplee, Mike
To: Greeley, Carrie
Subject: FW: EPA comments on Circular DEQ-12A
Date: Tuesday, April 01, 2014 4:17:28 PM

From: Laidlaw, Tina [mailto:Laidlaw.Tina@epa.gov]
Sent: Tuesday, April 01, 2014 4:15 PM
To: Johnson, Elois
Cc: Mathieus, George; Suplee, Mike; North, John; Perkins, Erin
Subject: EPA comments on Circular DEQ-12A

Elois,

EPA's comments on Circular DEQ-12A (numeric nutrient standards) are attached and a hard copy is being mailed out today. Thank you for the opportunity to comment.

Tina Laidlaw

Tina Laidlaw
U.S. EPA, Montana Office
10 West 15th St., Suite 3200
Helena, MT 59626
phone: (406) 457-5016

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 8
1595 Wynkoop Street
DENVER, CO 80202-1129
Phone 800-227-8917
http://www.epa.gov/region08

APR 1 2014

Ref: EPR-EP

Montana Board of Environmental Review
Robin Shropshire, Chairman
Montana Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

Subject: EPA Comments on Montana's Nutrient Standards (Subchapters 2, 5, 6 and 7; Circular DEQ-12A)

Dear Ms. Shropshire:

This letter provides the comments of the Environmental Protection Agency (EPA) Region 8 on Montana Department of Environmental Quality's (MDEQ) draft nutrient rules contained in: 1) Circular DEQ-12A; and 2) Subchapters 2, 5, 6 and 7, ARM 17.30. The EPA received the public notice, published on February 14, 2014. The notice includes information about the proposed water quality standards changes, public hearing information, and invites public comment. The EPA has reviewed the proposed standards and recommends adoption to the Board of Environmental Review (Board).

MDEQ has spent the last decade developing the scientific rationale behind the proposed numeric nutrient criteria for wadeable streams to ensure they are protective of designated uses. The criteria proposed in Circular DEQ-12A and the associated technical documentation[1,2] reflect the MDEQ's commitment to develop scientifically defensible criteria through many years of research and data collection; incorporation of stakeholder and peer review comments; and review of the scientific literature.

The EPA has worked collaboratively with the State to ensure that not only are MDEQ's criteria protective of applicable designated uses and based on sound scientific rationale but also that the State's general and individual variance approaches are consistent with the Clean Water Act and the EPA's implementing regulations. As a general matter, the EPA supports the use of variances,

[1] Suplee, M.W, V.Watson, A.Varghese and J. Cleland. 2008. Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers. Helena, MT: Montana Dept. of Environmental Quality.
[2] Suplee, M.W., and V. Watson, 2013. Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers—Update 1. Helena, MT: Montana Dept. of Environmental Quality.

as appropriate and consistent with 40 CFR §131.10, to provide time to meet designated uses in certain situations. MDEQ's variance approaches will allow the State and its stakeholders time to implement a phased approach to improve water quality, while retaining the currently applicable designated uses as the long-term goal for the State's rivers and streams. The EPA would like to note that we have also submitted comments to MDEQ on the proposed variance rules and implementing guidance.

We have reviewed the following new water quality standards and support their adoption:

- Proposed numeric total nitrogen and total phosphorus criteria for wadeable streams;
- Proposed numeric total nitrogen and total phosphorus criteria for the Yellowstone River (Bighorn River confluence to the stateline); and
- Proposed reach specific criteria for Flint Creek, Bozeman Creek, Hyalite Creek and the East Gallatin River.

We support MDEQ's recommendation to delay rulemaking for proposed Flathead Lake criteria. The additional time will allow MDEQ to: 1) obtain and analyze the available data for the lake; 2) develop a robust rationale documenting the linkage between the proposed criteria and the designated uses for the lake; and 3) continue collaboration and outreach to local stakeholders and tribal partners. We look forward to working with MDEQ on future rulemaking efforts for Flathead Lake criteria.

Please note that the positions described in our comments, regarding both existing and proposed water quality standards, are preliminary in nature and should not be interpreted as final EPA decisions under Section 303(c) of the Clean Water Act (CWA). If there are questions concerning our comments, please contact Tina Laidlaw (406-457-5016).

Sincerely,

SW Spence

Sandra Spence, Chief
Water Quality Unit

From: Frank Stewart
To: DEQ WQP Admin
Subject: comments regarding discharge variances
Date: Friday, April 04, 2014 11:08:24 AM

My letter of comments is attached.

Sincerely,
Frank Stewart

/\/
Frank Stewart, PE
Stewart Engineering
3250 Prairie Smoke Road
Bozeman, MT 59715
Phone: 406-586-0790
Cell: 406-539-0954
email: fstewart@latmt.com

Stewart Engineering

Frank M. Stewart, P.E. ♦ **email fstewart@latmt.com**
3250 Prairie Smoke Road ♦ **Bozeman, MT 59715** ♦ **Phone (406) 586-0790**

April 4, 2014

Ms. Carrie Greely
Dept. of Environmental Quality
1520 E. Sixth Ave
PO Box 200901
Helena, MT 59620-0901
deqwqpadmin@mt.gov

Subject: Comments related to proposed variances for wastewater dischargers

Dear Ms. Greely:

I have reviewed Circular DEQ-12B and the related proposed New Rules 1 – 5 Nutrient Standards Variances. I am in favor of the goals of DEQ12A (tightening discharge standards) and also in favor of the goals of DEQ12B (allowing temporary variances in cases where achieving the new standards is not currently practical).

In cases where the DEQ is considering the allowance of general or individual variances, I think it would be reasonable for the DEQ to require some additional treatment to be initiated at these facilities in order to improve the quality of the discharge water, even if the new standards cannot be fully achieved in the short term. Possible additional treatment might include floating treatment wetlands, constructed wetlands, and "treatment-in-a-box" add-on systems.

I have extensive experience related to basic research and product development of floating treatment wetlands for wastewater treatment (I have served as Principal Investigator and Co-PI on two State-sponsored grants with Floating Island International, Inc., and I am a shareholder in the corporation). I believe that floating treatment wetland products are a viable alternative for enhanced treatment, particularly at facilities that need to optimize nutrient reduction within existing infrastructures. I encourage the DEQ to carefully consider the use these products when reviewing requests for variances. Additionally, since floating treatment wetlands are considered to be an emerging technology in Montana, I think it would be appropriate for the DEQ to encourage their implementation by helping communities obtain funding to install these systems.

Please let me know if you have questions or require additional information related to wastewater treatment with floating treatment wetlands.

Sincerely,

From: Guy Alsentzer
To: DEQ WQP Admin
Subject: Re-send of Nutrient Rule Pkg Comments
Date: Tuesday, April 01, 2014 5:03:15 PM
Attachments: Upper Missouri Waterkeeper Logo.jpeg

I received a bounce-back of my previous, timely submission of comments on the state's Nutrient Rule Package. I send those comments again, now.

Sincerely,
GA

Guy Alsentzer, Esq.
Upper Missouri WATERKEEPER® | Executive Director
Upper Missouri Waterkeeper, Inc. | P.O. Box 128, Bozeman, Montana 59771
406.570.2202 | Guy@uppermissouriwaterkeeper.org
www.uppermissouriwaterkeeper.org

April 1, 2014

Submitted via electronic mail to deqwpadmin@mt.gov & ejohnson@mt.gov

Carrie Greeley & Elois Johnson
Dept. of Environmental Quality
1520 E. Sixth Ave.
Helena, Montana 59620

Re: Combined Comments in Support of Montana's Proposed Numeric Nutrient Rule Package

On behalf of the Upper Missouri Waterkeeper, Guy Alsentzer, and the supporting water advocacy organization, Upper Missouri Waterkeeper, Inc., please accept the following combined comments addressing proposed rules from both the Board of Environmental Review ("BER") and Department of Environmental Quality ("DEQ"), commonly referred to as Montana's 'numeric nutrient rule package.' Public comments concerning the proposed rule package ("Rule Package") are due by April 1, 2014.

Upper Missouri Waterkeeper, Inc. ("UMW") is a non-profit membership organization dedicated exclusively to protecting and improving the ecological and aesthetic qualities of Southwest and West-central Montana's Upper Missouri River Basin. As part of its mission UMW engages in policy, science and rulemaking related to Montana's implementation of its Clean Water Act duties and citizens' guarantee to a clean and healthful environment under our constitution. We thank the BER and DEQ for the opportunity to comment on these proposals and participate in the lengthy public participation process associated with the proposed rulemaking.

Executive Summary

UMW supports the state of Montana's movement towards adopting more protective water quality standards. The proposed rulemakings are a needed update to Montana's oversight of water quality, and are particularly necessary in light of evolving land uses, population growth, and best available science. However, the proposed rulemakings also contain significant shortcomings and ambiguities that threaten Montana's implementation of its Clean Water Act duties and will potentially lead to unintended, adverse consequences for water quality and communities.

In these comments, we focus first on the BER's proposed numeric nutrient criteria, then on the DEQ's proposed nutrient standards variance rule. While these comments represent the majority of our concerns and suggested improvements, we also endorse comments written by the Clark Fork Coalition addressing the proposed rules and needed improvements that will better protect water quality and community health.

Introduction

Over forty years ago, Congress made the promise to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters.".[1] To that end, Congress established a national goal to *eliminate* discharges of pollutants into navigable waters by *1985*. Congress also set the national goal of achieving levels of water quality necessary to protect all human contact uses of the Nation's waters and quality necessary for the protection of fish, shellfish and wildlife by *1983*.[2]

Unfortunately, those promises and goals still await fulfillment. *See, e.g.,* EPA, *Nat'l Rivers and Streams Assessment* (Feb. 2013) where EPA reports that well over 50% of the waters assessed exhibited poor conditions and only 20% were classified as "good." The results by region were even more disappointing with 62% of the waters in the east classified as poor and 58% in the plains states. *See also*, EPA summary of states' reported water quality data at http://ofmpub.epa.gov/waters10/attains_nation_cy.control, showing that states have a poor record of assessment, but of the waters assessed, 53% of assessed rivers and streams, 68% of assessed lakes, and 66% of assessed bays/estuaries are *failing* to meet one or more water quality standards. Discharges of pollutants into our nation's water have not been eliminated. Almost thirty years after the stated deadline, the nation still uses its waters as disposal sites for a vast number of pollutants, and while there has been improvement, many waters still fail to meet basic requirements for being "fishable and swimmable."

Montana's proposed numeric nutrient rule package is a positive step in advancing the promises of the Clean Water Act in a few areas, but in others fall short of what is needed to address current problems with the development and implementation of water quality standards. There are some components of the proposed rules that are contrary to advancement of the goals and requirements of the Clean Water Act and we urge the BER and DEQ to reconsider those portions of the rule package and include stricter requirements.

Comments Specific to Proposed Rules

The BER's notice of public hearing concerning its proposed adoption of base numeric nutrient standards laid out in Circular 12-A aptly describes the state of Montana's approach to its Rule Package:

> The nutrient criteria concentrations being proposed for adoption as standards are generally low, particularly in the western region of Montana. In many cases, the concentrations are below the limits of current wastewater treatment technology, particularly for nitrogen. Therefore, when little or no stream dilution is available, dischargers will find it difficult or impossible to meet the standards. Senate Bill 95 (2009 Legislature) and Senate Bill 367 (2011 Legislature), now codified at 75-5-313, MCA, addressed the high cost and technological difficulties associated with meeting the nutrient standards in the short term. Section 75-5-313, MCA, allows dischargers to be granted variances from numeric nutrient standards, once the criteria have been adopted as standards, in those cases where meeting the standards today would be an unreasonable economic burden or technologically infeasible. Variances from the standards may be

[1] 33 U.S.C. §1251(a)

[2] 33 U.S.C. § 1251(a)(1) and (2).

granted for up to twenty years. Thus, 75-5-313, MCA, allows for the nutrient standards to be met in a staged manner, over time, as alternative effluent management methods are considered, nutrient removal technologies become more cost-effective and efficient, and nonpoint sources of nutrients are addressed

MAR Notice No. 17-356

I. THE BER'S PROPOSED BASE NUMERIC NUTRIENT STANDARDS

A. Department Circular 12-A

Overall we support adoption of the BER's proposed numeric nutrient standards and provide the comments below concerning certain outstanding questions. We anticipate that the BER and DEQ will thoughtfully consider and implement some of the practical changes noted and, in so doing, protect water quality with a strong, scientifically-sound final rule.

We strongly support the scientific basis for numeric nutrient criteria. Similarly, we support the decision to use ecoregion and geographic parameters in setting applicable WQS as these decisions reflect the best available science and a practical understanding of natural water quality conditions that should, and must, be protected.

B. Proposed 17.30.516 – "Standard Mixing Zones for Surface Water"

We agree that mixing zones are sanctioned water quality control tools under the Clean Water Act and as such understand the amendments to this section. However, we are concerned that the movement from the 7Q10 flow standard to the 14Q5 standard is a poor policy choice, regardless of the fact that both measurements are utilized in water quality modeling. Put simply, a 14Q5 low flow standard for a mixing zone in effect measures pollutant loading using the entire flow of a receiving water, instead of a fraction thereof as in the 7Q10 approach. The BER's notice in MAR 17-356 explains this decision as reflective of the non-toxic nature of nutrients in concentrations within Circular 12-A. We do not agree that this standard is appropriate based on two troubling inconsistencies and urge the BER to consider reapplication of the 7Q10 standard.

First, at a policy level, we believe there is ample reason to keep the 7Q10 as it represents a sound, scientifically-based standard that is more protective of receiving water quality than the 14Q5. Insofar as the state of Montana is taking significant steps forward in protecting its waters from nutrient pollution, it should not in the substance of new rules take a step backwards by functionally relying on lesser flow standards. As the saying goes, a model – and results thereof – are only as good as the data it relies upon. Here we urge BER to take a firm stance in line with the precautionary principle and keep the 7Q10 flow standard for its mixing zone calculations.

Second, although the concentrations of nutrients anticipated as discharges under Circular 12-A may be non-toxic, higher nutrient concentrations can and do become toxic. Indeed, DEQ Circular-7 at page 51 recognizes that the three primary incarnations of nitrogen are in fact toxic at certain concentrations and sets appropriate aquatic life and human health criteria. Insofar as the context here is the adoption of more stringent numeric nutrient criteria, and the fact remains that using a 14Q5 flow standard in a mixing zone equates to a less-protective modeling and that nutrients can be toxic – particularly as they possess an accumulative nature in water columns – there is the possibility that using the 14Q5 standard may create unintended, adverse consequences for downstream water quality. Hypothetically, allowing greater discharges of nutrients in this rule which result in nonattainment of downstream water quality triggers the Clean Water Act's antidegradation prohibition.

Indeed, EPA has described a mixing zone as "an allocated impact zone in the receiving water which may include a small area or volume where acute criteria can be exceeded provided there is no lethality (zone of initial dilution), and a larger area or volume where chronic water quality criteria can be exceeded if the designated use of the water segment as a whole is not impaired as a result of the mixing zone."[3] EPA policy "recommends that mixing zone characteristics be defined on a case-by-case basis after it has been determined that the assimilative capacity of the receiving system can safely accommodate the discharge."[4]

Moreover, emphasizing the site-specific nature of the evaluation, EPA states that the "assessment should take into consideration the physical, chemical, and biological characteristics of the discharge and the receiving system; the life history and behavior of organisms in the receiving system; and the desired uses of the waters."[5] Mixing zones should be authorized with care, according to EPA, "so as to not impede progress toward the Clean Water Act goals of maintaining and improving water quality."[6]

Here, the rule proposes uniform, relaxed mixing zone standards. Insofar as the rule only contemplates a revised, general nutrient mixing zone standard and refuses to consider site-specific mixing zones as per the EPA guidance noted above, we believe the 14Q5 is therefore too lax. Rather, the 7Q10 appears a well-traveled – if not adequately site-specific - path for modeling assimilative capacity within mixing zones while remaining reasonably protective of downstream water quality. Therefore we again urge BER to reconsider the propriety of movement towards the 14Q5 flow standard and instead stick with the proven, protective 7Q10 flow.

C. Proposed 17.30.715 – "Criteria for Defining Non-Significant Changes in Water Quality"

We do not agree with the inclusion of inorganic nitrogen and inorganic phosphorus as pollutants that are per se deemed to not cause changes that exceed trigger values set in Circular 7. It is a fact that most new dischargers of nutrients are subdivisions. By establishing a non-significance criteria for the above types of nutrients the rule creates a means by which new subdivision contributions of nutrients escape substantive review and the adage 'death by a thousand cuts' can move from proverb to reality. E.g. if new discharges and their contributions of pollutant loading are continually deemed insignificant, yet cumulatively pollutant loading increases, the purposes of the Clean Water Act are defeated as waters are in fact receiving more pollution, not less, based on a statutory sleight of hand. We urge BER to amend its nonsignificance criteria to reflect the reality that additions of nutrients in quantity, time and strength do in fact constitute significant changes in water quality.

II. THE DEQ'S PROPOSED NUMERIC NUTRIENT STANDARDS VARIANCES

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[3] Memorandum from Robert Perciasepe, Assistant Administrator, to Water Program Directors, EPA Guidance on Application of State Mixing Zone Policies in EPA-Issued NPDES Permits, August 6, 1996.

[4] EPA, Water Quality Standards Handbook – Second Edition, 1993 at 5-1.

[5] *Id.*

[6] *Id.* at 5-2.

As stated previously, BER and DEQ have proposed a rule package that has some environmentally-protective attributes but also some provisions that meet its needs for administrative convenience and protection of permittees. In doing so, it has provided less protection for water quality than the law requires. As it is likely that many permittees will be seeking variances over the years to come, and likely for many permitting cycles into the future, this rule needs to be clear and ensure the highest level of public health protection.

A. The Clean Water Act's Water Quality Standards Requirements

Development and implementation of water quality standards (WQS) are critical components of the Clean Water Act, with shared roles for states and EPA. The Clean Water Act imposes an initial obligation on states to develop water quality standards as necessary to protect designated uses of each state's waters. States must designate uses, which are the uses that existed in 1974 *or better* if waters have improved. Those standards are to be submitted to EPA for approval. If a state fails to develop adequately-protective standards, the Clean Water Act requires EPA to step in and develop the standards.[7] States are also required to adopt and implement meaningful antidegradation protections to ensure that waters that are meeting water quality standards are not allowed to degrade and that high quality waters (water quality that is better than standards) retain their high quality.[8]

A state must, not less than once every three years, hold public hearings for the purpose of reviewing applicable water quality standard and, where appropriate and necessary to meeting the requirements of the Clean Water Act, modify and adopt new standards.[9] ("Triennial Review"). Again, EPA is to review and approve water quality standards from a state as part of the Triennial Review. Where a state fails to modify or adopt standards that stay abreast of scientific and technical developments and as necessary to ensure protection of waters, EPA must and can step in and develop appropriately-protective standards for the state.[10] With these requirements, one sees the "forward motion" for water quality that is dictated by the Clean Water Act: set standards for quality that will dictate cleanup and set antidegradation requirements to preserve what is already clean.

The Clean Water Act then dictates that permitting will serve an important role in implementation of these requirements and maintaining 'forward motion' by being the tool for eliminating discharges of pollutants. Discharges without a permit are prohibited. Permits must include limitations on discharges as necessary to ensure: they will not cause or contribute to a violation of water quality standards; that discharges conform to effluent limitation guidelines; and that discharges conform to any Total Maximum Daily Load (TMDL) requirement that sets pollutant limits in order to meet water quality standards.[11] Compliance plans may be utilized where necessary to ensure that new technologies that may be necessary to meet the ever more stringent limits can be designed, built, and implemented in a timely fashion.

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[7] 33 U.S.C. § 1313(c)(3) and (4).

[8] 40 C.F.R. § 131.12.

[9] 33 U.S.C. §1313(c)(1).

[10] 33 U.S.C. § 1313(c)(4).

[11] 40 C.F.R. § 122.44.

As is plain from the structure of the Clean Water Act, efficient EPA oversight of Montana's role in this Rule Package is necessary to the full functioning of the Act. If Montana and EPA are not strong in their respective commitments to clean water, the entire structure of the Clean Water Act, and the promises that the Act is designed to fulfill, are in jeopardy. It is with that foundation, and those concerns, that we examine the Rule Package's heavy reliance on variances from its new, sound numeric standards.

B. Montana's Proposed Variance Standard Rule Assumes the Necessity and Categorical Availability of Variances and, In So Doing, Fails to Address the Larger Problem With Variances Under the Clean Water Act

We support the state of Montana's proposal to enact strong, protective numeric nutrient water quality standards and recognize that, in extreme circumstances, variances may be appropriate to usher dischargers into compliance. However, the proposed widespread use of variances raises serious Clean Water Act implementation and incompatibility problems that Montana must address before finalizing this portion of the Rule Package.

1. Department Circular 12-B

Proposed DEQ Circular 12-B contains information about variances from the base numeric nutrient standards. We oppose the widespread, categorical use of variances as a water quality tool in this rulemaking and urge the DEQ to instead consider the use of alternative means of securing compliance with WQS, particularly the use of compliance schedules.

i. Rationale: 131.10(g) Factors & Analysis

The state's demonstration of 131.10(g)(6) and the instant variance rule in large part is predicated on the assumption that the only way to achieve compliance with new numeric nutrient standards is by mandating reverse osmosis treatment on all wastewater dischargers of nutrients. In turn, the state has relied on studies – now approximately half a decade old, to show that it is economically infeasible to move all dischargers to reverse osmosis. However, those original studies of the feasibility of dischargers meeting proposed numeric criteria only contemplated treatment approximate to Level 2 in its analysis of impacts to private business. They did not entail analyses of hybrid approaches where the implementation of reasonable alternatives and other effluent management options may create a practical, legal means forward to creating compliance with WQS.

For instance, a recent EPA report shows that treatment of phosphorus at a state of the science facilities in the Puget Sounds region have routinely achieved total phosphorus concentrations of 0.02 mg/L, and total nitrogen concentrations of 2 to 3 mg/L.[12] While perhaps not applicable to all dischargers nor by itself capable of all necessary reductions, this recent science shows that technology – aside from reverse osmosis – is capable of approaching reductions contemplated under the Rule Package. Indeed, when contemplated alongside other alternative effluent management mechanisms there exists a realistic probability that technology has evolved significantly since the DEQ's initial study years past.

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[12] Lubliner, B., M. Redding, and D. Ragsdale, 2010. Pharmaceuticals and Personal Care Products in Municipal Wastewater and Their Removal by Nutrient Treatment Technologies. Washington State Department of Ecology, Olympia, WA. Publication Number 10-03-004, at p. 25.

Similarly, the Rule Package's reliance on factor six of 131.10(g) is also incumbent on a showing of substantial and widespread economic impacts. Previous documentation by the DEQ has, to our understanding, grouped dischargers together by sector to fulfill such a showing. However, such demonstration towards receipt of a variance – in line with the spirit and intent of the CWA – should be made by each individual discharger, not aggregated by sector. Such site and facility-specific determinations conform best to existing EPA guidance and statutory factors.

The sixth factor at minimum requires acute attention to detail to avoid abuse. This factor can be used as a mechanism for a state to avoid applying controls necessary to meet water quality standards and to lessen protections for the water body. Again, at this point in the Clean Water Act's history, states should be employing controls necessary to meet water quality standards, at a minimum for point sources.[13] Presumably then, the sixth factor should be read to address only non-point sources of pollutants, primarily agriculture. Given that Montana does not regulate nonpoint agricultural pollution, it seems that the sixth factor should have no application here, in rulemaking concerning point-source dischargers of nutrient pollutants.

We find it extremely disconcerting that the sixth applicability factor for obtaining a variance has been broadened to a catch-all contemplated as acceptable for lessening protection for waters for nearly all point-source dischargers of nutrient pollutants. Instead, we believe there is a place for the state to incorporate a discussion of feasibility that is not defined in strictly monetary terms, but should include consideration of whether a technology or practice is actually available to address a water quality issue. If there is truly no available method, then perhaps attaining a water quality standard in three years is not "feasible" and a water will remain impaired until other adequate solutions are implemented aside from point source reductions.

ii. General Variances Should Not Be Contemplated

75-5-313(5) provides in relevant part that:

> [A] permittee who meets the requirements established in subsection (5)(b) may, subject to subsection (6), apply for a general nutrient standards variance. (b) The department shall approve the use of a general nutrient standards variance for permittees with wastewater treatment facilities that discharge to surface water: (i) in an amount greater than or equal to 1 million gallons per day of effluent if the permittee treats the discharge to, at a minimum, 1 milligram total phosphorus per liter and 10 milligrams total nitrogen per liter, calculated as a monthly average during the period in which the base numeric nutrient standards apply; (ii) in an amount less than 1 million gallons per day of effluent if the permittee treats the discharge to, at a minimum, 2 milligrams total phosphorus per liter and 15 milligrams total nitrogen per liter, calculated as a monthly average during the period in which the base numeric nutrient standards apply; or (iii) from lagoons that were not designed to actively remove nutrients if the permittee maintains the performance of the lagoon at a level equal to the performance of the lagoon on October 1, 2011.

Likewise, Table 12B-1 in Circular 12-B reiterates those same standards by which an applicant may secure a general variance, specifically through use of their discharge permit.

We oppose the legislative creation and the nutrient variance rule's use of general variances because: (1) as discussed throughout this comment letter variances – particularly

[13] 40 C.F.R. § 122.44(d).

general variances - are antithetical to the CWA's intent of eliminating discharges of pollutants; (2) there is insufficient technical or analytical records justifying commonalities across sectors; and (3) waterbody-specific baselines and ecological needs differ across the state. EPA's 1998 Advanced Notice of Proposed Rulemaking discusses the exacting elements required for a variance, none of which support the categorical grouping of dischargers and waterbodies across a state under a general variance.

Further, there is serious doubt as to whether general variances will assure that dischargers meet the highest attainable limit. Logically, the broader and less specific a general variance becomes – e.g. more inclusive of a sector across the state – the less assurance that the variance has certainty of bringing a discharger closer to meeting WQS as so mandated by the CWA. The state has yet to demonstrate that compliance with general variance performance levels assures that the highest use attainable for one discharger within a sector is the same as another.

Similarly, there has been no explicit demonstration that proposed variances are protective of the aquatic life community that is expected in the receiving stream, vis-à-vis recognition of the role that antidegradation policy plays in implementation of WQS. We have not seen such an affirmative recognition expressed in the Rule Package. We also note that the emphasis on economic impracticability in this process appears to have stunted discussion of treatment more advanced than that required by Sections 303(c)(2)(A) and (B) of the CWA, and discussion of alternative effluent control strategies. For instance, non-point source controls have largely been ignored in the rulemaking although they continue to be a proportionally significant contributor to compliance with WQS. In fact, instead of addressing an alternative pollution control strategy like state based non-point source controls the Rule Package only contemplates protecting economic interests.

We believe ample EPA guidance, statutory authority and caselaw stands for the proposition that the scope of a variance must be both discharger and water body specific, and that a variance should also be pollutant-specific; it should extend for the shortest distance possible in the water body[14] and must be decided and supported with a full record, on a case by case basis. Montana should not entertain the use of the purported "general variance" that are allowed for an entire water body or an entire region or state for a nutrient pollutant.

iii. Reliance Upon Individual Variances Is Permissible If Done In A Judicious, Sparing Manner

We accept that, in certain circumstances, implementation of the proposed numeric nutrient criteria will create incredible economic hardship for small towns in Montana. These limited, rare circumstances are places where we believe that variances may have a specific, temporally-short role in moving a discharging facility towards compliance with WQS. As such we do not oppose the use of individual variances in the Rule Package with the caveat that all the other related indicia showing movement towards technological improvement and attainment of WQS are present. On the whole the mechanisms provided in Circular 12-B provide an adequate starting point for use of individual variances and support the continued ability to "re-open" and amend variances and water quality standards during triennial reviews.

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[14] This is basically consistent with EPA guidance now, but it is abused and Montana will be well-served to make that clear in this rule. This also points up the fact that variances aren't really necessary—mixing zones do the same thing. One or the other of these "anti-Clean Water Act" concepts should be eliminated.

However we still believe that, generally, variances appear to be a water quality tool that has outlived its usefulness. We believe there is scant justification for the use(s) of variances as their use is, per se, inconsistent with the basic structure and requirements of the Clean Water Act. Variances appear to be nothing more than an off-ramp away from steadily improving water quality and meeting standards. Contrary to the oft-heard yet unsupported claim, variances are rarely an "aid" to states to meeting water quality standards but, more often, an excuse to avoid them. A discussion of water quality standards implementation and impaired waters – a hypothetical directly applicable to Montana - illuminates our concerns.

iv. Implementation Issues

Montana's Rule Package intends to use variances broadly to "temporarily" avoid compliance with water quality standards; however, doing so threatens to exacerbate (or possibly create) impairment situations by allowing more pollution over time making ultimate attainment of WQS lengthier and more difficult. It is a potentially self-defeating path that is the precise opposite of the Clean Water Act's goals and requirements. If dischargers need time to employ new technologies or methods to meet stricter permit limits, the use of compliance plans and schedules ensures they use that time to install aggressive pollution controls, without weakening standards.

In fact, variances can work against the very things many stakeholders in Montana claim might require time. For example, Montana's nutrient loading pollution problem is significantly related to non-point source loading and, therefore, providing point source dischargers with variances – which weakens water quality - provides a disincentive to moving quickly and aggressively to deal with water quality problems. Application of a 'safety valve' like variances simply derails the statutory process of identifying troubled bodies of water and getting to work on a plan for clean up. We urge the DEQ to rethink providing a broad array of variance uses in its Rule Package, and instead revise its variance rule to narrow their availability to very limited circumstances.

1. Variances & Discharging to Impaired Waters

As aforementioned, states must set water quality standards to protect designated uses. Montana's movement towards incorporating the best available science and, in turn, more protective numeric water quality standards for nutrients is a positive step forward! However, in some instances standards are plainly not being met. Where water quality standards are not attained, a state must report this fact to EPA and the water is added to a § 303(d) or impaired water list.[15] Once on the list, the water body is in the queue for preparation of a clean-up plan--a Total Maximum Daily Load ("TMDL") plan. States have a significant amount of time to prepare and finalize TMDLs. A TMDL sets a Waste Load Allocation ("WLA") which assigns specific load limits to specific point source discharges. In setting the WLA, a state has determined that these are the discharge limits necessary to return the water to meeting water quality standards (along with whatever reductions have been assigned to the Load Allocation ("LA")). If the WLA's do not meet that definition, then the TMDL is deficient and must be redone.

Similarly, if the WLA and LA reductions are expected to take an extremely long time it could be argued that the TMDL is deficient because it is impossible to say with any reasonable assurance that the reductions will actually occur, a requirement in EPA's TMDL guidance.

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[15] 33 U.S.C. § 1313(d).

Rather, as work on a water body progresses, states reassess and readjust a TMDL as necessary. The water body remains “impaired” in status (and thereby subject to the TMDL clean up plan) until it achieves water quality standards. This is the straightforward way that waters are to be cleaned up under the Clean Water Act procedures adopted by Congress. Water quality standards serve as the goal and guiding principle toward which the TMDL and it implementation must always be geared.

Likewise, point sources must have permits to discharge and those permits are to include effluent limitations and other provisions (for example compliance plans) to ensure that the permit is designed to not cause or contribute to violations of water quality standards. In a TMDL situation, a point source will have been assigned a wasteload allocation, a part of the TMDL with which point sources must comply. The point source’s permit must include limits as necessary to comply with the wasteload allocation. Again, compliance plans are a method to help point sources reach compliance over the course of a permit. *See also below.* We are concerned, however, by language in Circular 12-B proposing that a general variance’s terms take precedence over permit limits determined pursuant to a TMDL. Assuming that such a situation would essentially permit increased pollution greater than the permitted under the WLA, the proposed rule thus runs afoul of TMDL caselaw prohibiting discharges that would cause or contribute to a violation of WQS and the variance would be incompatible with the CWA.

Given the Clean Water Act’s structure, there is no need to countenance “variances” from water quality standards. We do not see why variances are seen as necessary to provide time to make progress towards attaining WQS. This implies, incorrectly, that the Clean Water Act imposes some sort of penalty on a state for failing to achieve water quality standards by a certain date. Regrettably, it does not. A variance does not “create” additional time; whatever time is genuinely needed to meet water quality standards, that time will be taken regardless whether the state adopts a variance.

The purported “time” issue is not a genuine problem. When a water body is added to a state impaired water list, it has likely been already impaired for some time. Once a water is on the list, states have ample time to prepare a TMDL for EPA approval. This is not the timeline for completing the TMDL and bringing the water into compliance with standards. This is just the period of time a state has to propose and finalize the cleanup plan. During that time, states should be working aggressively with point sources, at a minimum, to ensure that permits are meeting the requirements of 40 C.F.R. § 122.44(d) which will make the TMDL process easier. Once the TMDL is approved by EPA, there is nothing in the Clean Water Act requiring that the TMDL goals be met in some set period of “time.”

While it is true that a water body may not yet attain water quality standards even when the point sources implement their reductions, it simply means that the water will remain listed under 303(d) as impaired until standards are attained. That is how the law works. The claim that “long term” strategies necessitate variances to come into compliance is unfounded. The “long term strategy” is a TMDL - the clean up plan to meet water quality standards, not weaken them. There is no need to weaken protections with variances, even temporarily, for Montana’s waters under the existing structure.

v. Preferred Alternative – Compliance Schedules

Generally a compliance schedule is necessary when a new effluent limit is included in a permit either because of new effluent limit guidelines from EPA associated with new technologies or where a water quality based effluent limit requires stricter controls that had previously been imposed. It is understandable that a discharger will need some time to design, build, and pilot whatever new technology or processes might be necessary to meet stricter permit limits. However, five years should be adequate for any new technology or process. After all, the Clean Water Act dictates that discharges of pollutants should have been eliminated decades ago, a goal that is still very far away. Except in the most egregious of economic circumstances we do not see any reason for allowing the continued discharges of pollutants without work toward compliance with at least the latest control technologies or techniques within a permit term.

There is no right to pollute the nation's waters. Five years is ample and consistent with the five year structure for permits. EPA should be absolutely clear in this regard and we urge Montana to revise its proposed variance rule to (a) rely primarily on compliance schedules in lieu of variances as a mechanism for bringing dischargers into compliance with numeric nutrient standards, and (b) to include a five year limitation on compliance schedules. Here, DEQ is proposing variances for as long as twenty years. This time frame is wholly unacceptable because it is contrary to the concept of WQBELs and TMDLs.

C. DEQ's Proposed Variance Rule for Numeric Nutrient Criteria Should be Significantly Narrowed and Circumscribed to Ensure Their Use Does Not Defeat Proper Functioning of the Clean Water Act

Variances to water quality standards are currently allowed, and while the rule is plain they must be used sparingly, Montana should use this rulemaking effort to further limit their application, not broaden it.[16] We are greatly concerned by the proposed widespread use and the state of Montana's near exclusive reliance thereon in its Rule Package. Variances are water quality standards in their own right and as such, must be approved by EPA.[17] Variances must also be reviewed every three years in the required triennial review and the state must report to EPA on whether a variance is being retained and must justify its retention.[18] Variances are required to be as short as possible and during the course of the variance, the discharger must regularly demonstrate that reasonable progress is being made to attain water quality standards.[19] Variances are not appropriate for anything other than portions (generally small) of water bodies and they pertain only to a single discharge or possibly a small group of discharges into that reach. As with Use Attainability Analyses (UAAs), some conditions for a variance are more prone to abuse (such as where there is a stiff economic price to pay to return water to meeting WQS). It is *never* appropriate to grant a variance where standards can be attained with reductions on point and nonpoint sources, including elimination of discharges. Montana's proposed Rule Package entails many of these safeguards and for those that are included we are grateful and in turn support those sections. However, as discussed below there are still significant gray areas in the proposed Rule Package that need further work.

1. Variance Term

Montana's final variance rule must be clear that variances have a specific expiration date and that they are water quality standards and as such are subject to review every three years. We

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[16] 40 C.F.R. § 131.10.

[17] 33 U.S.C. § 1313(c) and 40 C.F.R. § 131.10(g) and (h). *See also* EPA Water Quality Standards Handbook, parts 2.7 and 2.8.

[18] *Id.*

[19] *Id.*

do not agree with the proposal of twenty-year terms, regardless of the alleged surety of review of variances alongside triennial reviews. Variances should in most instances not extend beyond three years—at most, they might extend for the length of a single permit term with a review at the three-year mark.[20] Renewal of a variance should be fully-justified at each three-year mark as again, they are highly contrary to Clean Water Act requirements and purposes and should be carefully monitored and generally disfavored.

Further, Montana should specify in its variance rule that a variance absolutely cannot be obtained if the water quality criterion can be achieved with either or a combination of technology-based requirements *and* aggressive permit requirements for best management practices such as low impact development for new development and retrofits. Montana should not promulgate rules that are a disincentive to consistent forward progress on improving water quality and meeting water quality standards.

2. Variances Cannot Be Allowed for New Sources

DEQ's proposed variance rule is silent as to whether variances will be allowed for new sources. As a matter of policy, the state should want new sources to either comply at the date of initial discharge or be subject to compliance schedules. With a compliance schedule, a permittee is held to a date certain to meet an effluent limit certain. Surely this is the standard to which Montana would want to hold new pollution sources.

We strongly maintain that DEQ's variance rule specify variances can never be an option for new or expanding discharges as such a concept is completely contrary to the requirements of the Clean Water Act and existing EPA regulation. EPA's regulations prohibit the issuance of an NPDES permit "when the conditions of the permit do not provide for compliance with the applicable requirements of the CWA, or regulations promulgated under the CWA" or "when the imposition of conditions cannot ensure compliance with the applicable water quality requirements of all affected states."[21] Specifically, EPA's regulations prohibit the issuance of an NPDES permit for a new discharge where the discharge may "cause or contribute to the violation of water quality standards."[22] In order for a discharge of the pollutant at issue to be allowed, the regulations require strict assurances that the receiving water can handle the new discharge and meet water quality standards and that specific plans are in place to ensure that it will be restored from its condition of impairment.

Specifically, EPA regulations require that:

> The owner or operator of a new source or new discharger proposing to discharge into a water segment which does not meet applicable water quality standards or is not expected to meet those standards even after the application of effluent limitations required by 301(b)(1)(A) and 301(b)(1)(B) of CWA and for which the State or interstate agency has performed a pollutants load allocation for the pollutant to be discharged, must demonstrate before the close of the [NPDES permit] public comment period that:

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[20] Here too, the facts demonstrate that this very same thing can be readily accomplished with compliance plans in permits. Variances don't really make sense and are just a duplicative off-ramp from compliance.

[21] 40 C.F.R. § 122.4(a), (d).

[22] 40 C.F.R. § 122.4(i).

(1) There are sufficient remaining pollutant load allocations to allow for the discharge; and

(2) The existing dischargers into that segment are subject to compliance schedules designed to bring the segment into compliance with applicable water quality standards.[23]

In *Friends of Pinto Creek v. U.S. E.P.A.*, 504 F.3d 1007 (9th Cir. 2007), *cert. denied*, 129 S. Ct. 896 (2009), the Ninth Circuit Court of Appeals held that without a plan to achieve water quality standards, a permitting agency cannot allow new discharges that will exacerbate the existing water quality standards violations. The court held that all existing dischargers must be subject to compliance schedules[24] and that "[i]f there are no adequate point sources to do so, then a permit cannot be issued unless the state or the [discharge permit applicant] agrees to establish a schedule to limit pollution from a nonpoint source or sources sufficient to achieve water quality standards."[25]

In other words, a TMDL is a necessary condition for a source to use the exception provided in EPA rules to the general prohibition on new sources into impaired waters but a TMDL by itself is not sufficient. Sources under compliance schedules are also necessary. Instructively, EPA's Great Lakes Initiative rules *prohibit* the application for variances to new or recommencing sources.[26]

3. Variances Must Comply With Antidegradation Policy

Tier I of the antidegradation policy, as framed by federal rules, applies to all water bodies regardless of their quality and requires a level of protection to assure that "[e]xisting instream water uses and the level of water quality necessary to protect the existing uses shall be maintained and protected."[27] Existing uses are defined as "those uses actually attained in the waterbody on or after November 28, 1975, whether or not they are included in the water quality standards."[28]

Existing provisions at ARM 17.30.705(2)(a) appear to provide that existing uses and the water quality necessary to protect those uses must be maintained and protected. If accurate, this is consistent with the "Tier 1" existing use provisions of 40 CFR131.12(a)(1) which provide that "[e]xisting in-stream water uses and the level of water necessary to protect the existing uses shall be maintained and protected."

With regard to Tier II high-quality waters, agency guidance makes it clear that variances are limited-term exemption from otherwise applicable water quality standards intended to support incremental movement toward attainment of those standards. Any variance that would authorize degradation of high quality water below a currently attained designated use is inconsistent with antidegradation and the requirements of the CWA.

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[23] *Id.*

[24] *Pinto Creek* at 1012-13.

[25] *Id.* at 1014.

[26] Great Lakes Initiative [hereinafter "GLI"] Pt. 132, App F, Procedure 2 §A.1.

[27] 40 C.F.R. § 131.12(a)(1).

[28] 40 C.F.R. § 131.3(e).

Likewise, under §75-5-303, DEQ may not authorize any degradation of Tier II high-quality waters - through an MPDES permit or otherwise - unless the prospective polluter has affirmatively demonstrated by a preponderance of the evidence that: (a) degradation *is* necessary because there are no economically, environmentally, and technologically feasible modifications to the proposed -project that would result in no degradation; (b) the proposed project will result in important economic or social development and that the benefit of the development exceeds the costs to society of allowing degradation of high-quality waters; (c) existing and anticipated uses of states waters will be fully protected; and (d) the least degrading water quality protection practices determined by the department to be economically, environmentally and technologically feasible will be fully implemented by the applicant prior to and during the proposed activity.[29] This process imposes a high burden on a polluter and DEQ: to override the nondegradation requirement they must demonstrate that there "are no feasible modifications that would result in no degradation" and that the "least degrading water quality practices" are implemented.

i. Variances Must Include a Requirement to Maintain and Protect Existing Uses and the Water Quality Necessary to Support Them

The proposed variance rule found in DEQ Circular 12-B does not discuss protection of existing uses. This omission falls short of what is necessary to meet EPA's implementing regulations because: (1) there is no requirement for variances to meet the antidegradation policy, and therefore it falls short of the protection of existing uses that is required, (2) even if DEQ does have a requirement that our review missed, there is no implementation methods for Tier I of the antidegradation policy which it could use to ensure that any such provision is followed and to demonstrate precisely what provision is provided, and (3) the Department is unlikely to enforce any existing use protection requirement without explicit language here to do so because it has failed to acknowledge that existing use protection is a required aspect of water quality standards in its TMDLs, its NPDES permits, its 303(d) lists of impaired waters, and its 401 certifications.

EPA has stated repeatedly that variances are subject to the "same substantive and procedural requirements as removing a designated use."[30] The requirement to protect existing uses in the issuance of variances derives from several sources. First, existing use protection is the "floor" of water quality, below which State standards may not go.[31] Because variances are changes to water quality standards they too may not go below that floor. This is encoded in the requirement to classify existing uses[32] as well as the antidegradation provisions to protect those uses,[33] which must be read together.[34] Existing use protection is specifically noted – *twice* – in EPA regulations concerning the removal of designated uses, the same provision that is used for variances.[35] EPA notes that the protection of existing uses is a site-specific exercise, which is

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[29] Mont. Code Ann. 75-5-303(3).

[30] Handbook at 5.3; EPA Interim Economic Guidance Workbook, EPA-823-B- 95-002; March 1995 [hereinafter "Economic Guidance"] at 1-3.

[31] Handbook; EPA Questions & Answers on Antidegradation, August 1985 [hereinafter "Questions and Answers"]; 48 Fed. Reg. 51402 (November 8, 1983)

[32] 40 C.F.R. § 131.10.

[33] 40 C.F.R. § 131.12.

[34] Water Quality Standards Regulation Proposed Rule, Advance Notice of Proposed Rulemaking, 63 Fed. Reg. 36741, July 7, 1998 [hereinafter "ANPRM"] at 36752.

[35] 40 C.F.R. §§ 131.10(g) & (h)(1).

wholly consistent with the issuance of variances.[36]

EPA considers protection of existing uses as essential in issuing variances.[37] EPA notes that it is the necessity of preserving existing uses, as well as making reasonable progress towards ultimate attainment, that requires the conditions of a variance to be set as close as possible to the designated uses and "always retained at the level needed to preserve the existing use."[38] These conditions include various prohibitions, control requirements, monitoring, and evaluation.[39] The requirement to protect existing uses pursuant to the antidegradation policy applies during triennial reviews and water quality standards revisions, of which a variance is one, as well as the issuance of NPDES permits.

The six factors of 40 C.F.R. § 131.10(g) cannot be read outside the context of the text of 40 C.F.R. § 131.10(g), of § 131.10(h), and of the antidegradation policy, all of which specify the protection of existing uses. Similarly, the GLI rules explicitly require that in addition to the six factors governing use attainability, the variance seeker show the antidegradation requirements have been met.[40] Consistent with these policies, EPA has also held that permits issued pursuant to variances must still comply with antidegradation requirements, including existing use protection.[41] A variance applies to the applicable criterion and does not modify the application of the existing use and designated use provisions of the water quality standard.[42]

In addition, the antidegradation policy, of which the Tier I protections for existing uses and level of water quality necessary to protect them is one, require a state to "identify the methods for implementing such policy."[43] In contrast to EPA's regulations, guidance, and policies, DEQ's proposed rule in Circular 12-B only references assessment of downstream, which we assume to be an implicit recognition of the applicability of the state's nondegradation policy, but does not set out how this end will be achieved. EPA's regulations require much more than this. The existing use protection in EPA regulations does more than prohibit elimination or impairment of existing uses. It states that "[e]xisting instream water uses and the level of water quality necessary to protect the existing uses shall be maintained and protected."[44] In other words, on the continuum between eliminating existing uses and full support of existing uses, the language "shall be maintained and protected" requires full support. There is no legal or policy reason to countenance anything less than full support of those uses that constitute the floor of water quality in this nation. Merely not entirely eliminating or impairing existing uses is inadequate protection.

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36 ANPRM at 36752.

37 40 C.F.R. § 131.10(h)(1); ANPRM at 36759, 36760.

38 EPA, Guidance: Coordinating CSO Long Term Planning with Water Quality Standards Reviews, EPA-833-R-01-002, July 31, 2001 [hereinafter "CSO Guidance"] at 34.

39 *Id*. at 35.

40 GLI Pt. 132, App F, Procedure 2 §C.2.a; GLI Supplementary Information Document, EPA-820-B-95-001, March 1995 [hereinafter "GLI SID"] Sec. VIII.B.3.c.

41 EPA, Guidance for State Implementation of Water Quality Standards for CWA Section 303(C)(2)(B), December 1988 [hereinafter "Guidance for Implementation"] at 6.

42 EPA Memorandum, from Kenneth Mackenthun to Regional WQS Coordinators, Re: Definition of WQS Terms, July 3 1979 at 1.

43 40 C.F.R. § 131.12(a).

44 40 C.F.R. § 131.12(a)(1).

ii. The Proposed Variance Rule is Incorrectly Limited in its Requirement for All Cost-Effective and Reasonable Nonpoint Source Controls

EPA has stated repeatedly that variances are subject to the "same substantive and procedural requirements as removing a designated use."[45] This use provision applies to issuance of a variance as a temporary removal of designated uses governed by the same EPA regulations.[46]

In the GLI rules, this requirement was changed to mean that BMPs must be implemented (1) *by the discharger (2) before* a variance is granted, two requirements that are specific to the GLI, one of which is arguably less stringent (the scope) and one of which is arguably more stringent (the timing). In contrast, the national regulations that apply to Montana are consistent with, and identical to, the Tier II antidegradation protection language which requires all "cost-effective and reasonable nonpoint source controls" for nonpoint sources.[47] Because the use removal provisions apply to water bodies and variances apply only to the specific discharger seeking the temporary suspension of one or more standards, DEQ cannot suspend requirements of the water quality standards on other sources – point or nonpoint – as an outcome of the variance. Therefore, the BMP requirements of 40 C.F.R. §131.10(h)(2) apply to all nonpoint sources in the consideration of a variance application. EPA has supported this position by noting that in issuing variances, the economic impacts that can be considered are only those that result from treatment beyond that required by technology-based regulations. This includes both technology-based limits on point source discharges as well as BMPs to nonpoint sources.[48]

In addition, while the GLI's more limited BMP requirements for permittees seeking variances must be met *prior* to issuance of the variance, the clear language of the non-GLI language that applies to Montana discusses the State's finding that designated uses "*will be* attained . . . by implementing [nonpoint source controls],"[49] a finding related to *future* attainability. To the extent Montana has enforceable controls on nonpoint sources, they must be implemented as part of the Tier II protections. Likewise, to the extent that Montana has enforceable controls on nonpoint sources, they must be implemented when the Department or a source seeks to remove designated uses through the provisions of 40 C.F.R. § 131.10, including a temporary removal in a variance.

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[45] Handbook at 5.3; EPA Interim Economic Guidance Workbook, EPA-823-B- 95-002; March 1995 [hereinafter "Economic Guidance"] at 1-3; CSO Guidance at 34.

[46] ANPRM at 36760.

[47] EPA Memorandum from Tudor Davies, EPA, to Water Management Division Directors, February 22, 1994, Re: Interpretation of Federal Antidegradation Regulatory Requirement [hereinafter "Interpretation"] at 2.

[48] Economic Guidance at 1-1. ("This workbook provides guidance for those seeking to . . . obtain a variance based on economic considerations, or to lower water quality in a high-quality water. In addition, it provides guidance to States and EPA regions responsible for reviewing requests for variances and modifications to designated uses, and for approval of antidegradation analyses.

…

The economic impacts considered are those that result from treatment beyond that required by technology-based regulations. Since water quality cannot be lower than that resulting from technology-based limits applied to direct and indirect point source discharges and reasonable Best Management Practices (BMP) applied to nonpoint sources, these are considered to be the baseline.")

[49] 40 C.F.R. §131.10(h)(2) (emphasis added).

The Rule Package's language is at best ambiguous as to the timing of such controls and could be read to be concurrent or in the future. The variance rule proposed appears to be less protective than either the GLI or the nationally-applicable regulations by narrowing the scope of nonpoint sources to be controlled and by allowing those controls to happen concurrently or in the future. Given the absence of any discussion of non-point source controls, let alone discussion as to what types of management practices are cost-effective or reasonable for nonpoint sources, one can only come to the reluctant conclusion that DEQ intends to ignore this provision.

As demonstrated above, EPA regulations link the fate of point and nonpoint sources together. When DEQ proposes to separate their fate, it can be sure that the result will be a continuation of the existing ineffective and nonexistent nonpoint source practices and the dirty water those sources create. The perpetuation of the same approach used by DEQ in its TMDLs – the unfounded belief that nonpoint sources are or will reduce loads – is now proposed to be incorporated into variance rules, from which no good will come.

4. Variances Must Include Substantive Requirements for Reasonable Progress Towards Attainment and Variance Renewal Must be Based On Substantial Information

EPA believes that variances can be used to implement water protection actions, assess their results, and study the water quality problem to better understand it.[50] We believe DEQ understands the only difference between a source with a compliance schedule and a source with a variance should be that the latter is not able to commit to a date certain by which it can meet waste load allocations. We support this general policy.

In order that this policy may be carried out, however, conditions for pollution control and monitoring must be included in a variance and incorporated into the applicable NPDES permit. This gives meaning to the stated notion that variances are "short-term" exemptions from meeting standards. Likewise, this approach ensures that renewal is not automatic but, rather, requires a new affirmative showing by the applicant.[51] We agree that the required triennial review is a time when the public should be able to evaluate whether the conditions of the variance have been met and the conditions the variance was based upon still apply.[52]

We are troubled that DEQ's rule fails to contain an explicit requirement that permittees seeking variances must submit a type of pollutant reduction plan that includes any actions to be taken by the permitee that would result in reasonable progress toward meeting the underlying water quality standard. Nor does DEQ's proposed variance rule require a clause that establishes and incorporates into the discharger's NPDES permit all condition necessary to implement an approved variance and associated pollution reduction plan. The proposed variance should should be amended to include these items.

The key importance of such a "reasonable progress" requirement is ensuring that variances are, indeed, temporary. Required studies and monitoring should not be limited to ensuring compliance with the variance conditions but also so that DEQ, and the public, can determine in the likely event of an application for renewal whether the water quality is improving or

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[50] Handbook at 5.3; ANPRM at 36758-60.

[51] ANPRM at 36759; *see also* GLI.

[52] ANPRM at 36759.

deteriorating and whether any reasonable progress has been achieved. The reality is that DEQ permit writers will be under significant pressure to agree to as little as permittees want to do. This is particularly true in the case of asking permittees to do in-stream monitoring. The only way to strengthen the position of those permit writers is to make the requirements for measuring any reasonable progress or lack thereof more clear and certainly mandatory.

In addition, with regard to municipal sources, it is clear that there are some significant ways in which source control can be achieved – through controls on discharges to municipal sewage collection systems from un- and under-regulated industries beyond federal pretreatment requirements, unregulated commercial sources, and from runoff that could be controlled by municipal ordinances. Without clear direction from DEQ concerning the degree to which these unpopular restrictions would need to be taken by municipal NPDES permittees, they will likely seek to avoid them as much as possible for political and budgetary reasons.

Respectfully submitted,

Guy Alsentzer

The Upper Missouri Waterkeeper and Executive Director
Upper Missouri Waterkeeper, Inc.
PO Box 128
Bozeman, MT 59771
guy@uppermissouriwaterkeeper.org

From: James Thomas
To: DEQ WQP Admin
Subject: comments re: mt nutrient criteria
Date: Tuesday, April 01, 2014 2:37:29 PM

From: Anne Kania <floatingisland@me.com>;
To: James Thomas <jamesmanddebrathomas@yahoo.com>;
Cc: Bruce Kania <bruce@floatingislandinternational.com>;
Subject: DEQ comment
Sent: Tue, Apr 1, 2014 8:13:40 PM

RESPONSE LETTER to request for comments,
CIRCULAR 12B "Variances" for Nutrient Criteria
DATE: April 1, 2014

To: Montana Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901
Attention : Ms. Carrie Greeley

From: Thomas's Tight Squeeze Farm (Providing Sustainable Products and Environmental Consulting Services)
1270 Pine Cone Road
White Swan,WA 98952

James M.Thomas, Sole Proprietor

RESPONSE TO REQUEST FOR COMMENTS TO THE PROPOSED WATER QUALITY STANDARDS, NUTRIENT CRITERIA FOR MPDES PERMITTEES

I am an Environmental Scientist trained by the US Environmental Protection Agency in developing and implementing water quality standards. I used that training for the development and implementation of water quality standards for the Yakama Reservation on behalf of the Confederated Tribes and Bands of the Yakama Nation within the boundaries of Washington State. These water quality standards were approved and adopted by the Yakama Tribal Council, including nutrient criteria. Therefore, I am well aware of the implications associated with Montana Department of Environmental Quality (MDEQ) initiating development and implementation of nutrient criteria as a component of water quality standards for the State of Montana.

Specifically, I recognize the potential difficulties for MPDES dischargers in complying with the additional permit requirements compared to existing permit requirements and the difficulties for MDEQ in administering those permit requirements. Nation-wide addressing the environmental degradation occurring from eutrophication attributable to excessive nutrients is proving to be a challenge to permitted dischargers and discharge regulators alike. Therefore, I commend MDEQ for taking these courageous first steps of submitting these documents for comment toward development and implementation of these nutrient criteria incorporated into Montana Water Quality Standards. It is my hope that the leadership being demonstrated by Montana DEQ in managing nutrients in waters of Montana will set an example for other states to follow, thereby improving water quality not only in Montana waters but nation-wide.

I'm directing my comments toward Supplemental Document Circular12B, Section 1.0 wherein "variances" for permittees are discussed. I recognize the perceived difficulties for dischargers with limited resources of staff and funding in meeting yet another MPDES Permit requirement and the almost knee-jerk response of small municipality permittees to apply for variances because of the perceived additional expense and the, until recently, limited technology available to small municipal dischargers.

In my work as an Environmental Scientist/Water Quality Specialist I have been reviewing available technologies for helping small municipalities meet nutrient criteria for their waste water discharge permits since 2010. My review led me to the Montana based companies of Floating Islands International and Biohaven Inc., which are both based in Shepherd, Montana. As indicated by scientifically sound studies and numerous existing installations of the technologies, which these companies provide, nutrient management in waste water to the point of meeting nutrient criteria is now generally speaking, affordable and greatly simplified. I envision the technology provided by these two Montana companies to potentially be of great benefit to MPDES permittees. I'll use a small town waste water treatment plant near my home of South Central Washington State as an example of what I envision. This small town is similar to many rural Montana small municipalities with a resident base of less than 700 people,therefore a limited tax base to fund waste water management and a limited land base to treat waste water, with no room for waste water facility expansion. Nevertheless, because this town is within the confines of an Indian Reservation the NPDES regulator, the US EPA is requiring the municipality to meet a new NPDES permit requirement for Ammonia, which is a component of the Total Kjehdahl Nitrogen test discussed in the text of your proposed criteria. Because I am desirous of helping this small community meet the NPDES permit obligations without suffering "economic Impacts" such as referred to in the document (12B , 1.0) I will be presenting to the Town Council the opportunities for installing and operating in their existing waste water lagoons the technologies offered by the aforementioned Montana based companies to accomplish NPDES permit nutrient requirements. I am convinced this can be accomplished and if the technology is installed and managed properly a request by the Town Council for the US EPA version of a "variance" will not be necessary.

In my professional opinion, the more I've investigated the existing installations of this technology, the more I'm becoming convinced this technology is helping communities and individuals around the globe address eutrophication problems with the waters they are stewarding. Because water quality stewardship is my passion and profession I'll be doing all I can to proliferate the application of this technology in my home state of Washington.

Similarly the fact that these technologies are the products of Montana based companies which are in collaborative relationships with University based researchers and private professionals skilled in nutrient flow modeling and other aspects of water quality management indicates to me that at the least, these technologies should be included on a list of available and affordable technologies for MPDES permittees. Should this actually occur and the technology be widely applied I envision that MPDES permittees using these technologies will be enabled to become actively involved in: 1) nutrient trading as discussed in the nutrient criteria document ; 2) modeling of nutrient inputs and outputs with the assistance of private professionals previously mentioned and as discussed in the document circular 12B ; 3) be enabled to meet or exceed nutrient criteria even during cold temperature periods as indicated in the attached graphs (Figures 1 and 2).

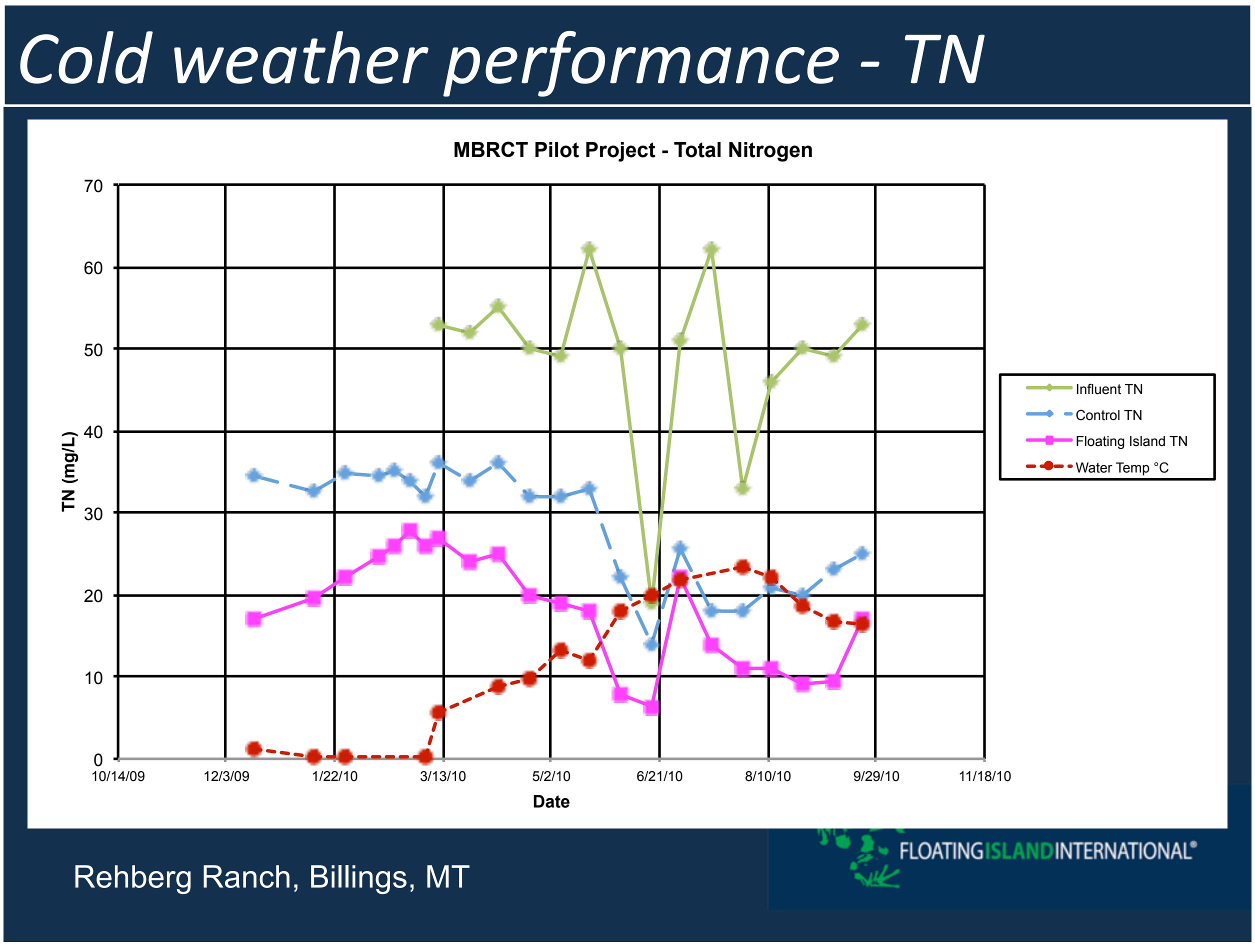

Figure 1

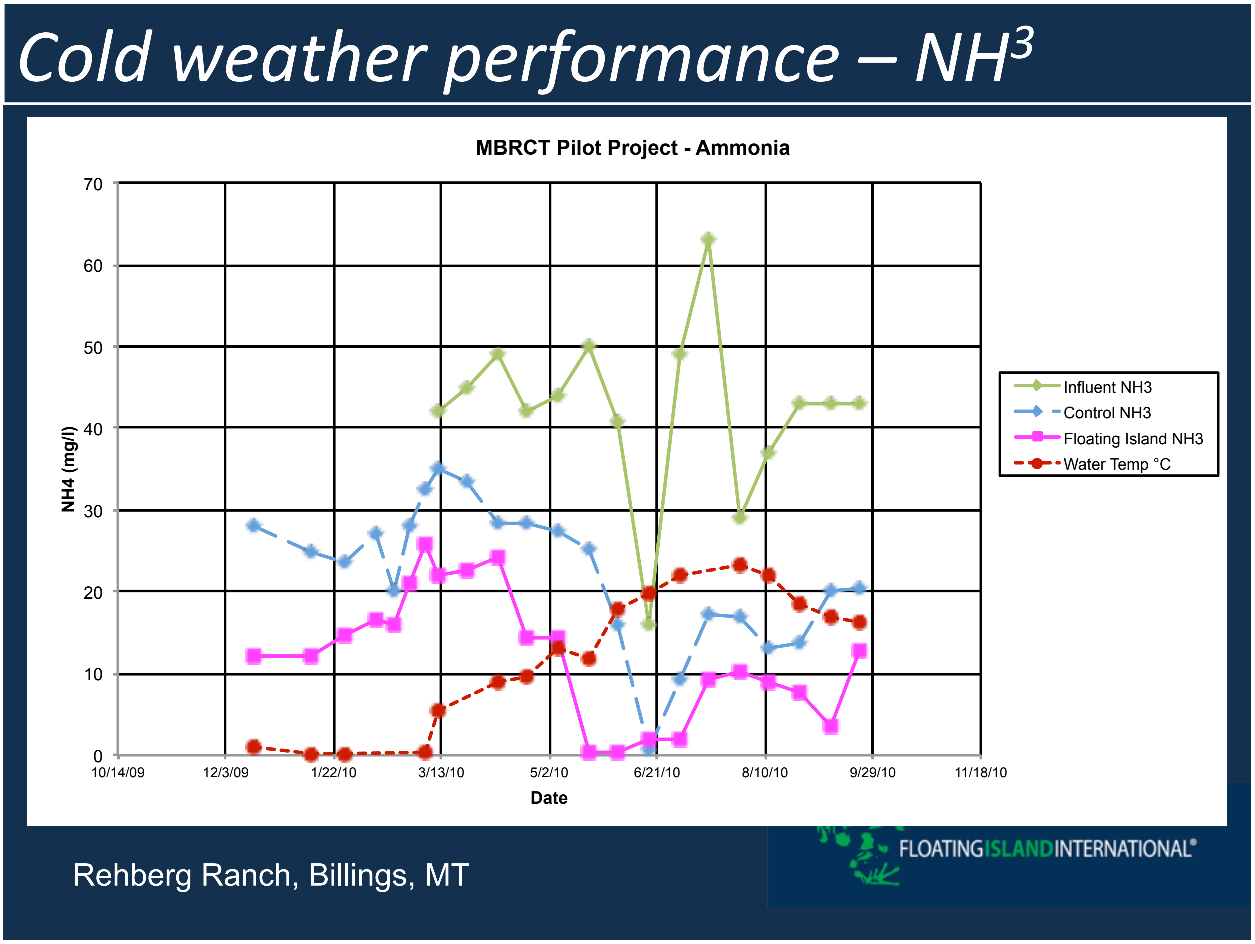

Figure 2

I'm convinced that the use of this emerging affordable technology by many MPDES permittees will make the knee-jerk response of dischargers to apply for a variance to be a thing of the past. Most importantly, the waters of Montana will be protected from excessive nutrients from point source dischargers while simultaneously dischargers using the technology will be spared from : "Economic Impacts" referred to in the document.

In closing, thank you for reading the comments of a Washington State based Environmental Scientist/Water Quality Specialist. Water quality is of global concern inasmuch as water does not recognize state boundaries. For example what occurs to water in Montana will eventually affect downstream water users, even across state boundaries. Once again thank you for your demonstrated leadership in this matter and please consider my request to include the technologies and services provided by the Montana based companies and their collaborators on a list of available technologies to help meet the proposed nutrient criteria for MPDES permittees.

Wittenberg, Joyce

From: Blend, Jeff
Sent: Wednesday, April 02, 2014 10:26 AM
To: Greeley, Carrie
Subject: FW: Nutrient analysis

Carrie:

Per John North's instructions below, I sent you an electronic copy of the Small Business Impact Analysis for the Nutrient Criteria and cover letter. I also put paper copies on your desk and initialed the cover letter. Let me know if that was not correct, or if I need to do something else. Thanks.

Jeff

From: North, John
Sent: Tuesday, April 01, 2014 4:52 PM
To: Blend, Jeff
Subject: RE: Nutrient analysis

I think it's enough. Go ahead and initial the memo and get the memo and analysis to Carrie Greeley before 5:00.

From: Blend, Jeff
Sent: Tuesday, April 01, 2014 1:52 PM
To: North, John
Subject: Nutrient analysis

Not sure if this is enough. I kept it to two sentences. Also attached is the analysis with your one final change.

Jeff Blend
(406) 444-0218
jblend@mt.gov

Economist and Energy Analyst
Energy and Pollution Prevention Bureau
Montana Dept. of Environmental Quality
1520 East Sixth Ave
P.O. Box 200901
Helena, MT 59620-0901

Montana Department of
ENVIRONMENTAL QUALITY **MEMO**

TO: Board of Environmental Review

FROM: Jeff Blend, Economist

DATE: April 1, 2014

SUBJECT: Small Business Impact Analysis for MAR Notice No. 356

Attached is a Small Business Impact analysis for MAR Notice No. 356. This analysis fulfills the statutory obligations of Senate Bill 139 passed in the 2013 Legislature, which requires an analysis of the direct impacts to small businesses from a new rule.

Small Business Impact Analysis of the Nutrient Standards Rule

The Nutrient Standards Rule (MAR Notice No. 17-356, Circular DEQ 12A) and Variance Rule (MAR Notice No. 17-355, Circular 12B) will significantly and directly impact a couple of small businesses in the state of Montana. For the purposes of this study, small businesses are defined as less than 50 employees. An estimated two existing small businesses will see larger cost impacts from having to meet higher water quality standards. These two businesses are located in a small town and rural area in Montana. The two businesses that might have to upgrade would likely experience significant impacts from the rule.

Impacts

Direct Costs to Two Existing Small Businesses to Meet WERF Level 2 and Level 3[1]

Out of the thousands of businesses in Montana, about 50 were identified as ones that would be affected directly by the nutrient criteria. This identification was done in the document entitled, " Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met by Entities in the Private Sector in 2011/2012". Included were businesses that have a discharge permit into state waters, and are not otherwise hooked up to a municipal system. Therefore, the numeric nutrient water quality standards only apply to those few business entities that have a surface water discharge permit and treat their own water. Out of these 50 businesses, only about 4 or 5 would qualify as small businesses (less than 50 employees) that would be directly affected by this rule. Indeed, most of the 50 businesses are mines, refineries, natural gas companies and other large entities over 50 employees. Only two very small businesses out of the four were found to have treatment levels that do not currently meet WERF Level 2 and thus the general variance level allowed at this time under the Variance rule. Thus, only two existing small businesses in Montana would likely have to upgrade their self-owned wastewater treatment systems to meet WERF 2 (or find alternative forms of water disposal). The other two or three small businesses were estimated to be already meeting WERF level 3 and 4, or meeting higher standards than the general variance.

DEQ examined the wastewater permits and the Statement of Basis for these 2 affected businesses. These records are located within DEQ's Permitting Division. Current effluent nutrient levels and estimates of current treatment costs at the two businesses were compared to costs that would be needed to meet WERF levels 2 and 3 based on the WERF study. In this way, annual capital and operations costs needed for meeting base nutrient criteria (above current nutrient treatment costs) were applied to each business.

[1] Much of this section is taken from From 'Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met by Entities in the Private Sector in 2011/2012'

Costs are under-estimated for small facilities and those with low flows, because the WERF cost data was multiplied by effluent flow providing a linear cost estimate based on flow. Clearly, there will be a minimum capital cost of treating to WERF level 2 for facilities with small flows such as pouring concrete, hiring labor, etc. that is greater than the linear cost estimates for these low-flow and small facilities. Thus, we multiply the WERF cost numbers by 5 to account for a lack of economies of scale in these two businesses.

An analysis of the life-cycle costs for a number of technologies used to control nitrogen and phosphorus in wastewater treatment plants estimated that labor costs are between 15-21 percent of the annualized capital costs for nitrogen and 15-48 percent of annualized capital costs for phosphorus to treat nutrients (Kang and Omstead, 2011).[2] Thus, we add the addition of the high estimate of labor costs (48 percent additional costs annually) as a percentage of capital costs were considered across each scenario.

The two businesses were assumed to be at WERF Level 1 which suggests little or no nutrient treatment. The businesses were calculated to have annual costs of $6,700 and $1,500 annually to meet WERF level 2. These figures were calculated from numbers found in the DRAFT Interim WERF study *"Striking the Balance Between Wastewater Treatment Nutrient Removal and Sustainability, Considering Capital and Operating Costs, Energy, Air and Water Quality and More"* (Falk, et al., 2011a). Because these businesses have such small outflows, these business might choose to either find an alternative discharge method (that is not into a state water), apply for an individual variance or perhaps change some operating procedures rather than investing in the capital expenditures to reach WERF level 2.

If the two businesses did invest in the equipment to get to WERF level 2, the $6,700 and $1,500 numbers for annual expenditure are probably significantly underestimated because the WERF numbers were for larger businesses with economies of scale. If we take into account economies of scale, it might be reasonable to multiply these numbers by a factor of 5. If we add 48 percent for labor costs on to this multiple of 5, then the numbers would be about $49,500 and $11,000 annually for these two businesses to reach WERF level 2. It is not clear how far beyond WERF level 2 these businesses would have to go as time goes on. These annual costs could effectively harm or shut down these two businesses. Thus, it is likely that both businesses would apply for an individual variance or find an alternative discharge method to avoid substantial impacts and these annual costs.

For the two businesses (one located in Madison County and the other in the northern Powder River Basin), non-discharge options include, a. land application, b. total/seasonal retention, c. piping water long distances away from state waters or to larger state waters with dilution, and d. trading. These non-discharge options, including land application, could be expensive or might not be feasible in certain areas (such as places far from open land or with few trade partners) or during the cold months.

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[2] Based on information in: Introduction of Nutrient Removal technologies Manual, EPA, 2008 and WEF/WERF Cooperative Study of Nutrient Removal Plants: Achievable Technology Performance Statistics for Low Effluent Limits)

As noted above, these two affected businesses are located in or near small towns. Since most small towns do not have diverse economies, even a small decrease in business and in population can have a large effect on them. For example, some small Montana towns have less than 10 businesses total (e.g. Fromberg, MT).

There would be paperwork and other reporting tasks and sampling costs that the two small businesses directly affected would have to undertake in meeting the nutrient standards rule. It is difficult to say what these costs would be, but water sampling could be the largest cost along with extra time needed to comply. The other two or three business that already meet general variance standards would also have additional paperwork to comply with in their water discharge permit, although probably not significantly more than the current paperwork.

New small businesses with new wastewater discharges that want to located in Montana could also be affected greatly by the Nutrient Standards rule and may need to install water treatment technologies up to the Limits of Technology (stricter than the general variance) due to Non-Degradation rules. Some new businesses may choose to not locate in Montana if they were required to be in compliance immediately, while other states may not have this requirement. Eventually, all U.S. states will have to meet nutrient criteria, so this effect will probably decline over time.

The multiplier effects from the two small businesses directly affected would be minimal for the state as a whole and could be significant to the small towns where they are located.

Appendix A: Company Sizes

Matt Betcher of the Montana Department of Labor and Industry, Research and Analysis Bureau, entered the size classes of the businesses from the approximately 50 affected businesses (that have discharge permits) studied in the 'Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met in 2011/2012'. The obvious large businesses greater than 50 employees were not looked at, but businesses that might be under 50 employees were looked at by Mr. Betcher. He was able to find ranges of numbers of employees for these businesses using an alpha search of business names. The size classes are based on the average employment over last year of Quarterly Census of Employment and Wages (QCEW) data. Jeff Blend looked up the companies that were not in that database.

Quarterly Census of Employment and Wages is found at: http://www.bls.gov/cew/

Appendix B-Small Business Statute

http://openstates.org/mt/bills/2013/SB139/documents/MTD00006286/

2013 Montana Legislature, SENATE BILL NO. 139, INTRODUCED BY E. WALKER

NEW SECTION. **Section 1. Small business impact analysis -- assistance.** (1) Prior to the adoption of a proposed rule, the agency that has proposed the rule shall determine if the rule will adversely or positively impact small businesses. If the agency determines that the proposed rule will impact small businesses, the determination must be published in the register when the proposed rule is published. If the agency determines that the proposed rule may have an adverse or positive impact on small businesses, the agency shall prepare a small business impact analysis that, at a minimum, must:

(a) identify by class or group the small businesses probably affected by the proposed rule;

(b) include a statement of the probable adverse or positive effects of the proposed rule on the small businesses identified in subsection (1)(a); and

(c) include a description of any alternative methods that may be reasonably implemented to minimize or eliminate adverse effects described in subsection (1)(b), while still achieving the purpose of the proposed rule.

(2) The agency shall provide documentation for the estimates, statements, and descriptions required under subsection (1).

(3) The office of economic development, established in 2-15-218, shall advise and assist agencies in complying with this section.

Section 2. Section 2-4-102, MCA, is amended to read:

"**2-4-102. Definitions.** For purposes of this chapter, the following definitions apply:

(1) "Administrative rule review committee" or "committee" means the appropriate committee assigned subject matter jurisdiction in Title 5, chapter 5, part 2.

(2) (a) "Agency" means an agency, as defined in 2-3-102, of state government, except that the provisions of this chapter do not apply to the following:

(i) the state board of pardons and parole, except that the board is subject to the requirements of 2-4-103, 2-4-201, 2-4-202, and 2-4-306 and its rules must be published in the ARM and the register;

(ii) the supervision and administration of a penal institution with regard to the institutional supervision, custody, control, care, or treatment of youth or prisoners;

(iii) the board of regents and the Montana university system;

(iv) the financing, construction, and maintenance of public works;

(v) the public service commission when conducting arbitration proceedings pursuant to 47 U.S.C. 252 and 69-3-837.

(b) The term does not include a school district, a unit of local government, or any other political subdivision of the state.

(3) "ARM" means the Administrative Rules of Montana.

(4) "Contested case" means a proceeding before an agency in which a determination of legal rights, duties, or privileges of a party is required by law to be made after an opportunity for hearing. The term includes but is not restricted to ratemaking, price fixing, and licensing.

(5) (a) "Interested person" means a person who has expressed to the agency an interest concerning agency actions under this chapter and has requested to be placed on the agency's list of interested persons as to matters of which the person desires to be given notice.

(b) The term does not extend to contested cases.

(6) "License" includes the whole or part of an agency permit, certificate, approval, registration, charter, or other form of permission required by law but does not include a license required solely for revenue purposes.

(7) "Licensing" includes an agency process respecting the grant, denial, renewal, revocation, suspension, annulment, withdrawal, limitation, transfer, or amendment of a license.

(8) "Party" means a person named or admitted as a party or properly seeking and entitled as of right to be admitted as a party, but this chapter may not be construed to prevent an agency from admitting any person as a party for limited purposes.

(9) "Person" means an individual, partnership, corporation, association, governmental subdivision, agency, or public organization of any character.

(10) "Register" means the Montana Administrative Register.

(11) (a) "Rule" means each agency regulation, standard, or statement of general applicability that implements, interprets, or prescribes law or policy or describes the organization, procedures, or practice requirements of an agency. The term includes the amendment or repeal of a prior rule.

(b) The term does not include:

(i) statements concerning only the internal management of an agency or state government and not affecting private rights or procedures available to the public, including rules implementing the state personnel classification plan, the state wage and salary plan, or the statewide accounting, budgeting, and human resource system;

(ii) formal opinions of the attorney general and declaratory rulings issued pursuant to 2-4-501;

(iii) rules relating to the use of public works, facilities, streets, and highways when the substance of the rules is indicated to the public by means of signs or signals;

(iv) seasonal rules adopted annually or biennially relating to hunting, fishing, and trapping when there is a statutory requirement for the publication of the rules and rules adopted annually or biennially relating to the seasonal recreational use of lands and waters owned or controlled by the state when the substance of the rules is indicated to the public by means of signs or signals; or

(v) uniform rules adopted pursuant to interstate compact, except that the rules must be filed in accordance with 2-4-306 and must be published in the ARM.

(12) (a) "Significant interest to the public" means agency actions under this chapter regarding matters that the agency knows to be of widespread citizen interest. These matters include issues involving a substantial fiscal impact to or controversy involving a particular class or group of individuals.

(b) The term does not extend to contested cases.

<u>(13) "Small business" means a business entity, including its affiliates, that is independently owned and operated and that employs fewer than 50 full-time employees.</u>

~~(13)~~<u>(14)</u> "Substantive rules" are either:

(a) legislative rules, which if adopted in accordance with this chapter and under expressly delegated authority to promulgate rules to implement a statute have the force of law and when not so adopted are invalid; or

(b) adjective or interpretive rules, which may be adopted in accordance with this chapter and under express or implied authority to codify an interpretation of a statute. The interpretation lacks the force of law."

From: Suplee, Mike
To: Greeley, Carrie
Subject: FW: Public comment on the Board of Environmental Review (Board) is proposed numeric standards for nitrogen and phosphorus concentrations in surface waters.
Date: Friday, April 04, 2014 3:39:04 PM

Hi Carrie;

Below is Jon Cuthbertson's comment

From: Johnson, Elois
Sent: Friday, March 28, 2014 10:17 AM
To: Orr, Katherine; Suplee, Mike
Subject: FW: Public comment on the Board of Environmental Review (Board) is proposed numeric standards for nitrogen and phosphorus concentrations in surface waters.

Elois M. Johnson
Paralegal
Department of Environmental Quality
PO Box 200901
Helena, MT 59620-0901
Telephone: 406.444.2630
Fax: 406.444.4386
Email: ejohnson@mt.gov

From: M.E. Lab [mailto:info@melab.us]
Sent: Friday, March 28, 2014 10:10 AM
To: Johnson, Elois; Greeley, Carrie; wpippin@energylab.com
Subject: Public comment on the Board of Environmental Review (Board) is proposed numeric standards for nitrogen and phosphorus concentrations in surface waters.

Elois Johnson,

In Circular DEQ 12 on page 5 you list an RRV for total kjeldahl nitrogen of 0.15 mg/L or 150 ug/L.

This is unobtainable using that method.
A normal RRV for total kjeldahl nitrogen is 0.5 mg/L.
A really good lab, under perfect conditions MAY be able to achieve 0.2 mg/L.
If you doubt me, call the various labs in Montana and ask them.
Even your own State DPPHS lab cannot achieve that RRV.
There is too much cross contamination between the tubes.

Passing a rule that requires using a 40 year old method, and then requiring an unrealistic RRV is wrong.

You are just building in a problem.
Don’t do it.

Sincerely,

Jon Cuthbertson
Laboratory Director
Montana Environmental Laboratory LLC.

From: Johnson, Elois
To: Suplee, Mike; Greeley, Carrie
Subject: FW: Comments on Nutrient Standards and Variances DEQ-12A and DEQ-12B
Date: Friday, April 04, 2014 8:29:50 AM

Mike:

As you can see below, Mr. Harvala emailed this comment to deqwpdadmin@mt.gov and me. I didn't forward this comment, assuming you or Carrie would have received it as it was addressed to deqwpdadmin. There were, if I remember correctly, a couple of other comments that were addressed to this same email address, and I didn't forward them either, assuming you already received them. So you received all the other comments that are on the list I sent you? Thanks.

Elois

From: Jon Harvala [mailto:jharvala@co.missoula.mt.us]
Sent: Tuesday, April 01, 2014 5:01 PM
To: deqwpdadmin@mt.gov; Johnson, Elois
Subject: Comments on Nutrient Standards and Variances DEQ-12A and DEQ-12B

Elois;
The Missoula Water Quality District submits the proposed comments.
Jon

Jon Harvala
Missoula Valley Water Quality District
406-258-3109
jharvala@co.missoula.mt.us

Missoula Public Health
City-County Health Department

Missoula City-County Health Department
WATER QUALITY DISTRICT
301 West Alder Street | Missoula MT 59802-4123
www.co.missoula.mt.us/wq
Phone | 406.258.4890
Fax | 406.258.4781

April 1, 2014

Elois Johnson, Paralegal
Department of Environmental Quality
P.O. BOX 200901
Helena, MT 59620-0901

Re: Comments on Proposed Numeric Nutrient Standards DEQ-12A and Variances DEQ-12B

To Whom it May Concern:

The Missoula Valley Water Quality District (MVWQD) submits the following comments on the proposed Montana Base Numeric Nutrient Standards (Montana Department of Environmental Quality (DEQ) Circular DEQ-12A) and associated Rule amendments. The comments are based on over 30 years of effort that led to the first nutrient TMDL in Montana and current numeric standards in effect on the Clark Fork River. The comments include a brief summary of Missoula's nutrient reduction efforts, support for the proposed nutrient standards and comments on proposed Rule 1 (Department Circular DEQ-12B) that implements 75-5-313, MCA.

Clark Fork River Impairment – VNRP, TMDL & Clark Fork River Standards

Nutrient pollution of the Clark Fork drainage and resultant nuisance algae growth has focused concern about water quality in the Clark Fork River for many years. In the 1980's a proposal to allow year round discharge from the Frenchtown kraft paper mill eventually resulted in a comprehensive water quality monitoring plan for the Clark Fork Basin. The 1989 Nutrient Pollution Source Assessment, completed by the Montana Department of Health and Environmental Sciences (predecessor to Montana Department of Environmental Quality (DEQ) identified major sources of nutrients and has provided the essential baseline water quality data that subsequent monitoring and nutrient reduction efforts used to measure progress.

In 1988 the City of Missoula and Missoula County jointly enacted the Missoula Phosphate Ordinance, which prohibited the sale of laundry detergent containing more than 0.5% phosphorus, became effective in 1989. This resulted in an almost immediate 40% reduction in phosphorus discharged to the Clark Fork River from the City of Missoula WWTP.

A Voluntary Nutrient Reduction Program (VNRP) for the upper Clark Fork River was negotiated among the Cities of Butte, Deer Lodge and Missoula along with Stone Container Corporation, Montana DEQ, Missoula County and the Tri-State Implementation Council in 1998 and approved by the United States EPA as a TMDL. The VNRP specified the options the dischargers chose to implement to reduce critical summer discharges of the nutrients nitrogen and phosphorous.

Recommended nutrient limits were proposed and accepted by the major dischargers and Montana DEQ in the 1998 Voluntary Nutrient Reduction Plan (VNRP). The agreed upon load limits were also accepted by the United States Environmental Protection Agency (USEPA) in its Total Maximum Daily Load (TMDL) approval in 1998. The in-stream nutrient standards proposed in the VNRP have also been adopted as in-stream water quality standards in ARM 17.30.631(2)(a)(b).

The City of Missoula's updated Wastewater Facility Plan (April, 1999) reflects commitments to nutrient reduction negotiated in the VNRP. The City of Missoula committed to provide a level of treatment equivalent to a conventional biological nutrient removal (BNR) facility. Construction of these improvements to the City of Missoula wastewater treatment plant was completed and fully operational by 2004. The Missoula WWTP BNR treatment process is effectively removing nitrogen and phosphorus even as the population of the City of Missoula has increased and over 3,500 homes using septic systems connected to the WWTP. During the period of the VNRP Missoula's WWTP discharge peaked at just over 1,800 pounds per day of nitrogen in 2002 and over 200 pounds per day of phosphorus in 2003. The BNR upgrade reduced daily discharge to 675 pounds per day of nitrogen and 54 pounds per day of phosphorus. Optimization of WWTP operations has further reduced the discharge of nitrogen and phosphorus to 515 and 38 pounds per day respectively. The City is also moving toward land application of effluent on a poplar plantation to further reduce summer nutrient loads to the Clark Fork River. Following up on the success of the 1989 Phosphate Ordinance, the City of Missoula and Missoula County supported legislation in 2009 that resulted in limits on phosphorus in dishwashing detergent that went into effect in 2010.

The recently released Clark Fork River Nutrient Water Quality Status and Trends Report, 1998-2012 documents improvements in water quality resulting from these efforts. The monitoring station located below the City of Missoula WWTP discharge was the only monitoring site that demonstrated a significant or highly significant downward trend in total and soluble nitrogen and phosphorus as well chlorophyll-a. The history of Missoula and the Clark Fork River demonstrates that continuous effort has been necessary over the last 25 years to reduce nutrients enough to effect measurable reductions in nuisance algae.

Proposed Rule Amendments and Circular DEQ-12A

The Missoula Valley Water Quality District fully supports the proposed nutrient standards contained in proposed circular DEQ-12A. The numeric criteria proposed to be enacted as standards in DEQ-12A are scientifically sound. The approach uses reference stream data within ecoregions to develop numeric standards for total nitrogen and phosphorus within each ecoregion. This method accounts for regional differences in Montana's wadeable streams.

The proposed standards should prevent impairment of many Montana streams. It has taken decades of effort and millions of dollars of investment to reduce nutrient pollution in the Clark Fork River. Measurable standards can also help prioritize investment in improved wastewater treatment or other nutrient source reduction within watersheds. Our experience in Missoula has demonstrated that measurable nutrient standards informed and motivated investment in improved wastewater treatment and also motivated efforts to look for other source controls (Phosphate ordinance and legislation). The City of Missoula also has achieved significant nutrient reduction and maintained affordable sewer rates while the population and economy has grown.

Proposed Rule Amendments and Circular DEQ-12B

The MVWQD recognizes that planning for necessary improvements in wastewater treatment will take time and financing these improvements may a be significant financial burden to many communities. Larger communities like Missoula have advantages in terms of economies of scale that smaller communities do not. The MVWQD suggests DEQ consider developing examples and regional case studies of available technologies and alternative wastewater practices for smaller communities as early as possible to inform evaluation of variances under DEQ-12B. Our experience in Missoula also recognizes that Montanans cares about water quality ant are willing to invest in it.

Sincerely,

Jon Harvala

From: Mark Reinsel
To: DEQ WQP Admin
Subject: Circular DEQ-12B comments
Date: Tuesday, April 01, 2014 1:59:26 PM

Please find attached my comment letter on Circular DEQ-12B.

Thank you,

Mark

Mark Reinsel, Ph.D., P.E.
Apex Engineering, PLLC
4050 Fieldstone Crossing
Missoula, MT 59802
(406) 493-0368 phone/fax
(406) 459-2776 cell
http://apexengineering.us
www.linkedin.com/pub/mark-reinsel/20/5b4/160

Apex Engineering, PLLC
4050 Fieldstone Crossing
Missoula, MT 59802
http://apexengineering.us
Phone: 406-493-0368
Fax: 406-493-0368
mark@apexengineering.us

April 1, 2014

Re: Comments on DEQ Circular 12B

Ms. Carrie Greeley
Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901
deqwqpadmin@mt.gov

To Whom It May Concern:

As a consulting engineer specializing in water treatment, I read with interest the new Circular DEQ-12B and the associated solicitation for public comment. Although not part of the Nutrient Work Group, I have been well aware of its activities over the past several years. I agree with both concepts presented in Circular 12A (some tightening of nutrient standards) and Circular 12B (possible granting of variances to prevent unnecessary economic expense).

While specializing in water treatment technologies for almost 20 years, I constantly research and evaluate new technologies, so I am well aware of the “availability of reasonable alternatives” discussed in New Rule I(5). I believe that experts such as myself will be able to assist DEQ and dischargers in arriving at treatment solutions to either avoid the need for variances or to minimize their duration. One of my consulting clients is Floating Island International, which has a simple, cost-effective technology for nitrogen and phosphorus removal that may be applicable in many situations. I have also designed, evaluated and optimized more “high-tech” approaches for nutrient removal that may be applicable in other cases.

Sincerely,

Mark A. Reinsel, Ph.D., P.E.
President
Apex Engineering, PLLC

From: Mark Vander Meer
To: DEQ WQP Admin
Subject: Public Comment - Proposed Adoption of New Rule Pertaining to Nutrient Standards
Date: Tuesday, April 01, 2014 11:27:25 AM
Attachments: image001.jpg

To: Carrie Greeley
Department of Environmental Quality
1520 E. Sixth Ave
P.O. Box 200901
deqwqpadmin@mt.gov

From: Mark Vander Meer
Restoration Ecologist & Soil Scientist
Watershed Consulting LLC
Missoula, MT

Re: Public Comment - Proposed Adoption of New Rule Pertaining to Nutrient Standards

Dear DEQ,

I am writing to submit comments on the proposed adoption of new rule 1 pertaining to nutrient standards. First, I applaud the DEQ for recognizing and coming to terms with nutrient loading issues. I support the adoption of the rule.

Second, I have worked in the natural resource field for 30 years on many types of projects, including nutrient management. As you are aware, there are many approaches and techniques that can be applied to manage nutrients. I want to enthusiastically endorse one specific method we have found to be especially effective and efficient: floating islands. I have been working with *Floating Island International* of Shepard Montana, to reduce nutrients in ponds, streams, stormwater and sewage treatment facilities and livestock yards. I strongly recommend the DEQ to consider floating islands as a powerful tool used to combat nutrient loading issues, and permit the use of this technology in future projects.

Thanks for considering public comment.

Sincerely,

Mark Vander Meer

Mark Vander Meer
Restoration Ecologist/Forester/Soil Scientist
Watershed Consulting, LLC
Missoula, Montana
(406) 541-2565
www.watershedconsulting.com

From: Rowlands, Mike
To: DEQ WQP Admin
Subject: Comments concerning proposed Nutrient Standards
Date: Monday, March 31, 2014 12:47:08 PM

Carrie:

Attached please find Otter Creek Coal's comment concerning the Proposed Nutrient Standards. Thank you for the opportunity to comment.

Mike

Otter Creek

Mike Rowlands
Director, Otter Creek Operations

mrowlands@archcoal.com

March 31, 2014

Montana Department of Environmental Quality
Carrie Greeley
Montana Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

Dear Ms. Greeley:

Otter Creek Coal appreciates the opportunity to comment on the proposed numeric nutrient standards, the proposed new variance rule, and the associated circulars. We hope you find these comments helpful. If you have questions or would like to discuss these comments, please let me know. Our team here at Otter Creek Coal would welcome the opportunity to further explain our concerns and work with you to solve some of the issues noted.

Numeric Nutrient Standards

By adopting the proposed rule amendments, Montana is committing industry and other dischargers to nutrient limits that cannot be attained with current or foreseeable technology. The large disparity between the numeric limits and the limits for a general variance show the severity of this issue. Adoption and implementation of numeric standards at levels below the level of viable treatment technology and in advance of numeric nutrient standards adoption by most other states is problematic for industry in Montana.

It is not clear how storm water permits could be affected by the numeric nutrient standards. Although the current general storm water permits and most individual storm water permits are based on BMP compliance, DEQ has not indicated how the seasonal nutrient standards will be applied to storm water discharges. It should be clarified that numeric standards will not be extended to storm water permits before finalizing this rule.

Further, the non-severability clauses, as written in the proposed rule amendments (Admin. R. Mont. 17.30.619(2) and 17.30.715(4)), are only focused on EPA action adverse to the state and do not address the situation an applicant may face if a variance is granted by the DEQ, then later rejected by the EPA. The non-severability clauses should be revised to correct this issue.

Guidance Document

The Board and Department Public Comment Notices do not reference the guidance document, *Base Numeric Nutrient Standards Implementation Guidance*, Draft 1.3 (December 2013), nor is the guidance document referenced by rule or in the circulars. This document is important in understanding how the standards will be interpreted and implemented; therefore, it is appropriate to include the guidance document and allow public review and comment on it in conjunction with the rule package. For example, the important term "Limits of technology" is only defined in the guidance document and not in the new rule, rule amendments, or circulars and therefore may not have been reviewed by the public.

Otter Creek Coal LLC
P.O. Box 7152
Billings, MT 59103
office: 406.245.0990
archcoal.com

Downstream Compliance
Circular DEQ-12A, endnote 2 states that "base numeric nutrient standards of the downstream reaches or other downstream water bodies must continue to be maintained." However, the extent of downstream compliance that a discharger will be held accountable for is not defined. The guidance document reiterates this issue in section 6.2. The rule, circular and guidance document should be amended to clarify that the dischargers will only be accountable within the mixing zone or until the next source of nutrients –whether it be another point source, a non-point source, or natural source.

Variances
The legislative intent is that variances would be available to all dischargers; however, the proposed rule, rule amendments, and circulars are silent on the availability of the general variance to new and increased discharges that are private entities. DEQ has included a comment on this issue in the Guidance document, but this provides a lesser degree of certainty as to how this issue will be addressed in permitting.

Although the variances may be valid for up to twenty years, they require review through a public rulemaking process every three years. This adds too much uncertainty where industry and companies such as Arch Coal, need long term stability commensurate with their long term investment.

Proof of Economic Impact
Specific to the individual variance process based on economic impact, the proposed rule, DEQ-12B, and the guidance document should not rely on the 1995 EPA draft guidance. Instead, they should be amended to require a simpler showing of economic impact, perhaps a cost increase, for two reasons. First, this amendment would align more closely with the legislative intent behind the statute. In 2011, when the legislature passed Senate Bill 367 (codified as Montana Code Annotated § 75-5-313), discussion of the "substantial and widespread economic impacts" was in the context of a statewide basis – not an individual basis. The legislature established and the statute codifies the finding of economic impact on a statewide basis; therefore, there should be no need for an individual finding to qualify for a variance:

> The department, in consultation with the nutrient work group, shall develop guidelines for individual nutrient standard variances to **ensure that the economic impacts** from base numeric nutrient standards on **public and private** systems are **equally and adequately** addressed. In developing those guidelines, the department and the nutrient work group shall consider economic impacts appropriate for application within Montana, **acknowledging** that advanced treatment technologies for removing nutrients **will result in significant and widespread economic impacts.**

Mont. Code Ann. §75-5-313(2)(a) (emphasis added).

> Because the treatment of wastewater to base numeric nutrient standards **would result in substantial and widespread economic impacts on a statewide basis**, a permittee...may ... apply for a general nutrient variance.

Mont. Code Ann. § 75-5-313(5)(a) (emphasis added).

See also Montana House of Representatives, Natural Resources Committee, Hearing on Senate Bill 367, Audio Record at 31:36 (March 21, 2011). Translating the discussion of substantial and widespread economic impacts from a statewide basis to a requirement that each individual applicant must show substantial and widespread economic impacts is inappropriate. Certainly, the language of the statute and the legislative history show that the legislature did not intend for economically viable industries to be singled out for compliance with numeric standards that may not be attainable given current and immediately foreseeable technology.

The second reason to amend the economic requirement is based on Montana's liberal public disclosure of agency documents. Because Montana's Constitutional Right to Know is much broader than federal disclosure under the Freedom of Information Act, this issue was likely not considered in the EPA guidance but warrants special attention here in Montana. To show substantial and widespread economic impact, the new rule and the DEQ guidance rely on economic data from the applicant, specifically the applicant's cash flow to total debt ratio and their debt to equity ratio. While Montana Code Annotated § 75-5-314 provides some measure of assurance that the DEQ will protect trade secrets, there is no guarantee that the company's financial data will not be subject to public disclosure. Given Montana's strong Right to Know Constitutional provision, it is unlikely that non-trade secret information, such as financial data, would be protected once it is in the DEQ's possession. Therefore, a simpler showing should be required, based on cost increases.

Types of Variances

Montana Code Annotated § 75-5-313(2) requires a variance for "economic impacts, limits of technology, or both." However, the proposed rule and DEQ-12B do not clearly present a variance for the limits of technology and they do not address dischargers who qualify for a variance in both categories. The language in the guidance document articulates this better than DEQ-12B. Further, the language of DEQ-12B adds another layer to the qualification by referring to the "highest attainable condition within the receiving water." This verbiage is not necessary. Variances should be granted based on the elements of the statute without adding this requirement.

Additionally, the alternate variance required by Mont. Code Ann. § 75-5-313(10)(a) is not clearly defined in the proposed rule or in DEQ-12B. An addition or revision should be made to account for alternate variances. Section 3.2 of DEQ-12B seems to speak more to an alternate variance than an individual variance.

Again, thank you for the opportunity to outline our concerns to this proposed rule.

Sincerely,
OTTER CREEK COAL, LLC

William M. Rowlands

William M. Rowlands,
Director of Operations

From:	Mathieus, George
To:	Urban, Eric; Greeley, Carrie
Subject:	FW: Comments concerning proposed Nutrient Standards
Date:	Tuesday, April 01, 2014 8:19:50 AM

From: Johnson, Elois
Sent: Monday, March 31, 2014 1:02 PM
To: Orr, Katherine; Mathieus, George; Suplee, Mike
Subject: FW: Comments concerning proposed Nutrient Standards

Attached is a comment I received pertaining to MAR Notice No. 17-356 regarding proposed nutrient standards.

Elois M. Johnson
Paralegal
Department of Environmental Quality
PO Box 200901
Helena, MT 59620-0901
Telephone: 406.444.2630
Fax: 406.444.4386
Email: ejohnson@mt.gov

From: Rowlands, Mike [mailto:MRowlands@archcoal.com]
Sent: Monday, March 31, 2014 12:45 PM
To: Johnson, Elois
Subject: Comments concerning proposed Nutrient Standards

Elois:

Attached please find Otter Creek Coal's comment concerning the Proposed Nutrient Standards. Thank you for the opportunity to comment.

Mike

Otter Creek

Mike Rowlands
Director, Otter Creek Operations

mrowlands@archcoal.com

March 31, 2014

Montana Board of Environmental Review
Elois Johnson, Paralegal
Montana Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

Dear Ms. Johnson:

Otter Creek Coal appreciates the opportunity to comment on the proposed numeric nutrient standards, the proposed new variance rule, and the associated circulars. We hope you find these comments helpful. If you have questions or would like to discuss these comments, please let me know. Our team here at Otter Creek Coal would welcome the opportunity to further explain our concerns and work with you to solve some of the issues noted.

Numeric Nutrient Standards

By adopting the proposed rule amendments, Montana is committing industry and other dischargers to nutrient limits that cannot be attained with current or foreseeable technology. The large disparity between the numeric limits and the limits for a general variance show the severity of this issue. Adoption and implementation of numeric standards at levels below the level of viable treatment technology and in advance of numeric nutrient standards adoption by most other states is problematic for industry in Montana.

It is not clear how storm water permits could be affected by the numeric nutrient standards. Although the current general storm water permits and most individual storm water permits are based on BMP compliance, DEQ has not indicated how the seasonal nutrient standards will be applied to storm water discharges. It should be clarified that numeric standards will not be extended to storm water permits before finalizing this rule.

Further, the non-severability clauses, as written in the proposed rule amendments (Admin. R. Mont. 17.30.619(2) and 17.30.715(4)), are only focused on EPA action adverse to the state and do not address the situation an applicant may face if a variance is granted by the DEQ, then later rejected by the EPA. The non-severability clauses should be revised to correct this issue.

Guidance Document

The Board and Department Public Comment Notices do not reference the guidance document, *Base Numeric Nutrient Standards Implementation Guidance*, Draft 1.3 (December 2013), nor is the guidance document referenced by rule or in the circulars. This document is important in understanding how the standards will be interpreted and implemented; therefore, it is appropriate to include the guidance document and allow public review and comment on it in conjunction with the rule package. For example, the important term "Limits of technology" is only defined in the guidance document and not in the new rule, rule amendments, or circulars and therefore may not have been reviewed by the public.

Otter Creek Coal LLC
P.O. Box 7152
Billings, MT 59103
office: 406.245.0990
archcoal.com

Downstream Compliance
Circular DEQ-12A, endnote 2 states that "base numeric nutrient standards of the downstream reaches or other downstream water bodies must continue to be maintained." However, the extent of downstream compliance that a discharger will be held accountable for is not defined. The guidance document reiterates this issue in section 6.2. The rule, circular and guidance document should be amended to clarify that the dischargers will only be accountable within the mixing zone or until the next source of nutrients –whether it be another point source, a non-point source, or natural source.

Variances
The legislative intent is that variances would be available to all dischargers; however, the proposed rule, rule amendments, and circulars are silent on the availability of the general variance to new and increased discharges that are private entities. DEQ has included a comment on this issue in the Guidance document, but this provides a lesser degree of certainty as to how this issue will be addressed in permitting.

Although the variances may be valid for up to twenty years, they require review through a public rulemaking process every three years. This adds too much uncertainty where industry and companies such as Arch Coal, need long term stability commensurate with their long term investment.

Proof of Economic Impact
Specific to the individual variance process based on economic impact, the proposed rule, DEQ-12B, and the guidance document should not rely on the 1995 EPA draft guidance. Instead, they should be amended to require a simpler showing of economic impact, perhaps a cost increase, for two reasons. First, this amendment would align more closely with the legislative intent behind the statute. In 2011, when the legislature passed Senate Bill 367 (codified as Montana Code Annotated § 75-5-313), discussion of the "substantial and widespread economic impacts" was in the context of a statewide basis – not an individual basis. The legislature established and the statute codifies the finding of economic impact on a statewide basis; therefore, there should be no need for an individual finding to qualify for a variance:

> The department, in consultation with the nutrient work group, shall develop guidelines for individual nutrient standard variances to **ensure that the economic impacts** from base numeric nutrient standards on **public and private** systems are **equally and adequately** addressed. In developing those guidelines, the department and the nutrient work group shall consider economic impacts appropriate for application within Montana, **acknowledging** that advanced treatment technologies for removing nutrients **will result in significant and widespread economic impacts.**

Mont. Code Ann. §75-5-313(2)(a) (emphasis added).

> Because the treatment of wastewater to base numeric nutrient standards **would result in substantial and widespread economic impacts on a statewide basis**, a permittee…may … apply for a general nutrient variance.

Mont. Code Ann. § 75-5-313(5)(a) (emphasis added).

See also Montana House of Representatives, Natural Resources Committee, Hearing on Senate Bill 367, Audio Record at 31:36 (March 21, 2011). Translating the discussion of substantial and widespread economic impacts from a statewide basis to a requirement that each individual applicant must show substantial and widespread economic impacts is inappropriate. Certainly, the language of the statute and the legislative history show that the legislature did not intend for economically viable industries to be singled out for compliance with numeric standards that may not be attainable given current and immediately foreseeable technology.

The second reason to amend the economic requirement is based on Montana's liberal public disclosure of agency documents. Because Montana's Constitutional Right to Know is much broader than federal disclosure under the Freedom of Information Act, this issue was likely not considered in the EPA guidance but warrants special attention here in Montana. To show substantial and widespread economic impact, the new rule and the DEQ guidance rely on economic data from the applicant, specifically the applicant's cash flow to total debt ratio and their debt to equity ratio. While Montana Code Annotated § 75-5-314 provides some measure of assurance that the DEQ will protect trade secrets, there is no guarantee that the company's financial data will not be subject to public disclosure. Given Montana's strong Right to Know Constitutional provision, it is unlikely that non-trade secret information, such as financial data, would be protected once it is in the DEQ's possession. Therefore, a simpler showing should be required, based on cost increases.

Types of Variances

Montana Code Annotated § 75-5-313(2) requires a variance for "economic impacts, limits of technology, or both." However, the proposed rule and DEQ-12B do not clearly present a variance for the limits of technology and they do not address dischargers who qualify for a variance in both categories. The language in the guidance document articulates this better than DEQ-12B. Further, the language of DEQ-12B adds another layer to the qualification by referring to the "highest attainable condition within the receiving water." This verbiage is not necessary. Variances should be granted based on the elements of the statute without adding this requirement.

Additionally, the alternate variance required by Mont. Code Ann. § 75-5-313(10)(a) is not clearly defined in the proposed rule or in DEQ-12B. An addition or revision should be made to account for alternate variances. Section 3.2 of DEQ-12B seems to speak more to an alternate variance than an individual variance.

Again, thank you for the opportunity to outline our concerns to this proposed rule.

Sincerely,
OTTER CREEK COAL, LLC

William M. Rowlands

William M. Rowlands,
Director of Operations

Arch. Hydrobiol. Spec. Issues Advanc. Limnol. **51**, p. 105–114, June 1998
Lake Erken – 50 years of limnological research

C:N:P ratios of freshwater benthic algae

MARIA KAHLERT

with 4 tables

Abstract: The measurement of internal C:N:P (carbon : nitrogen : phosphorus) ratios can be used as a tool to determine the nutrient status of freshwater benthic algae because it is both an easy and reliable method. It is possible to detect a severe nutrient limitation with the C:N:P method alone. A moderate limitation should be confirmed by other bioassays. This is shown by a review of published studies. C:N:P ratios of freshwater benthic algae at optimal (158 : 18 : 1 molar basis) and nutrient limiting conditions were derived from the literature. These values were used to state the nutrient status of freshwater benthic algae in Lake Erken, Sweden. The proposed values should be confirmed by more studies both in the laboratory and *in situ*, since there is still a lack of knowledge.

Introduction

The importance of substrata as nutrient sources for freshwater benthic algae is still a field of challenging questions. I chosed to compare the nutrient status of freshwater benthic algae growing on different substrates in Lake Erken, Sweden. Benthic algae tend to grow in a heterogeneous pattern (LOWE 1996). Therefore, it is necessary to process a great number of samples to characterize the scales of temporal and spatial heterogeneity that can be observed in these communities. Consequentely, I needed a simple and reliable method for rapid assessment of the nutrient status of benthic algae.

Many methods are available to measure the nutrient status of algae (e.g. measurement of enzyme activities or nutrient uptake rates). The simplest method is the estimation of C : N : P (carbon : nitrogen : phosphorus) ratios. HECKY & KILHAM (1988) and HECKY et al. (1993) recommended the measurement of C : N : P ratios as a good method to estimate nutrient status of phytoplankton. BJÖRNSÄTER & WHEELER (1990) recommended the C : N : P method for marine benthic algae, and DUARTE (1990) recommended the method for marine seagrass. I will demonstrate in this paper that one can use the C : N : P method for freshwater benthic algae as well.

C : N : P method

The principle of the C : N : P method is a comparison between the measured internal C : N : P ratio of the algae and an optimal ratio. This optimal ratio would occur in the algae when nutrients are not limiting nor in surplus.

Under P or N surplus algal P or N tissue level could be high, if the algae are able to store these nutrients. BJÖRNSÄTER & WHEELER (1990) showed that algae try to maintain a balanced internal N : P ratio. A surplus would not be optimal since the algae would waste energy on storage. Under P or N limitation, algal P or N tissue level would be low, because of the lack of sufficient ambient nutrients.

Therefore, under non-optimal (nutrient surplus or limitation) conditions algal C : N : P ratios would differ from the optimal ratio. For the effective use of the C : N : P method, the optimal ratio of the investigated algal population should be known. This optimal ratio is very difficult to obtain for several reasons. In practice, the "real optimal ratio" is replaced by an

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empirically estimated ratio, derived from the literature. Literature values are rather "ratios occurring under obviously non-limiting conditions", but in the interest of simplicity I am going to call them "optimal ratios".

The first advantage of the C : N : P method is its simplicity. The determination of C and N is easy with automated CN analyzers. The determination of P by persulfate acid digestion is not complicated either. The researcher can store the samples after filtration. In this way it is possible to get a high number of replicates.

Second, the "nutrient history" of the algae is included, so not only the ambient nutrient situation is investigated, but also some time before sampling (BJÖRNSÄTER & WHEELER 1990, FONG et al. 1994). This gives a better picture of the real situation for the algal community.

Third, the nutrient situation can be studied without any manipulation. All of the physical, chemical and biological parameters are natural.

However, the C:N:P method has also disadvantages. Detritus and other non-algal material can influence the measured C:N:P ratios. Detritus, for example has more C in relation to N and P than algae (ALLAN 1995, MAKAREVICH et al. 1992). Bacteria on the other hand can play a role as a significant P-pool when their biomass is high (RHEE 1972, PETTERSSON et al. 1993). Separation is one possibility to solve these problems. Larger detritus and animals can be sorted out by hand. Filtration can separate algae from bacteria and small detritus. Some authors recommend filters with a pore size of 0.5–0.8 µm (SAKSHAUG 1980, PORTER et al. 1988), others use filters with a pore size of 3 µm (HARRIS 1986, ISTVÁNOVICS et al. 1990, PETTERSON et al. 1993). Also the use of a microscope as well as the calculations of the C:Chl *a* ratio can help to assess the amount of non-algal material in the sample. Furthermore, HECKY et al. (1993) considered in the case of phytoplankton, the impact of detritus on the C : N : P ratios of minor significance.

However, it is still a subject of controversy wether the C:N:P ratios can be altered by other factors than nutrients, for example by light or by grazing.

WYNNE & RHEE (1986) conclude that light can strongly influence the C : N : P ratios. The tissue content of C and N might be influenced more than the P content (WYNNE & RHEE 1986, NIELSEN 1992). Different algal species can obviously react in an opposite way: some species decrease their C and N content under increasing light (NIELSEN 1992), others increase it (MAGNUSSON et al. 1996). But even if the content is altered, the ratio remains often almost unchanged (GOLDMAN 1986, MAGNUSSON et al. 1996). Based on evidence provided in the literature, I concluded that the influence of light in a mixed natural community is of minor importance to the C : N : P ratios.

Grazing can also change the C : N : P ratios, but the pattern is as unclear as in the impact of ligh. Grazed communities might be higher in nutrients with lower C : N ratios (PETERSON et al. 1993, ROSEMOND 1993).

Another problem of the C : N : P method is the occurrence of different optimal ratios for different algal species (FAIRCHILD et al. 1985, BJÖRNSÄTER & WHEELER 1990, DUARTE 1990, HECKY et al. 1993, CHOPIN et al. 1996). An algal community could have a sufficient nutrient supply despite a high C:N:P ratio, only because the algal species of the community are adapted to low nutrient concentrations. But HECKY & KILHAM (1988) and HECKY et al. (1993), DUARTE (1990) and BJÖRNSÄTER & WHEELER (1990) showed that the interspecific variation of C:N:P ratios is smaller than the intraspecific variation. Therefore a difference of C:N:P ratios indicates rather a different nutrient situation than different algal species.

Despite all the mentioned doubts, the C : N : P method is still used successfully to estimate the nutrient status of marine and freshwater phytoplankton and marine benthic algae. The method is widely accepted (e.g. REDFIELD 1958, RHEE 1978, GOLDMAN et al. 1979, GORDON et al. 1981, ATKINSON & SMITH 1983, RAO & INDUSEKHAR 1987, FAGANELI et al. 1988, PAASCHE & ERGA 1988, HECKY & KILHAM 1988, HECKY et al. 1993, BJÖRNSÄTER &

Wheeler 1990, Pihl et al. 1996). Optimally growing mixed natural phytoplanktic populations almost always have the Redfield ratio (C:N:P = 106:16:1 molar basis) as an optimal ratio (Hecky & Kilham 1988, Hecky et al. 1993 and references therein). Therefore, the Redfield ratio is considered a community wide optimum nutrient ratio (Borchard 1996).

My intention is to judge, whether a researcher can use the C:N:P method for freshwater benthic algae as well. To my knowledge this is the first time that facts about the use of the C:N:P method for freshwater benthic algae are collected.

I searched for literature on C:N:P ratios of freshwater benthic algae. I looked especially for studies dealing with the change of algal C:N:P ratios after nutrient enrichment. A clear change of the C:N:P ratios after enrichment would support the reliability of the C:N:P method. I searched also for studies that connected the C:N:P method and other bioassays. If the outcome would be the same, this would further strengthen the validity of the C:N:P method. I tried to calculate median ratios for optimal nutrient status and limitation. I tried furthermore to identify distinct C:N:P ratios that would indicate limitation clearly. I searched for differences between algal taxa. I found that studies on nutrient status of freshwater benthic algae employing the C:N:P method are rather few. The studies are also very different: sometimes they give many values, sometimes only average results. I took medians from the reported values in the literature. When necessary, I converted the reported values to C:N:P ratios (molar basis).

Results and discussion

C:N:P ratios and enrichment studies

Table 1 shows the studies on freshwater benthic algae reporting C:N:P ratios with and without artificial or natural nutrient enrichment.

Peterson et al. (1993) fertilized a whole tundra river with P. They reported an increase in C, N and P stocks of the diatom mat present and an increase of P relative to C and N. They also noticed a sharp increase in Chl *a* when they inhibited grazing.

Rosemond (1993) used small nutrient (P + N) releasing tiles in Walker Branch, a stream in Tennessee. She also recorded the influence of light and grazers. Algae (diatoms, green algae and bluegreen algae) growing on fertilized surfaces had on average lower C:P and N:P ratios. She noticed an increase in Chl *a* when light was increased and grazers were absent.

Neil & Jackson (1982) fertilized a shoreline site of Lake Erie with P. The present *Cladophora* reacted with decreased N:P ratios and a rapid growth.

Rosemarin (1982) used short term batch cultures of *Cladophora* to show the effect of P + N enrichment on tissue nutrient level. N:P ratios decreased after addition of P + N. Growth was not determined.

Table 1. Change of C:N:P ratios (molar basis) and biomass of freshwater benthic algae after an increase in ambient nutrients (P and/or N).

algal community	increased nutrient	kind of enrichment	C:N	C:P	N:P	change in biomass	source
diatoms *in situ*	P	artificial	10→ 9	207→121	22→13	increased	Petersen et al. 1993
diatoms *in situ*	P+N	artificial	20→23	443→301	22→13	no change	Rosemond 1993
green/bluegreen algae *in situ*	P+N	artificial	12→12	603→465	49→40	increased	Rosemond 1993
Cladophora in situ	P	artificial	*	*	72→58	increased	Neil & Jackson 1982
Cladophora in vitro	P	artificial	*	*	19→ 8	*	Rosemarin 1982
Cladophora in situ	N	natural	34→11	382→220	12→20	*	Lohman & Priscu 1992
Cladophora in situ	P(+N)	natural	*	*	42→17	no change	Freeman 1986
epiphyton *in situ*	P	natural	9→ 9	439→158	51→18	decreased	Sarvala et al. 1982

If necessary, original values were converted into median C:N:P (molar basis) values. * not reported.

Lohman & Priscu (1992) monitored the *Cladophora* population of Clark Fork of the Columbia River, Montana, influenced by wastewater. They recorded decreased C : N and C : P ratios and increased N : P ratios after a N peak in the water. They did not determine growth.

Freeman (1986) studied *Cladophora* in the Manawatu River, New Zealand and observed ambient nutrient concentrations together with C : N : P ratios and biomass. He showed a decrease in N : P ratios after a P peak in the ambient water, but biomass was primarily controlled by river flow. Freeman (1986) did not see a distinct correlation of low N : P ratios and high biomass.

Sarvala et al. (1982) monitored the epiphyton of an oligotrophic lake in Southern Finland. A low ambient P concentration in spring and early summer was followed by high N : P ratios. An increase of P later in the year led to lower N : P ratios but not to higher growth, most likely because of the low temperatures in autumn.

All of the presented studies show that the predicted change in C : N : P ratios is due to nutrient enrichment. The studies with an artificial nutrient enrichment also showed an increase in algal biomass after enrichment.

C : N : P ratios and other bioassays

Some studies use both C : N : P ratios and other tests to estimate the nutrient status of freshwater benthic algae.

In their studies on *Cladophora* of the Great Lakes, Auer & Canale (1980, 1982) and Canale et al. (1982) stated that P uptake is closely coupled both to external P as well as internal P. They developed a model showing that the P uptake depends on the external P concentration until the internal stores are filled and a feedback mechanism inhibits further P uptake. The model calculated a high P uptake at low internal P levels. Rosemarin (1982) showed a similar pattern for N uptake.

Freeman (1986) found a close coupling of P and N tissue levels and other tests like P and N nutrient uptake, estimation of luxury P and APA (alkaline phosphatase activity). He suggested seasonal P limitation in his study and based his suggestion on high N : P ratios, a high P uptake, a high APA and low luxury P.

Rosemond (1993) found a decreased APA together with decreased C : P and N : P ratios under nutrient (P + N) enrichment. Lohman & Priscu (1992) reported a low N uptake during a high ambient N concentration and at the same time a decrease of the C : N and an increase of the N : P ratios.

Reuter et al. (1986) investigated an N-deficient lake, Lake Tahoe in California-Nevada. They recorded a relatively high C : N ratio in the bluegreen algae present. Later in the season they measured decreased C : N ratios along with a high nitratreductase activity (high use of nitrate). The authors had expected a high nitratreductase activity under N limitation and had no explanation for their results. However, I consider the measurement of nitratreductase activity not to be a useful method to estimate N limitation. It is rather a measurement of nitrate usage. Nitratreductase activity can indeed indicate a shortage of ammonium, but does not necessarily indicate a N limitation. The algae in the study of Reuter et al. (1986) filled their N stores with nitrate instead of ammonium. The decreased C : N ratios indicated correctly that N limitation no longer persisted.

Four of the presented studies that investigated nutrient limitation reported a distinct correlation between the C : N : P method and other bioassays. One study failed to show this correlation. However, in that case the chosen bioassay was not useful to show a N limitation.

C : N : P ratios of freshwater benthic algae

Table 2 shows the C : N : P ratios of freshwater benthic algae measured *in situ*. I included the nutrient conditions for the investigated algal community, if the researcher made a comment on it.

The reported values for C:N:P ratios of freshwater benthic algae have a broad range. However, regardless of the nutrient situation, most of the values reported for marine benthic algae are higher (ATKINSON & SMITH 1983). Most values of freshwater benthic algae are reports on diatoms and *Cladophora* communities. The reported values for diatoms are in general lower than those for *Cladophora*, indicating a lower structural carbon and/or a higher nutrient content for diatoms. The ratios I measured in Lake Erken represent no exception.

The values of Table 3 and 4 are derived from Table 2. I could not use all values to calculate medians, because some researchers did not report the nutrient situation. Moreover, most of the researchers estimated the nutrient situation only roughly. Furthermore, even if in some cases I could calculate medians from several values, I had to rely on a few average values in other cases. Therefore the tables present only a gross overview of the field.

Table 3 shows the C : N : P ratios at nutrient limitation and at non-limiting conditions.

At non-limiting conditions the algae should have their optimal C : N : P ratio. The optimal median value is 158 : 18 : 1, which is close to the Redfield ratio. However, the estimation of the theoretical optimal C : N : P ratio suffers from a proper estimation of the nutrient conditions. Moreover, the investigated studies made no distinction between optimal and surplus conditions. C : N : P ratios would be lower than the theoretical optimal ratio under surplus conditions. The estimated optimal C : N : P ratio would then be too low, if studies with surplus conditions are included in the calculation.

Under P limitation the C : P value is slightly higher and the N : P value is clearly higher than the optimal value. The C : N ratio is the same as for optimal conditions. The relatively low C : P value is likely a result of the paucity of the values.

The median C : N ratio under N limitation is clearly higher than the optimal value, and there is no overlap with the optimal or P limited values. The N : P value is clearly lower than the optimal value. The C : P ratio is not only higher than the optimal value but it is even higher than under P limitation. A P and N limitation is shown by overall increased C : N : P ratios. The values lie between those for distinct P or N limitation. The C : P values under P and N limitation are even higher than under P or N limitation alone.

Table 4 shows ratios that indicate a nutrient limitation clearly. The C : P value for P limitation is somewhat weak as stated above. It is prudent to consider all three ratios C : N, C : P and N : P to estimate the nutrient situation. But it is also apparent that the calculated values for freshwater benthic algae fit very well to literature reports: ALLAN (1995) indicated N limitation with N : P < 10 and P limitation with N : P > 30 in freshwater benthic algae. The N : P value is the same for benthic marine algae (BJÖRNSÄTER & WHEELER 1990, CHOPIN et al. 1996). Phytoplankton values are lower (HECKY et al. 1993 and references therein) indicating that marine and freshwater benthic algae could have a different metabolism than phytoplankton. This is valid for both marine and freshwater algae. An explanation could be the possibly higher structural carbon of benthic algae (ATKINSON & SMITH 1983).

Nutrient situation of benthic algae and phytoplankton in Lake Erken

I assessed the nutrient status of the epilithon and epiphyton of Lake Erken, Sweden by relating the ratios found in 1996 to Table 4. The epilithon were slightly N and P limited throughout the year, the P limitation was somewhat higher in summer. The epiphyton were N limited only in autumn, if at all. The values showed instead a severe P limitation in summer.

It is now possible to compare the nutrient situation of the benthic algae with the nutrient situation of the phytoplankton of Lake Erken, investigated in previous studies (ULÉN 1971, PETTERSSON 1980, ISTVÁNOVICS et al. 1992, PETTERSSON, et al. 1993). In general, the phytoplankton of Lake Erken are severely P limited in spring/summer and can be N limited in late summer/autumn. The pattern is therefore the same as in the epiphyton. The onset of the P limi-

Table 2. C : N : P ratios (molar basis) of freshwater benthic algae estimated *in situ* and their nutrient status as stated by the authors.

algal community	nutrient status	C : N	C : P	N : P	locality of investigation	source
diatoms	not limited	8–9	99–142	11–15	Kuparuk River, Alaska	PETERSEN et al. 1993
	P limited	9–11	168–246	13–31	Kuparuk River, Alaska	PETERSEN et al. 1993
	P & N limited	20	443	22	small tributary of Walker Branch, Tennessee	ROSEMOND 1993
	*	23	301	13	small tributary of Walker Branch, Tennessee	ROSEMOND 1993
	*	11	89	8	Lake Naroch, Belorussia	MAKAREVICH et al. 1992
	*	6	*	*	Lake Tahoe, California-Nevada	LOEB 1981
	*	4	*	*	shallow pond, Michigan	HUNTER 1980
Cladophora spec.	not limited	10–11	231–369	23–32	Lake Erie, Ohio	LORENZ & HERDENDORF 1982
	not limited	*	*	16–18	Manwatu River, New Zealand	FREEMAN 1986
	P limited	*	*	33–50	Manwatu River, New Zealand	FREEMAN 1986
	P limited	*	*	47	Lake Ontario, Canada	ROSEMARIN 1982
	P limited	*	*	72	Lake Erie, Canada	NEIL & JACKSON 1982
	N surplus	11	220	20	Columbia River, Montana	LOHMAN & PRISCU 1992
	N limited	30–51	358–485	10–12	Columbia River, Montana	LOHMAN & PRISCU 1992
	P & N limited	*	*	29–44	Manwatu River, New Zealand	FREEMAN 1986
	*	10	208	20	Lake Erken, Sweden	ULÉN 1971
greenalgae	*	9	*	*	shallow pond, Michigan	HUNTER 1980
	N limited	16	192	12	streams on Signy Island, Antarctica	HAWES 1989
green- and bluegreen algae	P & N limited	12	603	49	small tributary of Walker Branch, Tennessee	ROSEMOND 1993
	*	12	465	40	small tributary of Walker, Branch, Tennessee	ROSEMOND 1993
bluegreen algae	N limited	14	*	*	Lake Tahoe, California-Nevada	REUTER et al. 1986
	*	14	*	*	Lake Tahoe, California-Nevada	LOEB 1981
diatoms, green- and bluegreen algae	*	12–17	31–176	3–12	Lake Naroch, Belorussia	MAKAREVICH et al. 1992

	N limited	16	192	12	shallow pond, Michigan	HUNTER 1980
green- and bluegreen algae	P & N limited	12	603	49	streams on Signy Island, Antarctica	HAWES 1989
					small tributary of Walker Branch, Tennessee	ROSEMOND 1993
	*	12	465	40	small tributary of Walker, Branch, Tennessee	ROSEMOND 1993
bluegreen algae	N limited	14	*	*	Lake Tahoe, California-Nevada	REUTER et al. 1986
	*	14	*	*	Lake Tahoe, California-Nevada	LOEB 1981
diatoms, green- and bluegreen algae	*	12–17	31–176	3–12	Lake Naroch, Belorussia	MAKAREVICH et al. 1992

Table 2. Continued.

algal community	nutrient status	C : N	C : P	N : P	locality of investigation	source
Nitella	*	26	312	12	lake (not specified)	MARTINOVA 1993
epiphyton	not limited	9	158	18	Lake Pääjärvi, Finland	SARVALA et al. 1982
	P limited	9	439	51	Lake Pääjärvi, Finland	SARVALA et al. 1982
epiphyton	?				Lake Erken, Sweden	KAHLERT, unpubl. data
spring		10	194	20		
summer		8	264	38		
autumn		12	196	19		
epilithon	?				Lake Erken, Sweden	KAHLERT, unpubl. data
spring		12	249	20		
summer		12	266	25		
autumn		12	275	22		

If necessary, original values were converted into median C : N : P (molar basis) values. * not reported.

Table 3. C : N : P ratios (molar basis) of freshwater benthic algae indicating optimal, P and/or N limited conditions. Median values and range derived from Table 2.

nutrient status	C : N	C : P	N : P
optimal	9	158	18
	(8–11)	(99–369)	(11–32)
P limited	9	246	47
	(9–11)	(168–439)	(13–72)
N limited	23	358	12
	(14–51)	(192–485)	(10–12)
P & N limited	16	523	37
	(12–20)	(443–603)	(22–49)

Table 4. C : N : P ratios (molar basis) indicating nutrient limitation.

nutrient status		freshwater benthic algae	marine benthic algae	freshwater phyto-plankton	marine phyto-plankton
P limited	C : P	>369		>129	>150
	N : P	> 32	>31	> 22	> 20
N limited	C : N	> 11		> 8	
	N : P	< 12	<12		< 10
P & N limited	C : N : P	all values relatively high			

source: freshwater benthic algae: Table 2; marine benthic algae: BJÖRNSÄTER & WHEELER 1990, CHOPIN et al. 1996; phytoplankton: HECKY et al. 1993 and references therein.

tation seems to be somewhat earlier in the phytoplankton than in the epiphyton. The epilithon on the other hand have a completely different nutrient status. The P limitation in summer is not severe and the N limitation occurs throughout the whole year.

Conclusions

Enrichment studies and studies connecting the C : N : P method with other bioassays showed that the measurement of C : N : P ratios is a reliable method to estimate the nutrient status of freshwater benthic algae. Some of the reported values did not fit, but the general pattern was clear.

The optimal ratio for freshwater benthic algae is 158 : 18 : 1 with a relatively large variation.

It is possible to estimate severe P limitation with N : P ratios and N-limitation with C : N and N : P ratios (see Table 4). Very high C : N : P ratios were always associated with nutrient limitation.

If the algae are only slightly limited it is not so easy to detect this limitation. The researcher has to consider the whole situation, including the ambient nutrient level, the amount of detritus, bacteria, grazing, light and the existing algal taxa. One must consider all three C : N : P values (C : N, C : P, N : P) and should combine the C : N : P method with other bioassays.

The present overview can only be preliminary, because the reported values are too few and there are no laboratory studies on C : N : P ratios of freshwater benthic algae. Additional attention should be given to the relationship of growth rate to C : N : P ratios. Although the calculation of the medians suffers from proper estimations of the nutrient situation and from a distinction between optimal and surplus conditions; the estimated values fitted well with those reported from the literature.

We need more information about optimal C : N : P ratios. It is most reliable to derive them from laboratory studies, therefore we have to improve the culture of freshwater benthic algae. More studies both in laboratory and *in situ* are necessary to address the lack of knowledge on the nutrient status of freshwater benthic algae.

In general I consider the measurement of C : N : P ratios as a useful method to scan the nutrient status of freshwater benthic algae since it is an easy and reliable method.

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Address of the author:

MARIA KAHLERT, Institute of Limnology, University of Uppsala, Norbyvägen 20, SE-75236 Uppsala, Sweden, e-mail: maria.kahlert@limno.uu.se

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Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met in 2011/2012

April 26, 2012

Prepared by:

Water Quality Planning Bureau, Water Quality Standards Section
Montana Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

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ACRONYMS

Acronym	Definition
BOD	Biochemical Oxygen Demand
CDBG	Community Development Block Grant
CEIC	Census and Economic Information Center
CFR	Code of Federal Regulations
DEQ	Department of Environmental Quality (Montana)
EPA	Environmental Protection Agency (US)
HUD	U.S. Housing and Urban Development
LMI	Low and Moderate Income rate
MFI	Montana's median family income
MHI	Median Household Income
MPS	Municipal Preliminary Screener
RO	Reverse Osmosis
SF	Summary File
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
WERF	Water Environment Research Foundation
WWTP	Wastewater Treatment Plant

EXECUTIVE SUMMARY

An analysis was undertaken to determine the degree and extent of economic impact that would occur in Montana as a result of publically owned wastewater treatment plants (WWTPs) having to comply to meet the base numeric nutrient standards. DEQ used technical data from engineers and published papers, U.S. census and demographic data, DEQ staff, EPA staff, and data from Montana WWTP operators to carry out the analysis. The analysis shows that affected communities across Montana would bear substantial and widespread economic impacts (i.e., economic hardship) if they had to meet base numeric nutrient standards today.

The treatment technology used to simulate costs to WWTPs consisted of advanced mechanical treatment combined with reverse osmosis. Treatment costs included those associated with nitrification/denitrification and biological phosphorus removal, high rate clarification, and denitrification Filtration. Costs were estimated from the DRAFT Interim WERF study "*Striking the Balance Between Wastewater Treatment Nutrient Removal and Sustainability, Considering Capital and Operating Costs, Energy, Air and Water Quality and More*" (Falk, et al., 2011a).

A sample of 24 affected WWTPs was used to estimate costs of having to meet Montana's base nutrient criteria. EPA's Economic Guidance (U.S. Environmental Protection, 1995) was used to determine whether affected WWTPs in Montana would be adversely affected economically by having to meet nutrient criteria. The three main tests from the guidance were used in this analysis and include the municipal preliminary screener, the Secondary score, and the Widespread test.

Out of the 24 town sample, 21 towns would experience a wastewater bill greater than 2% median household income in order to meet base nutrient criteria. When a sensitivity analysis is run, 23 out of 24 towns would experience a bill greater than 2% MHI. The one town that would not, Missoula, already meets nutrient criteria on the Clark Fork. After calculating the secondary scores for each of the 24 towns, all 24 would experience a 'Significant' impact using the "significance matrix" found in EPA guidance.

The widespread impact part of the test is open ended, and looks at the ripple effects from the significant impacts. A widespread impact is estimated to occur in almost all Montana town due to a more than doubling of the average wastewater bill (bills increase by 100% to 700% in the sample), a lower than average median household income for Montana, the current recession, and diminishing populations/narrow economies in most Montana towns. In additional, finding qualified WWTP operators for most Montana towns would be a challenge, as well as finding deep injection wells for the brine from reverse osmosis.

BACKGROUND

The Montana Department of Environmental Quality (DEQ) began developing numeric nutrient standards for state surface waters in 2001. A field pilot study was undertaken from 2001-2003 to identify and refine approaches for developing the criteria in the plains region of the state. Work from 2003-2008 focused on the selection of an appropriate zoning system by which the criteria would be applied, collection of data from reference streams to help with criteria derivation, and identification of harm-to-use thresholds for uses that nutrients affect. During this same period DEQ undertook a focused data collection to support the QUAL2K water-quality model which was then used to develop numeric nutrient criteria for a large river (lower Yellowstone). In addition, DEQ collected data to support lake nutrient standards (this work in ongoing, as are other field projects intended to further refine the flowing water criteria).

In 2008, DEQ released draft nutrient criteria for wadeable streams (Suplee, et al., 2008) and presented these to stakeholders. DEQ has subsequently refined the process by which wadeable stream criteria are derived, and is in the process of preparing those as of this writing; draft values are shown below (**Table 1**) along with draft criteria for the lower Yellowstone River. In **Table 1** and throughout this analysis, the N stands for nitrogen and the P for phosphorus. While stakeholders understand that the criteria were derived based on sound science and reflect values that are protective of the designated uses, the proposed criteria are stringent (**Table 1**). As a result, the stakeholder community has been concerned about what their permit limits will be as well as the opportunities for variances. Many WWTPs discharging into wadeable streams do not have instream dilution and would be required to meet the nutrient criteria end-of-pipe. For the lower Yellowstone River, the proposed criteria are above (i.e., have a higher concentration than) the ambient river concentrations during the seasonal low flow period. This situation means that WWTPs discharging directly to the Yellowstone may not need to meet the criteria at the end-of-pipe, although that has yet to be determined.

Table 1. Montana Draft Nutrient Criteria

Level III Ecoregion	Period When Criteria Apply	Parameter		
		Total P (mg/L)	Total N (mg/L)	Benthic Algae Criteria
Northern Rockies	July 1 -Sept. 30	0.025	0.3	120 mg Chl-*a*/m^2 (36 g AFDW/m^2)
Canadian Rockies	July 1 -Sept. 30	0.025	0.3	120 mg Chl-*a*/m^2 (36 g AFDW/m^2)
Middle Rockies	July 1 -Sept. 30	0.030	0.3	120 mg Chl-*a*/m^2 (36 g AFDW/m^2)
Idaho Batholith	July 1 -Sept. 30	0.030	0.3	120 mg Chl-*a*/m^2 (36 g AFDW/m^2)
Northwestern Glaciated Plains	June 16-Sept. 30	0.12	1.1	n/a
Northwestern Great Plains, Wyoming Basin	July 1 -Sept. 30	0.12	1.0	n/a
Yellowstone River (Bighorn R. confluence to Powder R. confluence)	Aug 1 -Oct 31	0.09	0.70	Nutrient concentrations based on limiting pH impacts
Yellowstone River (Powder R. confluence to stateline)	Aug 1 -Oct 31	0.14	1.0	Nutrient concentrations based on limiting nuisance algal growth

(Suplee, et al., 2008)

Due to the difficulty of currently meeting the draft nutrient criteria, Senate Bill 367 was signed by Governor Schweitzer on April 21, 2011.

SB 367 authorizes individual, general and alternative variances. Under the general variance limits established in SB 367, permit limits would be established at 1 mg/l TP and 10 mg/l TN for facilities discharging $\geq$ 1 MGD or 2 mg/l TP and 15 mg/l TN for facilities discharging $\leq$ 1 MGD. Lagoons would be capped at their current nutrient load.

The purpose of this paper was to quantify the costs of meeting the base numeric nutrients standards (**Table 1**) today, given the current state of treatment technology and the current economic status of the state. This paper demonstrates the substantial and widespread economic and social impact of nutrient criteria to the 107 affected public WWTPs in Montana. This document provides DEQ's demonstration supporting the statute language that all dischargers are, at the present time, exempt from meeting the base nutrient standards based on "Substantial and Widespread" economic impacts. Impacts to private dischargers will be demonstrated in a separate paper.

THE STUDY

MONTANA'S WWTPS

Out of the total number of WWTPs in Montana, which number about 200, 107 were identified as ones that would be affected by the nutrient criteria. WWTPs on Indian Reservations were not included as they are not regulated by the state (they have EPA permits). Also, a large number of WWTPs do not empty into a state surface water because either they land apply (spray irrigation), discharge to groundwater or landlocked lakes, are total containment systems, or are those for which these criteria would not apply (e.g., those that discharge to large rivers for which there is not yet a model/criteria). Thus, about half of Montana WWTPs would not have to meet these criteria, and most of these are smaller systems. The 107 WWTPs that would have to meet the criteria affect about 50% of Montana's population. The other 50% of Montana citizens are hooked up to one of the other 100 or so WWTPs not affected, or are on a septic system (generally more rurally based). These numbers are for residential hook-ups and do not include small and large businesses, schools or government.

Existing wastewater fees in affected Montana towns average about 0.9% of each town's median household income (MHI) across the state (based on a sample of 48 towns), with larger towns paying as little as 0.43% MHI and smaller towns paying up to 1.68% MHI (**Figure 1**). There is no clear correlation between town size and current wastewater fees, with the exception that the seven large towns over 19,000 in population are generally paying a lower MHI due to a larger population to spread out costs. Different towns pay different rates due to the age and effectiveness of the current system, past grant monies, current level of technology, size and quality of receiving stream, groundwater infiltration, and incoming wastewater quality. Most towns currently pay less than 1.5% MHI, with the majority of those paying less than 1.0% of MHI for wastewater treatment.

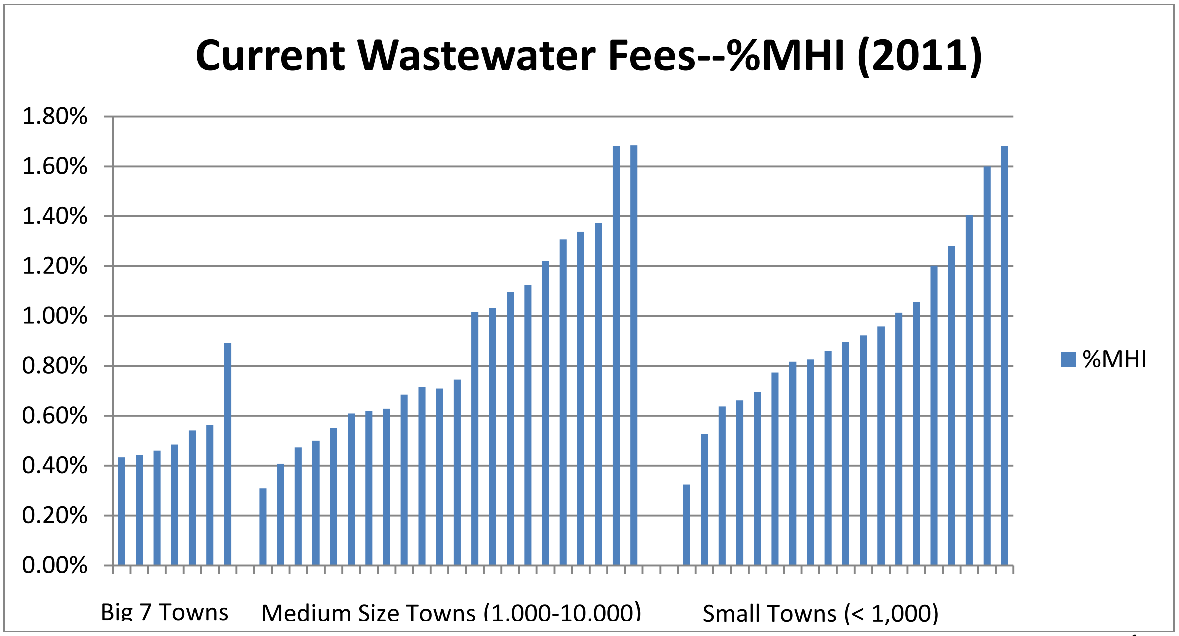

Figure 1. Current Annual Wastewater Costs as a Percentage of MHI in Montana Communities[1]

Summary of DEQ's Three-Step Process for Determining Substantial and Widespread Impacts

EPA regulations allow a variance from a water quality standard if the pollutant controls "…would result in substantial and widespread economic and social impact" (40 CFR 131.10(g)(6)). For public entities (e.g. WWTPs), EPA's 1995 guidance (U.S. Environmental Protection, 1995) suggests a three-step process to determine substantial economic impacts, and an additional analysis to determine widespread impacts. Although the guidance is typically used to evaluate individual WWTPs, DEQ followed the guidance in this demonstration to determine whether affected WWTPs in Montana as a whole would face economic hardship from base numeric nutrient criteria. This was done as a result of the impracticality of running an individual economic test on all 107 affected WWTPs.

Following the guidance, the first of two major "tests" in the Substantial determination (the first step) is to demonstrate that meeting the numeric nutrient criteria today would cost more than 2% of a community's Median Household Income (MHI) for most or all Montana communities with affected WWTPs. For this step, DEQ calculated the "Municipal Preliminary Screener (MPS)" value per the guidance for a subset of dischargers reviewed as part of DEQ's demonstration. The MPS is an estimate of the per household cost of proposed pollution controls—that is, meeting base nutrient criteria—plus existing wastewater fees as a percent of median household income for that town (%MHI). If the MPS value for these fees for an average household is equal to or greater than 2% MHI for a given town, then the Guidance suggests possible Substantial impacts and the discharger proceeds to the Secondary test, which is the second major "test" in the Substantial determination. The Guidance also allows a town with

[1] In **figure 1**, wastewater rates are expressed as a percentage of median household income as of 2011 and are stratified by town size. Communities for this rate comparison were initially selected via a stratified random process for three groups (small, medium, and large communities). More recently, 18 additional communities were added to this sample with a focus on larger and medium towns.

an MPS value of 1-2% to proceed on to the Secondary test, because the 1-2% range falls into an "uncertain effect" range.

For the Secondary test (step 2), DEQ evaluates a suite of five socioeconomic indicators for each affected town. Montana's Secondary test, as modified from the guidance, looks at the following economic metrics for a given town and compares the town level of each metric to the state average or to the average of a selected sample of towns. The socioeconomic indicators are:

- Poverty Rate
- Low and Moderate Income rate (LMI)
- Unemployment Rate
- Median Household Income (MHI)
- Current local tax and fee burden

LMI is an index number of the percentage of people in a town with an income below 200% of the poverty rate. Lower rates of poverty, LMI, and unemployment indicate a stronger economic situation in a given town. A high MHI does the same. A lower current local tax and fee burden also indicates a stronger economic situation, as more disposable income is generally available to households to be able to afford wastewater treatment improvements.

For each community, each of these five economic indicators are scored as either weak (a score of 1), average (a score of 2) or strong (a score of 3) compared to state averages or averages of a sample of selected Montana towns. The stronger the secondary score numerical rank is (the average score of the five economic metrics), the better able a town is to pay towards for meeting numeric nutrient criteria, and thus taking on a higher wastewater bill. The highest or strongest score a community could get would be a 3.0 (based on scoring a 3 score on all five categories—See **Appendix C**) and lowest would be a 1.0 (based on scoring a 1 score on all five socioeconomic categories). An average score of less than 1.5 for the five indicators is considered an overall weak Secondary score, 1.5 to 2.5 is considered mid-range, and over 2.5 is considered strong according to the Guidance. A weak Secondary score indicates a town with relatively weak economic health compared to the state average. A strong Secondary score indicated a town with a relatively strong economic health compared to the state average.

If a given town generally scored weak on the five indicators, say a 1.4 average value, this would be an indication that the town is already economically challenged and would be more significantly impacted by the higher wastewater rates, and thus more likely face a substantial impact. If it scored generally strong on the five indicators, say a 2.6 average value, this would indicate a town that is strong economically, and therefore the town might not be as significantly affected by additional wastewater fees and may not face a substantial impact (in which case it could better afford the new fees to meet the nutrient criteria). Although initially used in the Municipal Screener to determine if the 2% threshold was met, Median household income is applied differently in the context of the Secondary score and provides a general indicator of the health of the community.

The outcomes of both tests, the Screener and the Secondary test, are then assessed on a matrix (step 3) found in the guidance (**Figure 2**) to determine if water treatment costs to meet standards would cause 'Substantial' economic impact. If a town lands within a check mark or question mark within the matrix, then this constitutes a 'Significant' finding for that town with the affected WWTP. If a town lands on an 'x', then no Significant impact can be found, and the test is done. No variance from the numeric nutrient standards would be granted.

For example, a community with:

a. A mid-range (1.5-2.5) secondary test score and a high (> 2.0%) municipal preliminary screener score, would have substantial economic impact from meeting the new wastewater standards. The town would move on to the Widespread test.
b. A mid-range (1.5-2.5) secondary test score and a low (< 1.0%) municipal preliminary screener score, would not have substantial economic impact from meeting the new standards and no variance would be given.

		Municipal Preliminary Screener		
		> 2.0% (weak)	1.0% - 2.0% (mid-range)	< 1.0% (strong)
Secondary test score	< 1.5 (weak)	✓	✓	?
	1.5 – 2.5 (mid-range)	✓	?	✗
	> 2.5 (strong)	?	✗	✗

✓ = Substantial economic impact
? = Possible substantial economic impact
X = No substantial economic impact

Figure 2. Secondary Score Indicator Matrix from EPA Guidance

The third step in the economic hardship assessment, if a significant impact has been shown, is to demonstrate a 'Widespread' finding for all or almost all Montana communities with affected WWTPs. The guidance calls for a separate "widespread" demonstration that uses a variety of possible economic indicators, but with much more flexibility than the procedure for substantial impacts. The widespread demonstrations should assess the magnitudes of such indicators as increases in unemployment, losses to the local economy, changes in household income, decreases in tax revenues, indirect effects on other businesses, and increases in sewer fees for remaining private entities. While these widespread indicators are examples of things to look at, none are mandatory, and the analyst has discretion as to which to use. The Widespread analysis is discussed in more detail below.

Analysis Sample

Twenty-four publicly owned WWTPs were evaluated as a representative subset of the larger population of 107 affected Montana dischargers. The public dischargers selected for the analysis represented larger communities who are major dischargers with advance treatment systems (> 1MGD), large, medium and small towns who are minor dischargers with advanced treatment systems (< 1 MGD), and lagoon systems. Site-specific information on the existing treatment technologies, facility-specific effluent data and community demographics were obtained for this subset and extrapolated to publicly owned plants throughout the state with similar wastewater treatment trains and similar demographics.

Within Montana, the size and types of public wastewater treatment plants vary significantly, ranging from lagoon systems to systems using advanced biological nutrient removal. **Table 2** summarizes the number of major, minor and lagoon public dischargers in the State that would be affected by nutrient criteria, and then breaks down that same distribution within the selected sample. It is clear from the table that the major dischargers were completely represented within the 24 towns selected for analysis, while the lagoons were represented by a small subset of the lagoon total. This was done because it is assumed that all small towns with lagoons would experience significant and widespread impacts from having to meet criteria, while it was unclear whether that would be true for all major and minor dischargers. Therefore, the subsample included towns most likely to not experience economic hardship from having to meet standards, and thus be able to afford to reach base nutrient criteria. This was done to err on the side of being conservative in attaining a hardship finding for the state as a whole.

Table 2. Municipal WWTPs in Montana Affected by Nutrient Criteria

	Major Discharger (Big 7 Towns)	Advanced Discharger > 1 MGD	Advanced Discharger < 1 MGD	Lagoons
All affected Montana Dischargers	7	5	12	83
Percent of total affected WWTPs	6.5%	4.7%	11.2%	77.6%
Subsample	7	5	4	8

To address the first step in the Substantial test, the Municipal Preliminary Screener, DEQ developed a detailed Excel spreadsheet (**Appendix A**) to calculate the annualized capital and operations and maintenance costs (O&M) associated with meeting the base numeric nutrient standards for the 24 sample towns. The spreadsheet also estimated the percent of MHI associated with the increased sewer rates plus current sewer rates. For purposed of this analysis, reverse osmosis was assumed to be technology needed to attain the criteria. Capital and O&M costs for attaining nutrient standards were estimated from the DRAFT Interim WERF study (Falk, et al., 2011a). **Appendix A** presents two spreadsheets with the calculations and results of the analysis. **Appendix B** documents all the underlying assumptions applied for this demonstration.

The interim WERF study looked at five different levels of nutrient treatment from minimal treatment (level 1) to a treatment that is close to Montana's base criteria (level 5). In fact, level 5 would meet or be superior to some of Montana's criteria shown in **Table 1**. Level 1 treatment in the study is more advanced than lagoons, but still does not directly treat N and P. Level 2 treatment is about the same as the variance levels outlined in SB 367. **Table 3** summarizes the attainable effluent quality and costs of the five different treatment levels from the interim WERF study. **Table 4** summarizes the water treatment processes used in the study for each of those five levels.

Table 3. Effluent Quality and Associated Treatment Costs in the Interim WERF study (Falk, et al., 2011a)

Level	Description	Capital Cost (million dollars per 1 GPD design flow)	Operations Cost (dollars per day per 1 MGD actual flow)
Level 1	No N and P removal	9.3	250
Level 2	1 mg/l TP; 8 mg/l TN	12.7	350
Level 3	0.1-0.3 mg/l TP; 4-8 mg/l TN	14.4	640
Level 4	<0.1 mg/l TP; 3 mg/l TN	15.3	880
Level 5	<0.01 mg/l TP; 1 mg/l TN	21.8	1370

Table 4. Unit Processes per Treatment Level in WERF Study (Falk, et al., 2011a)

Level	Liquid Treatment	Solids Treatment	Comment
1	Primary Clarifier Activated Sludge Disinfection Dechlorination	Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Conventional Activated Sludge for BOD/TSS removal
2	Primary Clarifier Activated Sludge Alum (optional) Disinfection Dechlorination	Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Nitrification/Denitrification and Biological Phosphorus Removal
3	Primary Clarifier Activated Sludge Methanol (optional) Alum (filtration) Filtration Disinfection Dechlorination	Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Nitrification/Denitrification and Biological Phosphorus Removal and Filtration
4	Primary Clarifier Activated Sludge Methanol (optional) Alum/Polymer (Enhanced Settling) Enhanced Settling Filtration Disinfection Dechlorination	Fermentation Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Nitrification/Denitrification and Biological Phosphorus Removal, High Rate Clarification and Denitrification Filtration
5	Primary Clarifier Activated Sludge Methanol (optional) Alum/Polymer (Enhanced Settling) Enhanced Settling Filtration Microfiltration Reverse Osmosis Disinfection Dechlorination	Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Nitrification/Denitrification and Biological Phosphorus Removal, High Rate Clarification, Denitrification Filtration, and MF/RO on about Half the Flow

Costs for the S&W demonstration were estimated based on the assumption that reverse osmosis (RO) would be the technology used to best meet base nutrient criteria.[2] Current nutrient levels and treatment costs at the 24 sample towns were compared to nutrient levels and costs that would be needed to meet RO based on the WERF study. In this way, annual capital and operations costs needed for meeting base nutrient criteria were applied to each town, and new wastewater bills were calculated for a scenario where towns would have to meet RO and thus attempt to meet base nutrient criteria today. Towns that have lagoons were assumed to have to pay the entire listed costs (per MGD) of Level 5 to get to the criteria (use RO). Towns currently with advanced treatment were assumed to have already paid for some of the Level 5 costs. If a town already met WERF level 2 nutrient levels, for example, then the level 2 costs for both capital and operations were subtracted from level 5 costs. It is important to note that the operations costs of meeting base numeric criteria taken from the WERF study (**Table 3**) do not include labor and maintenance costs, so the costs estimates may be slightly low (conservative). This is addressed below. WERF level 5 is not quite as stringent as many of the Montana base nutrient criteria, so the costs to reach nutrient standards estimated for this demonstration are potentially underestimated in that sense as well, which is also addressed below.

RESULTS

SUBSTANTIAL IMPACT

Table 5 presents the Municipal Preliminary Screener results for the 24 communities evaluated in the analysis if they had to meet base numeric nutrient criteria. DEQ first examined the MHI results that would be incurred by the largest seven Montana towns (Billings, Great Falls, Missoula, Bozeman, Butte, Helena, and Kalispell). Missoula was assumed to already meet the criteria on the Clark Fork due to dilution (the only affected town to do so out of the 107), but was included anyway. The rationale for this approach was that if any WWTP could afford meeting numeric nutrient criteria, it would be Montana's largest towns due to the already-sophisticated systems in place and/or large populations across which additional costs could be dispersed (i.e., economies of scale). Differences in the resulting MHI levels for these seven towns (and all Montana towns) include current levels of nutrient treatment, town population, current MHI, and current wastewater fees. Based on our analysis, five out of seven of the largest towns in Montana would score over the 2% MHI threshold to meet base criteria (**Table 2**). Missoula (which already meets the standard) and Helena do not. Lolo also comes in under 2%. The three towns in the sample that would not hit the 2% threshold are highlighted in blue. All smaller towns with lagoons scored more than 2% MHI. The breakout of all 24 towns is given below.

[2] A 'Pilot Study for Low Level Phosphorus Removal' ([2010] Hal Schmidt, P.E.MWH Americas, Inc.), conducted in Florida shows that for TP, TN, and other micro-pollutants, RO was indeed the most effective method for removing TN and TP (better than membrane bioreactor, MBR). Dave Clark of HDR Engineering, agreed that RO is the treatment that results in the lowest TN levels, and that the WERF report accurately reflects capital and operations costs for RO. Thus, this study assumes the use of RO technology for this demonstration of economic hardship. (It is important to note that this does not mean that Montana WWTPs would be expected to implement RO to meet practical Limits of Technology [LOT] or nutrient criteria in practice.)

Table 5. % MHI Results for towns to reach Base Criteria

Community	Expected % MHI	Population	MGD (Design Flow)
The Big Seven Montana Towns			
Kalispell	2.58%	$39,953	5.4
Bozeman	2.92%	$41,661	13.8
Helena	1.74%	$47,152	5.4
Butte	2.15%	$37,335	8.5
Billings	2.41%	$45,004	26
Missoula	1.47%	$34,319	12
Great Falls	4.18%	$40,718	26
Other Large Montana Facilities > 1 MGD			
Livingston	6.85%	$35,689	5
Miles City	4.09%	$37,554	3.7
Hamilton	5.44%	$25,161	1.98
Lewistown	3.43%	$31,729	2.5
Havre	2.04%	$43,577	4.4
Non Lagoon Facilities < 1 MGD			
Columbia Falls	3.02%	$38,750	0.766
Manhattan	2.60%	$50,729	0.6
Lolo	1.81%	$46442	0.34
Stephensville	3.17%	33776	0.3
Lagoons			
Philipsburg	4.19%	$31,375	0.2
Cut Bank	2.68%	$44,833	0.643
Deer Lodge	3.89%	$40,320	3.3
Glendive	3.67%	$42,821	1.3
Red Lodge	5.16%	$50,123	1.2
Big Fork	2.65%	$44,398	0.5
Highwood	2.54%	$62,614	0.026
Circle	5.47%	$29,000	0.16

From the analysis is it clear that small towns in Montana, which comprise the vast majority of affected WWTPs in Montana (78%), would all exceed the 2% MHI threshold (Municipal Preliminary Screener). It is also important to note that the costs to reach WERF Level 5 underestimate the cost to reach nutrient criteria. **Figure 3** shows a plot of the 24 town sample comparing population to %MHI. The vertical red line shows the 2% MHI cost level. The main trend that stands out is that the largest towns (the seven points at or above the 20,000 population mark) would pay between 1.8% and 4% MHI to meet the nutrient criteria while all other towns in the sample cover a wider range of between 1.8% and almost 7%. Also, smaller towns in the sample scored a higher average MHI percent overall than the largest seven towns. This strongly suggests that smaller towns would all bear higher than a 2% MHI to reach base numeric criteria. **Figure 4** shows the estimated percentage increases in wastewater bills from having to meet criteria. (Note: Including town names in the figures was visually too crowded).

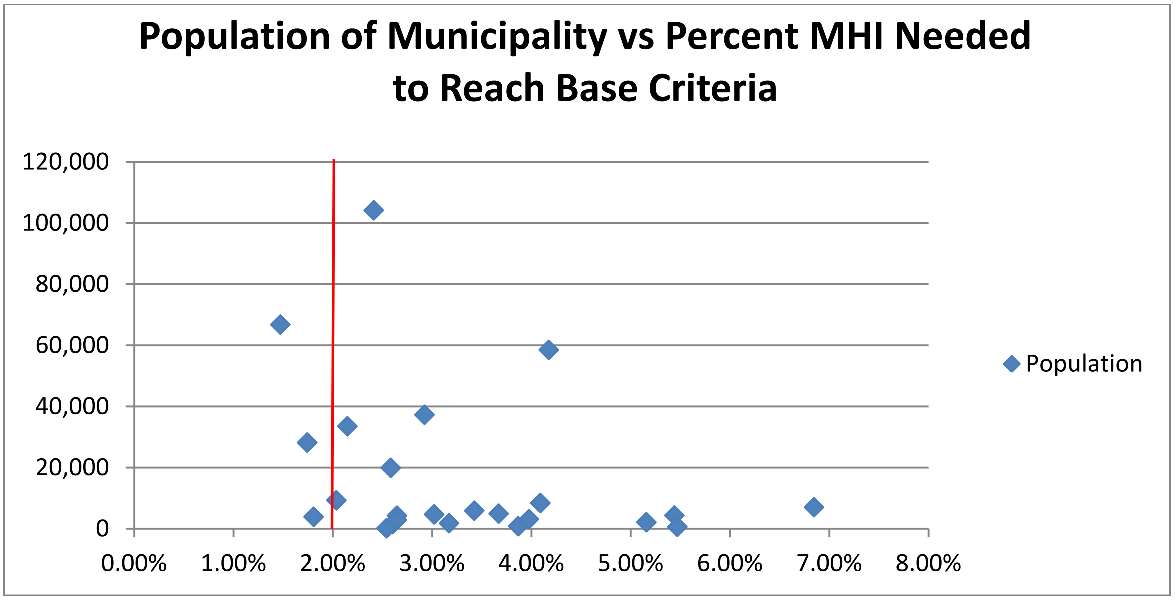

Figure 3. Population Versus Percent MHI Needed to Reach Base Nutrient Criteria

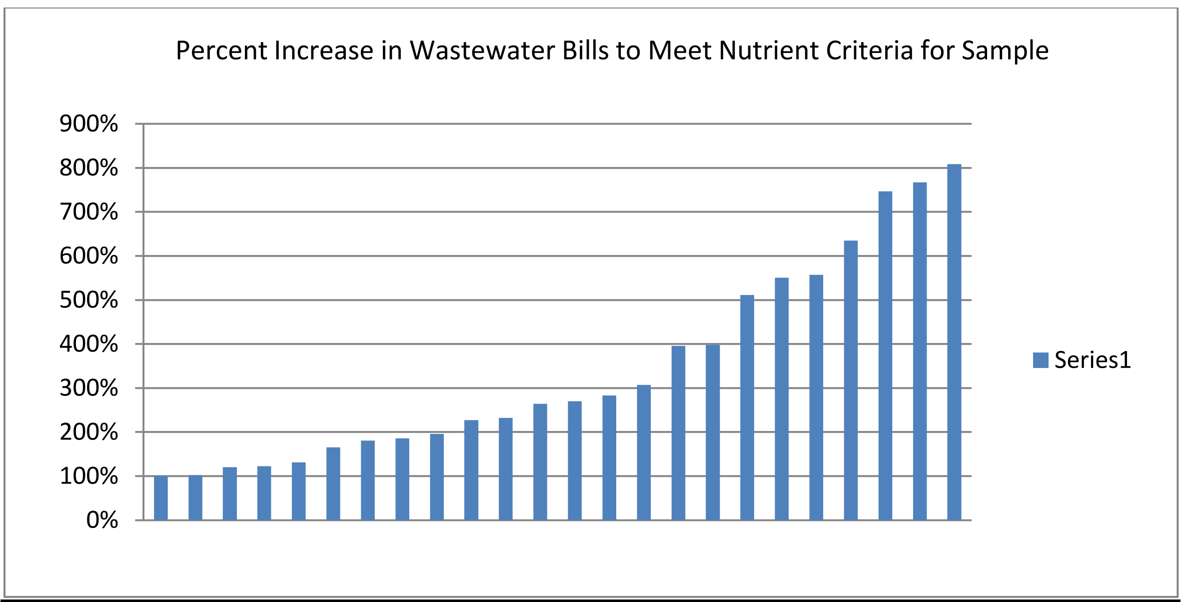

Figure 4. Percent Increase in Wastewater Bills to Meet Nutrient Criteria

SENSITIVITY ANALYSIS OF MUNICIPAL PRELIMINARY SCREENER

The demonstration so far has presented the results of expected treatment costs—the percentage MHI—as a single value. Because of the uncertainty associated with the underlying assumptions, we provide a range of values based on alternate, reasonable assumptions. Three 'alternate' assumptions are given, and those assumptions are combined in various ways to calculated alternate MHI values for each of the 24 towns and thus provide ranges for MHI.

Alternate Assumption #1: Discount Rate

DEQ assumed an alternative discount rate of seven percent for capital expenditures on new wastewater treatment equipment compared to the 5 percent modeled in DEQ's original analysis. In many cases, five percent interest is an appropriate discount rate to annualize the capital costs at the national level, but may not be appropriate for bonds that would be issued by smaller communities. Additionally, there exists some uncertainty on the rate depending on the general economic conditions at the time the bonds are issued and the debt capacity and rating of the borrower.

Alternate Assumption #2: Labor Costs

DEQ assumed the inclusion of labor costs of 15 and 48 percent of capital costs. The original DEQ analysis did not include labor costs, which can be a significant cost for a treatment process. The reason for this is those costs were not included in the WERF study. An analysis of the life-cycle costs for a number of technologies used to control nitrogen and phosphorus in wastewater treatment plants estimated that labor costs are between 15-21 percent of the annualized capital costs for nitrogen and 15-48 percent of annualized capital costs for phosphorus.[3] A range of 15% to 48% is used to add on to total costs.

Alternate Assumption #3: Reverse Osmosis

The WERF study, which was the basis for the costs in this study, included RO treatment for 50 percent of the flow after treatment Level 4. The treatment levels 1 through 4 represented progressively greater levels of treatment for each successive level. This was represented by the inclusion of additional unit processes (e.g., level 4 is the same as level 3 with some added processes to achieve more reduction of nutrients). Level 5 did not exactly follow this progression, since half of the flow remained treated by processes equivalent to Level 4 and the other half received an enhanced level of treatment (reverse osmosis or RO).

To meet the MT criteria, which are more stringent for TN than WERF level 5, one could assume that the highest level of treatment was needed for 100 percent of the flow--not half as specified in the cost analysis in the WERF study. Thus, cost estimates could be based on providing RO treatment to 100 percent of flow rather than 50% of flow, in order for WWTPs to achieve the Montana nutrient criteria. While it may be possible that some facilities' waste streams and effluent levels would not require 100 percent RO treatment, simulating at 50 and 100 percent provides an upper bounds estimate of the potential economic impact of the Montana nutrient criteria.
The WERF data were adapted to estimate the cost of treating all flow by RO by isolating the marginal unit processes used for Level 4 and Level 5 and calculating the cost for a treatment train with 100 percent RO.

SCENARIOS

For this analysis, multiple estimated treatment costs as a percentage of MHI values were calculated based on five additional scenarios to the original DEQ scenario (see **Table 6**). As explained below, the discount rate was varied from 5 to 7 percent and the addition of both high (48 percent) and low (15 percent) labor costs as a percentage of capital costs were considered across each scenario. Then, the

[3] POINT SOURCE STRATEGIES FOR NUTRIENT REDUCTION. TMDL Workshop. February 17, 2011. S. Joh Kang, Ph.D., P.E. and K. Olmstead, Ph.D., P.E. Tetra Tech Inc. Ann Arbor, MI. (Based on information in: Introduction of Nutrient Removal technologies Manual, EPA, 2008 and WEF/WERF Cooperative Study of Nutrient Removal Plants: Achievable Technology Performance Statistics for Low Effluent Limits)

100% RO is added on to the original estimates separately to isolate how that assumption alone would affect costs.

Table 6. Scenarios for Sensitivity Analysis

Scenario	Description	Discount Rate	Labor Cost
Original	5% discount rate and 0% labor cost	5%	0%
Scenario A	Change of labor cost to 48% of capital cost	5%	48%
Scenario B	Change of labor cost to 15% of capital cost	5%	15%
Scenario C	Discount rate increase from 5% - 7%	7%	0%
Scenario D	Discount rate increase from 5% - 7% AND change of labor cost to 48% of capital cost	7%	48%
Scenario E	Discount rate increase from 5% - 7% AND change of labor cost to 15% of capital cost	7%	15%

Results of Sensitivity Analysis

Figures 5 and **6** below present the results from Scenarios A-E. **Figure 5** shows the original MTDEQ analysis and the 5 scenarios percent MHI values for all communities. **Figure 6** is a condensed presentation of the results that displays the percent MHI results for the original scenario, the average of all scenarios, and minimum, median, and maximum values (indicated by the gray boxes on the figure), and the original MHI with 100% of treated water going through Reverse Osmosis.

It is clear that all of the communities included except for Missoula would be above the 2 percent MHI threshold under all alternate scenarios. As mentioned before, Missoula already appears to be meeting nutrient criteria. The analysis demonstrates that the two POTWs that were not above the 2 percent threshold in the original MTDEQ analysis (Havre, Helena), would most likely be above the threshold when uncertainty in the data and additional factors are taken into account.

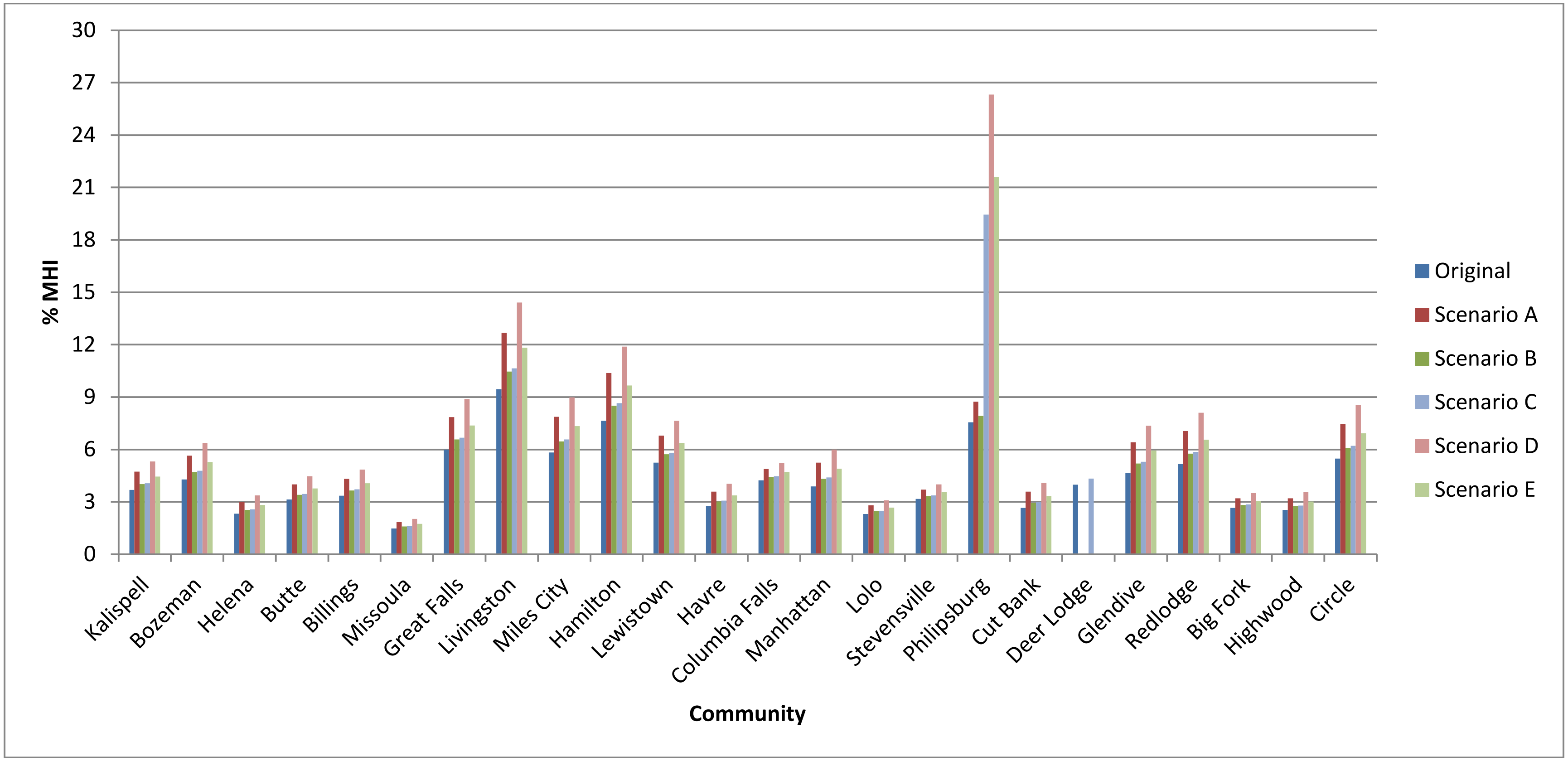

Figure 5. Expected % MHI to Meet Base Numeric Nutrient Criteria (plus current wastewater fees)

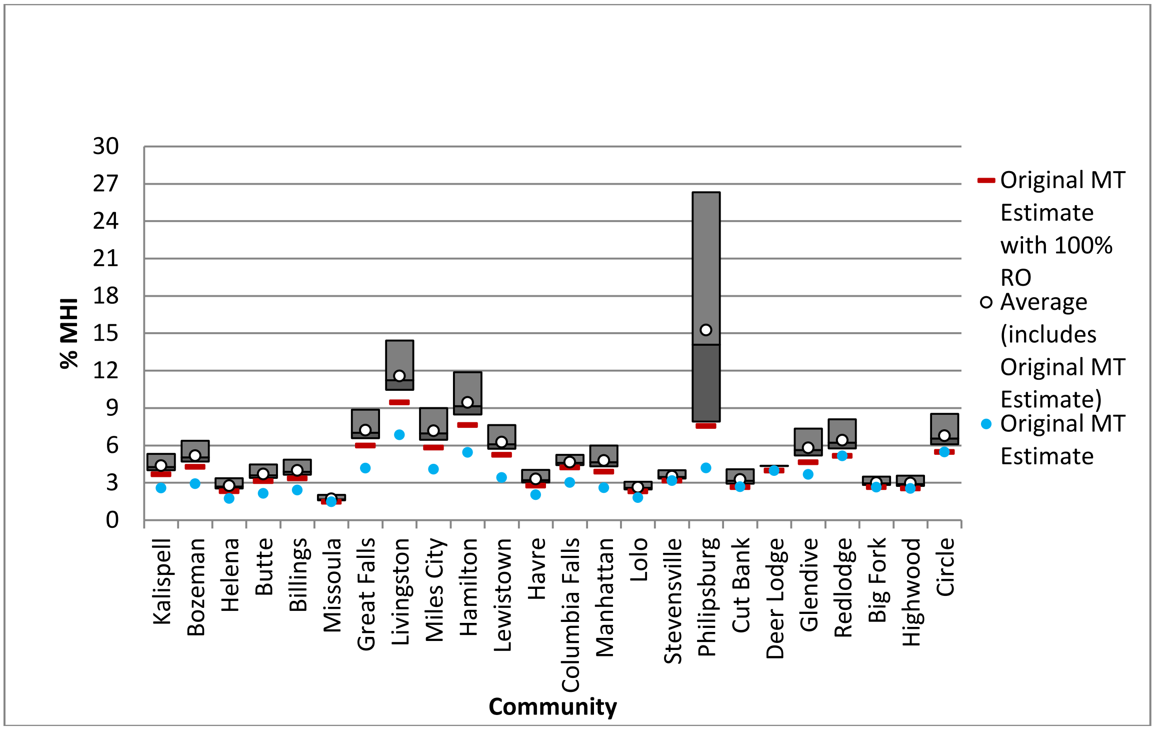

Figure 6. Expected % MHI to Meet Base Numeric Nutrient Criteria (plus current wastewater fees) - Condensed Presentation

CALCULATION OF THE SECONDARY SCORE

The second step in demonstrating Substantial effects from meeting nutrient criteria involves evaluating a community's current economic health. This is referred to in the guidance as the Secondary Score (**Table 7** and **Figure 7A**). DEQ calculated the secondary score values for the 24 sample communities (listed in **Table 5**) by obtaining data from the following sources. **Appendix C** provides the secondary scores for each community, along with the total secondary score value and the five socioeconomic indicators.

Out of the sample of 24, no town comes in below 1% MHI to meet nutrient criteria thereby eliminating two of the three 'x' squares in the matrix. No town with a strong secondary test score comes in under 2% MHI for meeting nutrient criteria eliminating the third x. Thus no towns fall in a square with an x. This means that all 24 towns would experience a Substantial or Possible Substantial impact from having to meet nutrient criteria. In fact, most towns fall within the square that is the check mark in the middle left square. **Figure 7B** shows the matrix and the number of towns out of 24 that fall within each corresponding square of the matrix.

Table 7. Data Sources for the Secondary Score Indicators

Secondary Score Indicator	Data Source	Notes and Web link
Poverty Rate	Montana Census Data, Montana Census and Economic Information Center (MT CEIC); 2009 American Community Survey Data and Social Explorer website	http://ceic.mt.gov/Demographics.asp http://www.census.gov/prod/2010pubs/acsbr09-1.pdf http://www.socialexplorer.com
Low and Moderate Income rate (LMI)	2005-2009 American Community Survey 5-Year Estimates	LMI was calculated by DEQ by taking the number of persons who live below 200% of the poverty level threshold for a town, and dividing by the total number of persons in a town
Unemploy ment Rate	Source: Montana Department of Labor and Industry Research and Analysis Bureau, Aaron McNay.	http://www.ourfactsyourfuture.org/ Montana: http://www.ourfactsyourfuture.org/cgi/databrowsing/?PAGEID=4&SUBID=123
Median Household Income	Montana Census Data (MT CEIC), U.S. Census Bureau, American Community Survey 5-Year Estimate (2005-2009); Small Area Income and Poverty Estimates	http://www.census.gov/hhes/www/saipe/index.html
Current local tax and fee burden	Annual Financial Reports of the Cities and Towns of Montana, sheet entitled "Government-wide Statement of Activity", Local Government Services Bureau, Dept of Administration, State of Montana, Kim Smith, (406) 841-2905.	DEQ calculated an index based on current local taxes and fees plus local property taxes, indexed by population and MHI to normalize towns. A histogram of all towns (using the normal distribution) in the “tax index sample’ (39 towns total) created a weak, medium and strong score for each town compared to the sample average

		Municipal Preliminary Screener		
		> 2.0% (weak)	1.0% - 2.0% (mid-range)	< 1.0% (strong)
Secondary test score	< 1.5 (weak)	✓	✓	?
	1.5 – 2.5 (mid-range)	✓	?	✗
	> 2.5 (strong)	?	✗	✗

✓ = Substantial economic impact
? = Possible substantial economic impact
X = No substantial economic impact

Figure 7A. Secondary Score Indicator Matrix.

	2.0% (weak)	**1.0%-2.0% (mid-range)**	**<1.0% (strong)**
< 1.5 (weak)	1	0	0
1.5-2.5 (mid-range)	**18**	3	0
>2.5 (strong)	2	0	0

Figure 7B. Where the 24 Sampled Towns Fell within the Matrix.

Secondary score values for the 24 Montana towns sampled ranged between 1.2 and 3.0 (**Table 8**). Larger towns (i.e, Billings, Bozeman, Helena, Great Falls, Missoula) had secondary scores between 1.8 and 2.4 thus falling in the mid-range. Combined with the MPS results, 24 out of 24 of the sample communities were considered to be "substantially" affected by requirements to meet the numeric nutrient criteria. Again, towns falling into a matrix square with a question mark are considered to have a borderline substantial impact. For more info on the Secondary scores for the 24 towns, see **Appendix C**.

Table 8. Secondary Scores for sample MT communities

Community	Secondary Score	MHI %
Kalispell	1.8	2.58%
Bozeman	2.0	2.92%
Helena	2.4	1.74%
Butte	2.0	2.15%
Billings	2.2	2.41%
Missoula	1.8	1.47%
Great Falls	2.0	4.18%
Livingston	1.6	6.85%
Miles City	2.0	4.09%
Hamilton	1.2	5.44%
Lewistown	2.0	3.43%
Havre	2.0	2.04%
Columbia Falls	1.8	3.02%
Manhattan	2.2	2.60%
Lolo	2.0	1.81%
Stephensville	1.6	3.17%
Philipsburg	1.6	3.87%
Cut Bank	1.6	2.65%
Deer Lodge	2.0	3.98%
Glendive	2.2	3.67%
Red Lodge	2.2	5.16%
Big Fork	2.25	2.65%
Highwood	3.0	2.54%
Circle	2.0	5.47%

As demonstrated above, no towns in Montana would score a strong Secondary score and less than 2% MHI (both of which would need to happen for a finding of non-Significant impact). Indeed, only three towns scored less than 2% MHI, and none of those has a strong secondary score. This is likely to be the case for all of Montana, as almost every town will score greater than 2% MHI and thus gain a significant finding per the matrix in the guidance. Thus, because it is estimated that step one and step two are met for 100% of affected Montana towns, a substantial impact has been demonstrated. We have shown this to be the case for virtually every town in Montana.

WIDESPREAD ANALYSIS

The third major metric in the S&W demonstration is the widespread test. The guidance does not provide direct ratios or specific tests for a Widespread finding, nor does it provide a straightforward method of proving Widespread impacts (as it does for a Substantial finding). In addition, it suggests looking at some of the economic metrics that are used in the two Substantial tests. From the guidance:

> "The financial impacts of undertaking pollution controls could potentially cause far-reaching and serious socioeconomic impacts. If the financial tests outlined in Chapter 2 and 3 suggest that a discharger (public or private) or group of dischargers will have difficulty paying for pollution controls, then an additional analysis must be performed to demonstrate that there will be widespread adverse impacts on the community or surrounding area. There are no economic ratios per se that evaluate socioeconomic impacts. Instead, the relative magnitudes of indicators such as increases in unemployment, losses to the local economy, changes in household income, decreases in tax revenues, indirect effects on other businesses, and increases in sewer fees for remaining private entities should be taken into account when deciding whether impacts could be considered widespread. Since EPA does not have standardized tests and benchmarks with which to measure these impacts, the following guidance is provided as an example of the types of information that should be considered when reviewing impacts on the surrounding community." (Chapter 4, first paragraph, found at http://water.epa.gov/scitech/swguidance/standards/economics/chaptr4.cfm)

DEQ considered the widespread analysis based on the following basic question: For Montana towns, which would all be Substantially affected by having to meet base numeric nutrient criteria, what are the economic and social ripple effects of that substantial impact on the local area? An important step in this question was to define the geographic area where project costs pass through to the local economy. For Montana's widespread analysis, DEQ established the entire state as the "geographic area" considered in the widespread demonstration.

The Widespread argument was made for all towns together rather than individual towns, due to the impracticality of showing widespread impact for each of the 24 towns in the sample, much less all 107 affected towns. Widespread Impacts were evaluated by their cumulative effect and by the DEQ analyst's Best Professional Judgment. Most towns are small and rural or small and a suburb of a larger town. Statewide, there are approximately 95 small towns (under 5,000 in population) out of the affected 107. The other 13 affected towns are "medium to large" and are more urban-based with more diverse economies. Six of these thirteen towns have more than 20,000 in population and a seventh town (Kalispell) is at an estimated 19,927 persons (Montana CEIC, American Community Survey). The other six are between 5,000 and 10,000 in population (see **Table 9**).

Table 9. Population Distribution of all 107 Affected Towns

	Large Towns (20,000 persons and over)	Medium Towns (between 5,000 and 10,000 persons)	Small Towns (under 5,000 persons)
Number	7	6	94
Percentage of Total affect towns	6.5%	5.6%	87.9%

Percentage of Montana households that would be affected by Nutrient Criteria – 50% (approximately)

DEQ believes that at least 95% of the 107 affected Montana towns (104 out of 107) would experience widespread impacts by having to meet base numeric nutrient standards today. DEQ's Widespread argument is as follows.

- The fact that almost every town in Montana (estimated 104 out of 107) would experience a cost of 2% or greater MHI from having to meet numeric nutrient criteria suggests widespread impacts across the state. Of the 24 communities examined, 21 showed a 2% MHI or greater, and almost certainly the other 86 towns of the 107 towns would as well (smaller and most with lagoons). With alternate assumptions, 23 out of 24 showed a 2% or higher MHI. The aggregated effects of the 2% MHI or greater on such a large number of individual communities would likely result in widespread effects at the statewide scale.
- Most small towns (< 5,000) have agricultural-based economies and use lagoons for wastewater treatment. The cost of achieving standards relative to MHI will be much higher than 2% for many of these small towns considering that most have lagoons that would need complete, major upgrades (including abandonment of the lagoon) and most have small populations over which to spread that cost. Many of these towns are currently losing population and business, especially in the eastern portion of the state. In addition, these small towns already currently have higher sewer rates within the state (on average) than the largest seven towns.
- Montana is currently 41st in the nation in per capita income as of 2009 at $22,881 (U.S. Department of Commerce, 2012). Prices in Montana are about average for the U.S. across all goods. Montanans on average do not have as much disposable income as the average American, and may have slightly higher living expenses due to long travel distances and higher heating bills.
- All affected towns but one in Montana (the one that already meets criteria) would pay at least 2% MHI in their total wastewater bill to meet base numeric nutrient standards, or significantly more than they are currently paying on average (current bills average about 0.9% across Montana). Thus, wastewater bills would at least double on average for affected communities to meet the numeric nutrient criteria. In a state with less disposable income than the U.S. average, a greater than 1% decrease in disposable income on average due to higher bills will produce widespread effects on households and businesses (some businesses more than others). A substantial increase in the wastewater bill could tip the scales for a percentage of residences based on decreased disposable income as a result of the increase in the wastewater bill. Residences below the MHI for a town could be hit especially hard.
- Town residents are used to small increases in utility bills. Having to meet nutrient criteria would cause a very large increase in most utility bills, and likely public outcry. As an example, a doubling of electric rates for members of the SME electric utility has resulted in a high-profile public battle.
- Since most small towns do not have diverse economies, even a small decrease in business and in population can have a large effect on small towns that are struggling. For example, some small Montana towns have less than 10 businesses total. Future businesses and homes could self-locate out of town to avoid high wastewater fees, although that is speculative.

- It is assumed that all towns under 5,000 persons would experience Widespread impacts.
- Towns with populations over 5,000 will likely show mixed results in terms of Widespread impact. The six large towns affected by nutrient criteria would experience Widespread impacts in terms of disposable income, but possibly not overall (e.g. would not see their economy collapse). In other words, these large towns would not shut down, but certain residences and businesses would experience substantial impacts. Another 12 or so medium to large towns would probably experience Widespread impacts overall for the same reasons as discussed above, but less severe impacts than the 95 smaller towns with affected WWTPs.
- The current Recession could complicate these effects. Even if one-third of these medium to large towns did not experience Widespread impacts per the guidance (4 total), more than 95% of Montana's affected towns still would meet the 'almost all' threshold for Widespread impacts, while all meet the criteria for Significant impacts.
- To meet the base numeric nutrient criteria will require hiring highly qualified wastewater engineers in each affected town. There could be widespread impacts associated with finding these qualified staff for facilities across the state and then paying them a competitive salary. Such operators may be hard to find for small Montana towns.
- The 2010 census data showed that Montana's population is aging. This trend, coupled with increased living expenses associated with meeting the base nutrient standards, could have negative impacts on a statewide scale.
- Small towns in Montana are struggling in certain cases to get basic infrastructure like broadband internet. A large jump in wastewater infrastructure costs could halt that progress.
- DEQ's substantial and widespread analysis assumed that reverse osmosis or some ion exchange treatment technology would be required. Either technology is both economically and environmentally costly. Reverse osmosis generates brine that must be disposed of properly and results in significantly higher greenhouse gas emissions. Aggregated at the statewide scale, both the economic and environmental implications of meeting Montana's criteria would have widespread impacts for the State of Montana.
- Benefits from meeting base numeric standards would likely not be widespread in terms of economics. Jobs created would be greatest in the short term for construction, and long-term jobs would tend to be small in relation to an area's entire work force, except for the smallest of towns. Environmental benefits would be widespread.

Conclusions

This demonstration shows that meeting the numeric nutrient criteria on a statewide basis would result in Substantial and Widespread economic impacts to Montanans (for public sector). Of the 24 publicly-owned dischargers reviewed in this analysis, 100% of them demonstrated Substantial impacts and at least 20 would likely demonstrate Widespread Economic impacts. DEQ believes that if 95% of the communities demonstrate Substantial and Widespread impacts, which this paper has done, then DEQ has shown economic hardship at the statewide scale.

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APPENDIX A - SPREADSHEETS OF COSTS AND MHI

Table A-1. Summary Demographic Data for the Sample Towns Including Current Wastewater Free

Community	Median Household Income (2010) - countywide MHI. Recommend updating for service area.	Population	Estimated Number of Households (Population / 2.5) based on 2000 Census	Current Average Annual Household Wastewater Bill	Design Flow (MGD)	Actual Flow (MGD)	Current wastewater MHI	Percent MHI needed to get to RO/Base Numeric Nutrient Criteria (including current fees)
Kalispell	$39,953.00	19,927	7,705	$216.00	5.4	3.10	0.54%	2.58%
Bozeman	$41,661.00	37,280	14,614	$372.00	13.8	5.80	0.89%	2.92%
Helena	$47,152.00	28,190	12,337	$265.44	5.4	3.00	0.56%	1.74%
Butte	$37,335.00	33,525	14,041	$360.00	8.5	4.00	0.96%	2.15%
Billings	$45,004.00	104,170	41,841	$218.28	26	26	0.49%	2.41%
Missoula	$34,319.00	66,788	27,553	$152.14	12	9	0.44%	1.47%
Great Falls	$40,718.00	58,505	23,998	$187.20	26	26	0.46%	4.18%
Livingston	$35,689.00	7,044	3,188	$600.00	5	2	1.68%	6.85%
Miles City	$37,554.00	8,410	3,518	$236.10	3.7	2	0.63%	4.09%
Hamilton	$25,161.00	4,348	2,092	$276.00	1.98	0.68	1.10%	5.44%
Lewistown	$31,729.00	5,901	2,727	$387.60	2.5	1.5	1.22%	3.43%
Havre	$43,577.00	9,310	3,709	$240.00	1.8	1	0.55%	2.04%
Columbia Falls	$38,750.00	4,688	1,621	$532.20	0.766	0.37	1.37%	3.02%
Manhattan	$50,729.00	1,520	523	$362.40	0.6	0.4	0.71%	2.60%
Lolo	$46,442.00	3,892	1,060	$363.00	0.34	0.38	0.78%	1.81%
Stevensville	$33,776.00	1,809	795	$535.08	0.3	0.29	1.58%	3.17%
Philipsburg	$31,375.00	820	399	$200.00	0.2	0.2	0.64%	4.19%
Cut Bank	$44,833.00	2,869	1,290	$138.48	0.643	0.643	0.31%	2.68%
Deer Lodge	$40,320.00	3,111	1,522	$409.56	3.3		1.02%	3.89%
Glendive	$42,821.00	4935	1,883	$213.96	1.3	N/A	0.50%	3.67%
Redlodge	$50,123.00	2125	1,055	$305.28	1.2	0.65	0.61%	5.16%
Big Fork	$44,398.00	4270	1,708	$580.36	0.5		1.31%	2.65%
Highwood	$62,614.00	176	53	$600.00	0.026	0.015	0.96%	2.54%
Circle	$29,000.00	615	234	$259.56	0.16	0.065	0.90%	5.47%

Table A-2. Detailed Costs for the Sample Towns of Meeting Criteria (next three pages)

Community	Current Treatment Technology	Design Flow (MGD)	Actual Flow (MGD)	Capital cost (million dollars) to meet the numeric nutrient criteria (WERF)	Annual Capital cost to meet the numeric nutrient criteria (L4 WERF) (dollars)	Annual Operations costs to meet the numeric nutrient criteria L4WERF (dollars)	Annual Capital and Operations cost ($)	Annual Additional Cost per Household (increase in sewer rate)	Predicted average household sewer fee to meet criteria	Expected % MHI to Meet Base Numeric Nutrient Criteria (plus current wastewater fees)	Percent increase in Wastewater bill
Big 7 Communities											
Kalispell	BNR (modified Johannesburg); 3.1 to 5.4 MGD; ~WERF Level 2--avg. .12 mg/l TP; 10 mg/l TN.	5.4	3.10	49.14	$3,941,028	$1,228,530	$5,169,558	$671	$1,033	**2.58**	186%
Bozeman	Some BNR now; 5-stage Barrdenpho; new plant will be ~WERF Level 2 on average--BNR (1 mg/l TP; 3 mg/l TN starting 2011); current 5.8 mgd; increasing to 13.9 mgd	13.8	5.80	125.58	$10,071,516	$2,298,540	$12,370,056	$846	$1,218	**2.92**	228%
Helena	BNR; ~ WERF Level 1--3 mg/l TP; 10 mg/l TN; design capacity of 5.4; current discharge ~3.0 MGD	5.4	3.00	67.50	$5,413,500	$1,298,400	$6,711,900	$544	$822	**1.74**	196%
Butte	Current technology is activated sludge (TN of 18.5 mg/l; TP of 2.11 mg/l); under Order to Construct to membrane BNR; current design is 8.5 MGD. Included in current fee is $27 million upgrade in new capital costs and $1.125 million in O&M costs which would bring them to 5 TN and 0.1 TP or ~WERF Level 3	8.5	4.00	62.90	$5,044,580	$1,161,800	$6,206,380	$442	$802	**2.15**	123%
Billings	Secondary treatment; Design flow of 26 MGD (avg.) and 40 MGD max. Costs are estimated from HDR.	26	26	312.50	$25,062,500	$11,252,800	$36,315,300	$868	$1,086	**2.41**	398%
Missoula	Already meets nutrient criteria in Clark Fork with mixing zone. Advanced secondary treatment facility with biological nutrient removal and ultraviolet disinfection. 8.2 mg/l TN; 0.16 -0.4 mg/l TP; get a mixing zone, meeting criteria currently. BNR. Design flow = 12 MGD ; actual flow = 9 MGD. (designed for 10 and 1). (HDR)	12	9	88.80	$7,121,760	$2,614,050	$9,735,810	$353	$505	**1.47**	232%
Great Falls	At WERF 1. Conventional Secondary activated sludge (max 21-MGD; avg. 10 MGD). Cost data from HDR.	26	26	312.50	$25,062,500	$11,252,800	$36,315,300	$1,513	$1,700	**4.18**	808%

Other Large Communities > 1 MGD											
Livingston	Assume WERF Level 1. Discharges into the Yellowstone; permit renewed in 2010; mechanical plant w/ 2 primary clarifiers, 3 rotating biological contactors, UV, installing co-composting. DMR shows 11 mg/l TN average (20 mg/l for May) and 2 mg/l TP (3 mg/l for May).	5	2	62.50	$5,012,500	$865,600	$5,878,100	$1,844	$2,444	**6.85**	307%
Miles City	Assume WERF 1. Secondary treatment plus oxidation ditch. 2011 permit. Algae plant study to remove nutrients. Extended aeration system w/2 oxidation ditches w/rotating brush aerators; 2 clarifiers and chlorine basin. TN avg of 23.5 mg/l; TP avg. 3.6 mg/l.	3.7	2	46.25	$3,709,250	$865,600	$4,574,850	$1,300	$1,537	**4.09**	551%
Hamilton	Assume WERF 2 (TN WERF 3 and TP WERF 1). BNR facility w/ extended aeration system. Oxidation ditch w/ rorating brush aerators. 3 clarifiers. Upgraded in 2010. TN avg. 5.5 mg/l; TP avg. 5 mg/l.	1.98	0.68	24.75	$1,984,950	$301,984	$2,286,934	$1,093	$1,369	**5.44**	396%
Lewistown	Assume WERF 3 based on current levels. BNR plant. Focus on TP removal. 0.8 mg/l TP; 3-4 mg/l TN.	2.5	1.5	18.50	$1,483,700	$423,675	$1,907,375	$699	$1,087	**3.43**	180%
Havre	Assumed WERF Level 1. Discharges into the Milk River. Permit renewed in 2011. Activated sludge facility with effluent chlorination. 2006-2010 data showed avg. TP of 3.4 (TN not required). 2011 DMR showed TN of 19.4 mgl; Tp of 1.3 mg/l.	1.8	1.38	$22.50	$1,804,500	$597,264	$2,401,764	$648	$888	**2.04**	270%
Non-Lagoon Facilities with < 1MGD											
Columbia Falls	Assume WERF Level 3. Newer plant with good control. Designed to achieve 8 mg/l TN	0.766	0.37	$5.67	$454,606	$580,900	$1,035,506	$639	$1,171	**3.02**	120%
Manhattan	Assumed WERF Level 2. Discharges into Diva Ditch. Permit renewed in 2010. Denitrification with fixed film suspended growth system, clarifiers and aerobic sludge digestion, UV. DMR data from winter quarter shows 11 mg/l TN and 1 mg/l TP. 2008-2010 showed avg. TN of 14 mg/l TN	0.6	0.4	$5.46	$437,892	$63,408	$501,300	$959	$1,321	**2.60**	264%
Lolo	WERF Level 1. No steps towards nutrient removal. For Lolo, TN is generally less than 30 mg/l and TP less than 7. Generally heaving loadings for Lolo. Sewer rates--Lolo $30.25-ish/mo - (RSID) based on property values	0.34	0.38	$4.25	$340,850	$164,464	$505,314	$477	$840	**1.81**	131%
Stevensville	WERF Level 1. TN generally below 20 and TP less than 4.	0.3	0.29	$3.75	$300,750	$125,512	$426,262	$536	$1,071	**3.17**	100%

Lagoons											
Philipsburg	WERF 1--Lagoon - ref: Gary Swanson, consulting engineer- 15TN, 2TP	0.2	0.2	$4.36	$ 349,672.00	94,810.00	$444,482.00	$1,114	$1,314	**4.19**	557%
Cut Bank	WERF 0--Lagoon.	0.643	0.643	$14.02	$1,124,195.48	246,140.40	$1,370,335.88	$1,062	$1,201	**2.68**	767%
Deer Lodge	WERF Level 0. Moving from an existing lagoon to mechanical plant with land application. Ref: planning document--To get to variance only. Because this would be a land application system, so theoretically, the N and P would be zero to the Clark Fork	3.3	1.06	$71.94	$1,261,145.00	$502,493.00	$1,763,638.00	$1,159	$1,568	**3.89**	283%
Glendive	WERF Level 0. Domestic WW lagoon; 3 cell facultative; current O&M costs are <$; 8-10 capital costs for new plant. O&M increase of ~$300,000. new avg. 1.15 MGD; PER completed to upgrade to mechanical SBR or BNR plant.	1.3	0.6	$28.34	$2,272,868.00	$284,430.00	$2,557,298.00	$1,358	$1,572	**3.67**	635%
Red Lodge	WERF Level 0--Lagoon.	1.2	0.65	$26.16	$2,098,032.00	$308,132.50	$2,406,164.50	$2,281	$2,586	**5.16**	747%
Big Fork	WERF Level 0--Lagoon.	0.5	0.3	$10.90	$874,180.00	$142,215.00	$1,016,395.00	$595	$1,175	**2.65**	103%
Highwood	WERF Level 0--Lagoon.	0.026	0.015	$0.57	$45,457.36	$7,110.75	$52,568.11	$992	$1,592	**2.54**	165%
Circle	WERF Level 0--Lagoon.	0.16	0.065	$3.49	$279,737.60	$30,813.25	$310,550.85	$1,327	$1,587	**5.47**	511%

NOTE: Operation costs include energy and chemical costs only and do not include labor and maintenance cos
NOTE: The numbers are intended to provide ROUGH ESTIMATES for discussion purposes and (
NOTE: Capital costs were assumed to cover a 20-year bond with 5% interest (used 0.0802 conversion factor)
NOTE: MHI is based on data from Montana CEIC based on 2010 estimates.

Table A-3. WERF Cost numbers

WERF

Level	Description	Capital Cost ($/gpd)	Operations ($1/ MG/day Treated)
Level 1	No N and P removal	9.3	250
Level 2	1 mg/l TP; 8 mg/l TN	12.7	350
Level 3	0.1-0.3 mg/l TP; 4-8 mg/l TN	14.4	640
Level 4	<0.1 mg/l TP; 3 mg/l TN	15.3	880
Level 5	<0.01 mg/l TP; 1 mg/l TN	21.8	1370

Table A-4. WERF Cost calculations for Sample

Costs to Meet Criteria	Capital Cost($million/MGD)	Design Flow	Facility Upgrade Capital Costs ($million)	Annualized Capital Costs (Assumed 20-yr bond & 5% interest; $million/year)	Annualized Capital Costs (Assumed 20-yr bond & 5% interest; $/year)	Operations ($1/ MG/day Treated)	Operations Costs ($/ year/ 1 MGD)	Actual Flow	Facility Upgrade Operations Costs (annual) based on Facility MGD	Membrane Replacement Cost ($24,000 /yr/1 MGD)*Actual Flow	Total Operations costs including membrane replacement
Kalispell	9.1	5.4	$49.14	$3.94	$3,941,028.00	1020	372,300.00	3.10	1,154,130.00	74,400.00	1,228,530.00
Bozeman	9.1	13.8	$125.58	$10.07	$10,071,516.00	1020	372,300.00	5.80	2,159,340.00	139,200.00	2,298,540.00
Helena	12.5	5.4	$67.50	$5.41	$5,413,500.00	1120	408,800.00	3.00	1,226,400.00	72,000.00	1,298,400.00
Butte	7.4	8.5	$62.90	$5.04	$5,044,580.00	730	266,450.00	4.00	1,065,800.00	96,000.00	1,161,800.00
Billings	12.5	25	$312.50	$25.06	$25,062,500.00	1120	408,800.00	26.00	10,628,800.00	624,000.00	11,252,800.00
Missoula	7.4	12	$88.80	7.12176	$7,121,760.00	730	266,450.00	9.00	2,398,050.00	216,000.00	2,614,050.00
Great Falls	12.5	25	$312.50	25.0625	$25,062,500.00	1120	408,800.00	26	10,628,800.00	624,000.00	$11,252,800.00
Livingston	12.5	5	$62.50	$5.01	$5,012,500.00	1120	408,800.00	2.00	817,600.00	48,000.00	$865,600.00
Miles City	12.5	3.7	$46.25	$3.71	$3,709,250.00	1120	408,800.00	2.00	817,600.00	48,000.00	$865,600.00
Hamilton	12.5	1.98	$24.75	1.98495	$1,984,950.00	1120	408,800.00	0.68	277,984.00	24,000.00	301,984.00
Lewistown	7.4	2.5	$18.50	1.4837	$1,483,700.00	730	266,450.00	1.50	399,675.00	24,000.00	423,675.00
Havre	12.5	1.8	$22.50	1.8045	$1,804,500.00	1120	408,800.00	1.38	564,144.00	33,120.00	$597,264.00
Columbia Falls	7.4	0.766	$5.67	0.45460568	$454,605.68	730	266,450.00	2.00	532,900.00	48,000.00	$580,900.00
Manhattan	9.1	0.6	$5.46	0.437892	$437,892.00	1020	372,300.00	0.16	59,568.00	3,840.00	$63,408.00
Lolo	12.5	0.34	$4.25	0.34085	$340,850.00	1120	408,800.00	0.38	155,344.00	9,120.00	$164,464.00
Stephensville	12.5	0.3	$3.75	0.30075	$300,750.00	1120	408,800.00	0.29	118,552.00	6,960.00	$125,512.00
Philipsburg	21.8	0.2	$4.36	$0.35	$349,672.00	1370	450,050.00	0.20	90,010.00	4,800.00	$94,810.00
Cut Bank	21.8	0.643	$14.02	$1.12	$1,124,195.48	1120	358,800.00	0.64	230,708.40	15,432.00	$246,140.40
Deer Lodge	21.8	3.3	$71.94	$5.77	$5,769,588.00	1370	450,050.00	1.06	477,053.00	25,440.00	$502,493.00
Glendive	21.8	1.3	$28.34	2.272868	$2,272,868.00	1370	450,050.00	0.6	270,030.00	14,400.00	$284,430.00
Red Lodge	21.8	1.2	$26.16	2.098032	$2,098,032.00	1370	450,050.00	0.65	292,532.50	15,600.00	$308,132.50
Big Fork	21.8	0.5	$10.90	0.87418	$874,180.00	1370	450,050.00	0.30	135,015.00	7,200.00	$142,215.00
Highwood	21.8	0.026	$0.57	0.04545736	$45,457.36	1370	450,050.00	0.015	6,750.75	360.00	$7,110.75
Circle	21.8	0.16	$3.49	0.2797376	$279,737.60	1370	450,050.00	0.065	29,253.25	1,560.00	$30,813.25

APPENDIX B - ASSUMPTIONS IN THE COST ANALYSIS

DESCRIPTION OF THE ASSUMPTIONS/ DETAILS IN THE SPREADSHEET

- The spreadsheet numbers are intended to provide ROUGH ESTIMATES for discussion purposes and do not reflect the site-specific conditions at each plant.
- The cost estimates for upgrading WWTPs are obtained from the Interim WERF study: "Finding the Balance Between Wastewater Treatment Nutrient Removal and Sustainability, Considering Capital and Operating Costs, Energy, Air and Water Quality and More" (Falk, et al., 2011b). This report is in Draft form and the capital costs are anticipated to increase in the final report based on feedback from the technical reviewers. Based on actual costs observed in Region 1, Region 1 considered the capital costs to be higher than experienced in the final facility plan.
- The total number of WWTPs in Montana that would have to meet base nutrient criteria would be 107. 83 of these are lagoons, and most of these lagoons are small (< 1 MGD).

- Larger, advance WWTPs in Montana would have an easier time meeting nutrient criteria than other WWTPs. In fact, all lagoon systems would face financial hardship meeting the base criteria (> 2% MHI). Therefore, the sample in this analysis focused on the 7 largest communities in MT, 7 medium sized communities with advanced wastewater treatment, 4 smaller communities with advanced treatment < 1MGD, and 8 smaller communities with lagoons.
- Reverse osmosis is assumed to be the technology that would allow WWTPs to have the best chance at meeting base numeric criteria. It is ultimately assumed that 100% of wastewater would need to go through the reverse osmosis process to reach Montana standards.
- The design flows of new RO plants would be the same as current plants, unless otherwise noted. This is a conservative assumption.
- Current sewer rates per household were obtained from direct calls to the municipalities to obtain sewer rate information. Paul LaVigne at DEQ was instrumental in collecting many of these numbers.
- Annual costs of both capital and operations estimates were used in the spreadsheet to calculate the increase in sewer rates and percent MHI.
- Capital costs were assumed to cover a 20-year bond with 5% interest (used a conversion factor of 0.0802). An alternate assumption used a 7% interest rate.
- Level 1 in the Interim WERF Study reflected secondary treatment, which is more advanced treatment than a lagoon system because it assumes a mechanical plant. For lagoons, the total cost of getting to WERF Level 5 (which uses RO) was used and was calculated on a pro-rated basis (per flow), minus the current O&M costs for a lagoon. Current O&M costs for a facultative lagoon are assumed to be $50,000 annually for all FLs and $150,000 for an Aerated Lagoon.
- WERF level 5 is not quite as stringent as the Montana base nutrient criteria for TN, so the costs to reach nutrient standards in Montana are underestimated. An alternate assumption addresses this issue.
- For the Montana towns in this analysis with advanced treatment, the cost associated with the WERF level they are currently at is subtracted from WERF level 5 costs in the study. That means that all WWTPs in our sample already at WERF level 2 will have the same estimated unit capital and O&M costs to meet base numeric criteria. Estimate total costs will differ based on facility flow.

- Operation costs in the WERF study, and therefore in this analysis, include energy and chemical costs only and do not include labor and maintenance cost. As such, the O&M cost numbers in this analysis are on the low side. An alternate assumption addresses this issue by adding labor costs.
- The costs in this demonstration do not include lagoon abandonment, so they may underestimate total costs.
- Capital and O&M costs for lagoons to get up to WERF 5 are based on building from scratch, assuming that no infrastructure exists. This assumption is valid, because for lagoon systems converting to RO, it would be the same as a greenfield project, since a lagoon would have to do a complete rebuild. In addition, a lagoon would have to be decommissioned and abandoned which could be expensive (abandonment costs are not included in this analysis_.
- To get to RO, a membrane Replacement Cost is added which is estimated at $24,000 /yr/1 MGD. Brine disposal costs are included within the WERF numbers.
- Design flow of a given WWTP was used to determine the capital costs and actual flow was used for the Operations costs. Flows for towns were taken from wastewater permits.
- A community's population was estimated from Census 2010. The number of households in a community was estimated from the American Community Survey 5-year estimate 2005-2009. The number of households was used as a proxy for the number of hookups per WWTP, as that number was often hard to obtain from operators.
- A threshold total cost per household of 2% of a town's median household income (MHI) includes: 1) current wastewater fees plus 2) additional wastewater fees to meet base criteria. Greater than 2% MHI of these two costs is considered a significant cost per the Guidance. A town then moves on to the second 'Significant Test' of secondary economic indicators. Because 104 out of 107 towns would experience costs of greater than 2% (MHI), and because current rates average just under 0.9% MHI, the average wastewater rate in Montana in affect towns would more than double to meet standards.

APPENDIX C - SECONDARY INDICATORS

Table C-1 Secondary Indicators for the Municipality.

Example of Town X: Poverty rate 20%, LMI 47%, Unemployment rate 7.1%, MHI $39,201, Property Tax index number 1.21%.

	Secondary Indicators			
Indicator	**Weak***	**Mid-Range****	**Strong*****	**Score**
Poverty Rate	More than 16%	4-16%	Less than 4%	1
Low to Medium Income Percentage (LMI)	More than 51%	23-51%	Less than 23%	2
Unemployment	More than 1% above State Average (>8.2%)	State Average 2009----7.2%	More than 1% below State Average (<6.2%)	2
Median Household Income	More than 10% below State Median	State Median $42,322 (2009)	More than 10% above State Median	2
Property Tax, fees and revenues divided by MHI and indexed by population	More than 3.0	3.0 to 1.5	Less than 1.5	3
* Weak is a score of 1 point				
** Mid-Range is a score of 2 points				
*** Strong is a score of 3 points			**SUM:**	10
			AVERAGE:	2.00

There are five socioeconomic criteria that are summed up and averaged to see where the households within a community fall in terms of financial health. For each of the five criteria, a strong score is recorded in the right hand column as a '3', indicating strong socioeconomic health for that criteria and thus a greater chance of being able to pay for additional wastewater treatment (and lesser chance of a variance). A mid-range score is recorded as a '2' and indicates moderate or average socioeconomic health for the particular criteria. A weak score should be recorded as a '1' and indicates poor socioeconomic health for the given criteria or less ability to pay (and a greater chance of being granted a variance). The average score of all five indicators falls into those same categories and should be judged in the same way.

For poverty rate and LMI, the strong, mid-range and weak score are derived by taking averages of each of these five indicators for a sample of 41 selected towns and then running a histogram. The histogram using the latest data gives us breaks for strong, mid-range, and weak scores using best professional judgment. The same method is used for Property tax, fees, etc. except that a sample of 49 towns was used to create the histogram, due to the large data requirements and that we had to calculate this figure ourselves. For unemployment and MHI, towns are compared to the state average.

The last criteria, Property tax, fees and revenues divided by MHI and population, gives an indication of the existing burden on local residents within the municipality of fees for local services and of local taxes. Those citizens of towns already paying a lot of money relatively for services such as wastewater and garbage and/or paying higher local taxes are assumed to be less able to pay additional monies for additional wastewater treatment since they already have a formidable local tax burden.

Specific assumptions for the Secondary test include:

- Population estimates were compiled by the Montana CEIC and are based upon Census 2010. Median household income and number of households per community were compiled from the Montana CEIC and are based on the American Community Survey 5-Year Estimate (2005-2009)
- Local area taxes, revenues and property taxes are from Fiscal year ending June 30, 2010. This information is from the Local government Services Bureau, Montana Department of Administration, Kim Smith, (406) 841-2905, kims@mt.gov. There is not tax data Big Fork and Highwood because they are not incorporated, and thus not required to report this data. Broadus and Columbia Falls gave unaudited financial statements in FY2010 and are 'audit report delinquent', but the numbers were used anyway. Ekalaka, Froid, Fromberg, Hamilton, Ismay, Lima and Sidney's FY 2010 reports are unaudited. Deer Lodge data from FY 2008 due to no recent reporting. For those towns for whom this tax data does not exist, their average secondary score was based on four economic metrics rather than five.
- To calculate the Local area taxes, revenues and property tax index, the following three items from each town are summed up: 1) General Government Activities (Charges for Services, Fines, Forfeitures, including public works, safety, interest on debt and health), 2) Business Type Activities (Hospital, water, sewer, solid waste, airport, business), 3) Local property taxes. The sum of these three items is then divided by that town's MHI. The town's population is divided by 50,000 to index it—create a population index. The sum of the three items divided by MHI is divided by the population index to come up with the Local area taxes, revenues and property tax index. The index numbers were taken for all towns in this study and a histogram was run in Excel to determine cutoff points for a weak score (the town already has a lot of local taxes to pay compared to other towns which translates to a high index number—greater than 3.0 index score), a mid-range score, or a strong score (the town currently has a low amount of local taxes/fees to pay compared to other towns which translates to a low index number—less than 1.5 index score)
- Unemployment rates are from July of 2011 from Aaron McNay, Economist, Montana Department of Labor and Industry, 406-444-3245. They only have unemployment estimates for cities that have a population that is 25,000 or larger. For all the other cities, we can only provide county level estimates for the county they are in. Butte and Silver Bow county are considered one entity, so the county number was reported. Only Billings, Bozeman, Helena, Missoula and Great Falls have actual unemployment rate estimates for the city.
- Low and Moderate Income Percent was calculated using a proxy for the HUD definition of LMI. Low and Moderate Income Percent is calculated by U.S. Housing and Urban Development (HUD) using data from the U.S. Census Bureau's Decennial Census, specifically for the Community Development Block Grant Program (CDBG). LMI families are defined as those families whose income does not exceed 80% of the county median income for the previous year or 80% of the median income of the entire non-metropolitan area of the State of Montana, whichever is higher. (U.S. Census Bureau, 2012). It is this method that was used to calculate Montana's 2000 LMI numbers. HUD did not update their figures from 2000, so DEQ had to calculate it own version of LMI.
- LMI for 2011 was calculated by DEQ by taking the number of persons who live below 200% of the poverty level threshold for a town, and dividing by the total number of persons in a town. The data used was the 2005-2009 American Community Survey 5-Year Estimates. The resulting numbers are similar to 2000 numbers using the HUD definition because 200% poverty level is close to 80% of Montana's median family income (MFI), which is close to the 2000 HUD definition for LMI. A histogram was used to create break points for strong, medium, and weak LMI scores.

- The source for poverty rate is the 2009 American Community Survey Data and the Social Explorer website. (http://www.socialexplorer.com/pub/reportdata/metabrowser.aspx?survey=ACS2009_5yr&ds=Social+Explorer+Tables%3A++ACS+2005+to+2009+(5-Year+Estimates)&table=T118&header=True) To determine a person's poverty status, one compares the person's total family income in the last 12 months with the poverty threshold appropriate for that person's family size and composition. If the total income of that person's family is less than the threshold appropriate for that family, then the person is considered "below the poverty level," together with every member of his or her family. If a person is not living with anyone related by birth, marriage, or adoption, then the person's own income is compared with his or her poverty threshold. The total number of people "below the poverty level" is the sum of people in families and the number of unrelated individuals with incomes in the last 12 months below the poverty threshold. The official poverty thresholds do not vary geographically, but they are updated for inflation using Consumer Price Index (CPI-U). The official poverty definition uses money income before taxes and does not include capital gains or noncash benefits (such as public housing, Medicaid, and food stamps). (http://www.census.gov/hhes/www/poverty/methods/measure.html). A histogram was used to create break point for strong, medium and weak LMI scores.

Table C-2. Secondary Score Case Studies--Public WWTPs

	Poverty Rate % (2009)	LMI % (2009)	Unemployment Rate % (July 2011)	MHI (estimated 2009 dollars)	Total Revenues, Fees and Taxes index
Baker	8.18	27.9	2.7	47,305	1.80
Big Fork	2.19	16.0	10.4	44,398	N/A
Billings	8.49	31.4	5.5	45,004	2.31
Bozeman	10.68	39.8	6	41,661	2.66
Butte	10.51	38	6.7	37,255	1.42
Broadus	0	24	5.3	45,938	3.71
Circle	3.97	54.4	2.9	29,000	2.88
Columbia Falls	6.38	42.8	10.4	38,750	1.94
Cut Bank	17.92	35.9	11.7	44,833	2.12
Deer Lodge	8.67	35.4	8.9	40,320	1.14
Ekalaka	9.48	34.1	3.9	32,917	3.02
Ennis	6.44	46.0	7.2	37,639	2.64
Eureka	12.85	61.4	14.6	37,813	1.96
Froid	8.16	26.9	9	24,706	3.50
Fromberg	6.18	26.0	6.2	42,011	1.34
Glendive	7.34	24.4	4.4	42,821	1.99
Great Falls	11.85	34.1	6.9	40,718	2.63
Hamilton	19.47	46.8	9.9	25,161	3.45
Havre	9.41	36.5	6.5	43,577	1.73
Helena	5.96	28.5	5.5	47,152	2.60
Highwood	0	7.50	5	62,614	N/A
Ismay	0	0.0	4.7	32,083	0.41
Kalispell	14.2	40.4	10.4	39,953	2.43
Lewistown	13.6	47.4	5.8	31,729	2.72
Libby	10.14	51.0	14.6	27,267	3.21
Lima	11.11	66.5	5.9	27,875	1.90

Table C-2. Secondary Score Case Studies--Public WWTPs

	Poverty Rate % (2009)	LMI % (2009)	Unemployment Rate % (July 2011)	MHI (estimated 2009 dollars)	Total Revenues, Fees and Taxes index
Livingston	8.08	34.0	7	35,689	3.31
Lolo	9.5	33.6	7.4	46,422	N/A
Manhattan	5.22	30.7	6.3	50,729	1.56
Miles City	11.5	38.1	4.7	37,554	2.17
Missoula	11.15	44.8	6.9	34,319	1.79
Neihart	9.52	12.1	5.6	42,312	3.32
Phillipsburg	10.57	48.7	10.1	31,375	2.26
Plentywood	1.57	34	3.8	36,632	1.70
Red Lodge	6.28	34.7	6.2	50,123	2.90
Roundup	17.27	51.4	6.9	33,750	1.75
Shelby	5.25	35.4	5.2	40,552	2.60
Sidney	23.76	38.6	3.5	49,784	0.74
St. Ignatius	29.63	56.6	10.9	28,542	1.62
Stevensville	20.19	56.1	9.9	33,776	1.72
West Yellowstone	14.35	38.5	6.3	39,231	3.06

Table C-3. Secondary Score Case Studies--Public WWTPs Actual Secondary Scores

	Poverty Rate Secondary Score	LMI Secondary Score	Unemployment Rate Secondary Score	MHI (estimated 2008 number) Secondary Score	Total Revenues, Fees and Taxes index Secondary Score	Average
Baker	2	2	3	3	2	2.4
Big Fork	3	3	1	2	N/A	2.25
Billings	2	2	3	2	2	2.2
Bozeman	2	2	3	2	2	2.2
Butte	2	2	2	1	3	2
Broadus	3	2	3	2	1	2.2
Circle	3	1	3	1	2	2
Columbia Falls	2	2	1	2	2	1.8
Cut Bank	1	2	1	2	2	1.6
Deer Lodge	2	2	1	2	3	2
Ekalaka	2	2	3	1	1	1.8
Ennis	2	2	2	1	2	1.8
Eureka	2	1	1	1	2	1.4
Froid	2	2	1	1	1	1.4
Fromberg	2	2	2	2	3	2.2
Glendive	2	2	3	2	2	2.2
Great Falls	2	2	2	2	2	2
Hamilton	1	2	1	1	1	1.2
Havre	2	2	2	2	2	2
Helena	2	2	3	3	2	2.4
Highwood	3	3	3	3	n/a	3
Ismay	3	3	3	1	3	2.6
Kalispell	2	2	1	2	2	1.8
Lewistown	2	2	3	1	2	2

Table C-3. Secondary Score Case Studies--Public WWTPs Actual Secondary Scores

	Poverty Rate Secondary Score	LMI Secondary Score	Unemployment Rate Secondary Score	MHI (estimated 2008 number) Secondary Score	Total Revenues, Fees and Taxes index Secondary Score	Average
Libby	2	2	1	1	1	1.4
Lima	2	1	3	1	2	1.8
Livingston	2	2	2	1	1	1.6
Lolo	2	2	2	2	n/a	2
Manhattan	2	2	2	3	2	2.2
Miles City	2	2	3	1	2	2
Missoula	2	2	2	1	2	1.8
Neihart	2	3	3	2	1	2.2
Phillipsburg	2	2	1	1	2	1.6
Plentywood	3	2	3	1	2	2.2
Red Lodge	2	2	2	3	2	2.2
Roundup	1	1	2	1	2	1.4
Shelby	2	2	3	2	2	2.2
Sidney	1	2	3	3	3	2.4
St. Ignatius	1	1	1	1	2	1.2
Stevensville	1	3	1	1	2	1.6
West Yellowstone	2	2	2	2	1	1.8

Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels

DECEMBER 2011

Prepared by:
Water Quality Planning Bureau, Water Quality Standards Section
Montana Department of Environmental Quality
Water Quality Planning Bureau
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

WQPBMASTR-01

Prepared by:
Water Quality Planning Bureau
Michael Suplee, Ph.D., Water Quality Standards Section
Rosie Sada de Suplee, Monitoring and Assessment Section

Suggested citation: Suplee, M.W., and R. Sada de Suplee, 2011 Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels. Helena, MT: Montana Dept. of Environmental Quality.

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ACRONYMS

Acronym	Definition
ADB	Assessment database
AFDW	Ash Free Dry Weight
ARM	Administrative Rules of Montana
BACI	Before-After-Control-Impact
BACIP	Before After Control Impact Paired
BOD	Biochemical Oxygen Demand
BPJ	Best Professional Judgment
CFR	Clark Fork River
CV	Coefficient of Variation
DEQ	Department of Environmental Quality (Montana)
DO	Dissolved Oxygen
DPHHS	Department of Health and Human Services
EPA	Environmental Protection Agency (US)
HBI	Hilsenhoff Biotic Index
LOWESS	locally-weighted regression line
QAPP	Quality Assurance Project Plan
SAP	Sampling and Analysis Plan
SOD	sediment oxygen demand
SOP	Standard Operating Procedures
SRP	Soluble Reactive Phosphate
TKN	Total Kjeldahl Nitrogen
TMDL	Total Maximum Daily Load
TN	Total Nitrogen
TP	Total Phosphorus
WQPB	Water Quality Planning Bureau (DEQ)

NITROGEN AND PHOSPHORUS IMPAIRMENT ASSESSMENT: METHOD SUMMARY

The following Method Summary should provide sufficient detail for an assessor to undertake an assessment of nitrogen and phosphorus impacts in a wadeable stream. *You will probably still need to refer to details provided later in the document.* Large rivers are <u>not</u> addressed by this methodology; a list of large rivers to which these methods do not apply is shown below in **Table S-1**.

Table S-1. Non-wadeable river segments within the state of Montana

River Name	Segment Description
Big Horn River	Yellowtail Dam to mouth
Clark Fork River	Bitterroot River to state-line
Flathead River	Origin to mouth
Kootenai River	Libby Dam to state-line
Madison River	Ennis Lake to mouth
Missouri River	Origin to state-line
South Fork Flathead River	Hungry Horse Dam to mouth
Yellowstone River	State-line to state-line

Part 1: Defining the assessment reach

1. Compliance determinations described in this document are carried out on an <u>assessment reach.</u> Here we define an assessment reach as: *a wadeable stream segment listed in the Assessment Data Base (ADB;(Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, 2009) and updates),* <u>*or*</u> *a sub-segment of an ADB stream segment.* A sampling unit within an assessment reach is defined *as: a sample collected from the assessment reach that is largely independent of other samples collected within the assessment reach and collected during the time when the numeric nutrient criteria apply.* Please consider the following:

   a. The aggregate of samples collected from an assessment reach should provide good overall representation of the assessment reach. Individual sites within the assessment reach that have known or suspected pollution problems should be sampled equitably along with sites where pollution problems are not suspected or are minimal or less pronounced. Do not just target the hotspots.

   b. <u>Given the guidelines in 1a above</u>, the assessor will have to judge if further stratification of the stream reach (i.e., create two or more sub-reaches) is warranted. If, for example, a relatively un-impacted upstream reach of an assessment reach can be isolated and its condition is substantially different from other downstream parts of the assessment reach, sub-segmenting may likely be justified. As a rule of thumb, it is better to lump than split reaches to avoid excessive sub-segmentation of streams and the consequential administrative and sampling requirements.

   c. Each sub-reach will have the same general data requirements (dataset minimums, tests, etc.) as the parent assessment reach would have had if it hadn't been divided.

   d. Samples should be collected when the criteria apply, during the ecoregion-specific Growing Season (**Table S-2**). However, a ten day window (plus/minus) on the Growing Season start and end dates is acceptable in order to accommodate year-specific conditions (e.g., an early-ending spring runoff). Samples collected outside the Growing Season may be useful for other purposes (e.g., isolating load sources), but should not to be used for compliance determination for the Growing Season.

Table S-2. Start and Ending Dates for Three Seasons (Winter, Runoff and Growing), by Level III Ecoregion

Ecoregion Name	Start of Winter	End of Winter	Start of Runoff	End of Runoff	Start of Growing Season	End of Growing Season
Canadian Rockies	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Northern Rockies	Oct.1	March 31	April 1	June 30	July 1	Sept. 30
Idaho Batholith	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Middle Rockies	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Northwestern Glaciated Plains	Oct.1	March 14	March 15	June 15	June 16	Sept. 30
Northwestern Great Plains	Oct.1	Feb. 29	March 1	June 30	July 1	Sept. 30
Wyoming Basin	Oct.1	April 14	April 15	June 30	July 1	Sept. 30

2. Samples from within an assessment reach may generally be considered independent of other samples from the assessment reach if they meet or if you do the following:
   - Sites (or very short reaches functionally equivalent to sites) should be located at least 1 stream mile apart.
   - Sites may be placed < 1 mile apart on an assessment reach **if** there is a flowing tributary confluencing with the reach between the two sites.
   - Along an assessment reach, try to sample sites moving from downstream to upstream to avoid potentially re-sampling the same stream water.
   - Land use changes and land form changes should be considered and can be used to help define (1) breaks between assessment reaches and/or (2) additional sampling sites within an assessment reach
   - Samples collected at the same site should be collected about 30 days apart. This does not apply to long-term or instantaneous measurement of dissolved oxygen.

Part 2: Assessment Methodology

The following are recommended to determine nitrogen and phosphorus impacts in wadeable streams.

For Mountainous and Transitional[1] Streams: Assessment is carried out as a level I, level II process. If the level I results are inconclusive, move to the 2nd level[2].

Level I:

a. Collect, within the assessment reach, benthic algal Chl *a* and AFDW (Ash Free Dry Weight) from one or more sites[3] (following DEQ SOPs, including approved low-chlorophyll visual

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[1] See **Table 4-1** later in this document for the list of ecoregions (levels III and IV) where this methodology applies.

[2] Nothing precludes the assessor from collecting, in a single sampling season, all data needed to carry out a level II assessment. Cases may arise (e.g.., land access issues) that may make this approach preferable.

[3] Treat each Chl *a* sampling event as an independent evaluation of use support Do not average together results from different sites within the assessment reach If Chl *a* is measured more than once at the same site, treat each sampling event as unique (NOT as temporal repeat measures)

estimation methods) for a minimum of three sampling events. A minimum of twelve or thirteen[4] independent nutrient samples should be collected within the same assessment reach. Use of diatom samples at level I are optional, but if the data exist (n ≥ 2 samples), they must be used in the assessment. Disperse sampling effort across sites as much as possible. The nutrient data are evaluated using the "Exact Binomial Test" and the "One-Sample Student's t-test for the Mean" which are housed in one of two Excel spreadsheets. If the assessment reach is a new, un-listed segment, use "MT-NoncomplianceTool.xls". If the assessment reach is already listed for a nutrient, use MT-ComplianceTool.xls". However, if a stream is currently listed for nitrogen but not for phosphorus, use the "MT-NoncomplianceTool.xls" to assess the phosphorus data.

b. In both spreadsheets, for either test, set alpha to 0.25 (25%) and the critical exceedance rate (p) to 0.2 (20%) in cells B5, B6. In the Binomial test the effect size (p2; gray zone) should be 0.15 (15%) and is set as a function of the exceedance rate. So, in "MT-NoncomplianceTool.xls" this means p2 should be set to 0.35 (cell B7), and in "MT-ComplianceTool.xls" p2 should be set to 0.05 (cell B7). If in the future DEQ decides that a lower exceedance rate (e.g., 10%) is needed, the gray zone will need to be adjusted accordingly.

c. Compliance with the nutrient criteria is determined via decision rules, which consider the Chl *a* and AFDW averages calculated for each sampling event, the results from the two nutrient statistical tests, and diatom metric results (if available). Go to the first tab of the Excel spreadsheet named "NtrntAssessFramework.xlsx". If the result is clear (assessment reach is or is not nutrient impaired), you are finished. If not, follow the instructions in the spreadsheet for level II assessment.

d. Most often, you will be assessing both N and P in an assessment reach. Consider the N and P results side-by-side; does is appear that one nutrient or the other is giving a clear signal (e.g., Binomial and T-test are both FAIL for Total Phosphorus (TP), but both PASS for Total Nitrogen (TN)? In this case, the best nutrient to list would be TP. Mixed results for both nutrients often will require a move to a level II assessment, and may lead to listing both N and P.

Level II:

a. Moving to level II often (not always) involves additional data collection, including more nutrient samples and benthic algal Chl *a*/AFDW samples. Level II data include both diatom and macroinvertebrate samples (at least two sampling events for each). **The exception to this is the Middle Rockies ecoregion, for which there are no validated diatom increaser metrics. In this ecoregion collect at least three macroinvertebrate samples.** As for level I, each sampling event for diatoms should be considered on its own merits (do not average results across sites, or across time at a site). In contrast, macroinvertebrate samples collected across time at a site should be averaged together; however, keep and assess data from different sites separate. When your dataset is ready, first pass data again through the

[4] Twelve independent nutrient samples for new, unlisted streams, and thirteen for streams already listed for nutrients on the 303(d) list. A nutrient sample is a type of nutrient (e.g., TP or TN) Sample minimums apply to *each* nutrient type. Smaller sample sizes may be justified; see **Section 3.2.2.1**, this document

Level I process using NtrntAssessFramework.xlsx. If results are clear, you are finished; if not, go to the 2nd tab.

b. Some data combinations at level II (2nd tab of the spreadsheet) will still lead to an unclear result. If this occurs, consult with your manager about how to proceed.

For Warm Water Plains Streams[5]: Assessment is carried out as a level I, level II process. If the level I results are inconclusive, move to the 2nd level[2].

Level I:

a. Determine, for at least 3 sampling events, the dissolved oxygen (DO) delta (i.e., the daily DO maximum minus the daily DO minimum). The daily minimum can be measured pre-dawn to 8:00 am, while the daily maximum usually occurs between 2:30 pm to 5:00 pm. Alternatively, collect a long-term DO dataset by deploying a YSI 6600 (or similar instrument) in at least one site; measure DO for at least 1 full day, with a 15-min time step. Even if you collect DO data with a deployed instrument, you still need a total of three sampling events (three days). **However, DO delta values need not be collected 30 days apart.** Also, collect within the assessment reach at least two diatom samples and a minimum of twelve or thirteen[4] nutrient samples. Disperse sampling effort across sites as much as possible. The nutrient data are evaluated using the tests "Exact Binomial Test" and the "One-Sample Student's t-test for the Mean" found in one of two Excel spreadsheets. If the assessment reach is a new, un-listed segment, use "MT'NoncomplianceTool.xls". If the assessment reach is already listed for a nutrient, use MT-ComplianceTool.xls". However, if a stream is currently listed for nitrogen but not for phosphorus, use the "MT-NoncomplianceTool.xls" to assess the phosphorus data.

b. See 1b above (in the Mountainous and transitional streams section) for instructions on setting test conditions in each Excel spreadsheet.

c. Compliance with the nutrient criteria is determined via decision rules, which consider together the results from the diatom metrics, the DO delta values, and the two statistical tests for nutrients. Go to the 3rd tab (plains level I) of the Excel spreadsheet "NtrntAssessFramework.xlsx". Long-term DO datasets require special consideration; see **Section 3.2.4**, scenario 2, and **Section 5.0** for details. If the result is clear (assessment reach is or is not nutrient impaired), you are finished. If not, follow the instructions in the spreadsheet for level II assessment.

d. Most often, you will be assessing both N and P in an assessment reach. Consider the N and P results side-by-side; does is appear that one nutrient or the other is giving a clear signal (e.g., Binomial and T-test are both FAIL for TN, but both PASS for TP)? In this case, the best nutrient to list would be TN. Mixed results for both nutrients often will require a move to a level II assessment, and may lead to listing both N and P.

[5] See **Table 5-1** later in this document for the list of ecoregions where this methodology applies.

Level II:

a. A level II assessment will often require additional data collection, including more nutrient and DO data, and (in some cases) biochemical oxygen demand (BOD_5) data. As for level I, each DO delta value should be considered on its own merits (do not average results across sites or across time), as is also the case for BOD_5 samples. When the data are ready, first pass the now-larger dataset back through the Level I assessment process. If results are clear, you are finished; if not, go to the appropriate scenario in the 4th tab (plains level II).

b. Some level II data combinations still lead to an unclear result. If this occurs, consult with your manager about how to proceed.

1.0 INTRODUCTION

The purpose of this document is to describe a framework for making decisions. Specifically, it defines a process by which one can determine if a wadeable stream is or is not impaired by nitrogen and phosphorus pollution (i.e., excess nutrients). The document covers a number of subjects including how to determine an appropriate sampling frame, which parameters are most useful for assessing nitrogen and phosphorus problems, how many samples are needed, how data are to be treated statistically, and how disparate data types are to be assembled in a final decision matrix.

In this document we have attempted to organize the information in a manner such that the users can locate what they needs quickly, and then read further for details only if they want to. The “why” discussions (i.e., why did we select a particular assessment parameter, why did we pick the impact threshold, etc.) are found in the appendices. We did this because we know that stream assessments are time-consuming undertakings, and therefore users will want to access the critical information easily.

1.1 SCOPE OF THE ASSESSMENT METHODOLOGY

Different assessment methods are recommended for different regions of the state (**Figure 1-1**). The assessment parameters that have been recommended for each region are the ones we believe are the most accurate and sensitive for determining nitrogen and phosphorus impacts for wadeable streams in those areas. For example, we recommend measuring dissolved oxygen in eastern Montana plains streams, but we have not recommended this approach for western Montana salmonid streams. This should not be construed to mean that DO concentrations in salmonid streams are never affected by nitrogen and phosphorus pollution, or that in any way this recommendation overrides existing DO standards for those state waters. Rather, we believe that in western Montana salmonid streams there are assessment tools other than DO that are more sensitive and will more readily detect nitrogen and phosphorous problems.

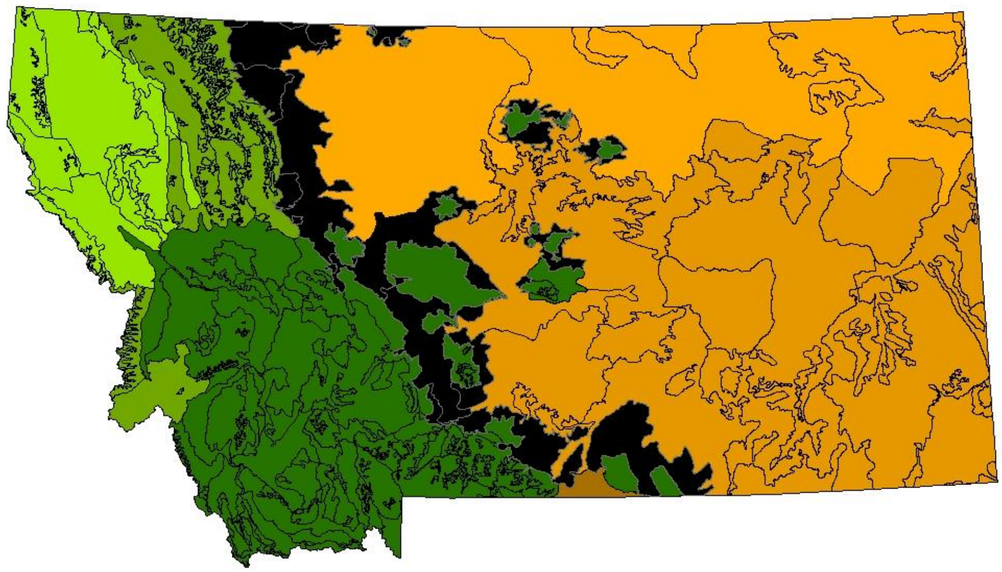

Figure 1-1. Map Showing Different Regions in which Different Assessment Methodologies Apply. The areas shown in shades of green and black comprise the mountain and transitional streams region. The areas shown in shades of brown comprise the eastern Montana plains region.

As mentioned above, methods in this document apply to wadeable streams. A DEQ workgroup spent considerable time working to define the break between wadeable streams and rivers and non-wadeable rivers, the results of which are presented in Flynn and Suplee (2010). The waterbodies that are not considered wadeable are provided below in **Table 1-1.**

Table 1-1. Non-wadeable river segments within the state of Montana

River Name	Segment Description
Big Horn River	Yellowtail Dam to mouth
Clark Fork River	Bitterroot River to state-line
Flathead River	Origin to mouth
Kootenai River	Libby Dam to state-line
Madison River	Ennis Lake to mouth
Missouri River	Origin to state-line
South Fork Flathead River	Hungry Horse Dam to mouth
Yellowstone River	State-line to state-line

2.0 Recommendations for Isolating a Stream Reach for Assessment (the Sample Frame)

Identifying and isolating an appropriate stream reach (sample frame) is the first required task. The following definitions are presented:

- **Sample Frame**: A wadeable[6] stream segment listed in the Assessment Data Base (ADB) (DEQ 2009, and updates) OR a sub-segment of an ADB stream segment. A segment such as this is referred to in this document as an "assessment reach".
- **Population**: All the water flowing through the assessment reach during the time period when the numeric nutrient criteria apply, and the surface area of the stream bottom over which the water flows.
- **Sampling Unit**: A sample collected from the assessment reach that is largely independent of other samples collected within the assessment reach and collected during the time when the numeric nutrient criteria apply.

A sampling frame must be representative of the population and, in stream assessment, this demands good judgment in the particular subject matter being studied. **Sections A.1** and **A.2** of **Appendix A** of this document, which is an updated and shortened version of an earlier Appendix H (Varghese and Cleland, 2008), contain a discussion on approaches to identifying assessment reaches.

The key idea presented in **Appendix A** is that each assessment reach that is assessed should be sufficiently homogenous that data collected from sites along the reach can be considered to represent the entire reach. To determine compliance with numeric nutrient criteria using statistical methods, it is important that (1) pollution sources generally be evenly dispersed along the reach, and (2) each sample is independent of the others. Following up on this idea, if an assessment reach appears to need further subdivision (e.g., into a reach above and a reach below a pollution point source), then each new assessment reach should generally be sampled with the same intensity (i.e., minimum sample size) as the parent reach would have been if it had not been subdivided. This will assure that the statistical rigor associated with specified sample-size minima (discussed below) is maintained. At the same time, as a general rule, it is better to lump than split to avoid unnecessary sampling and administrative work.

The need to create reasonably uniform assessment reaches is inherently in conflict with the need to "lump", the purpose of which is to keep stream reaches from being excessively subdivided (and all the additional work that entails). Judgment is needed on the part of the assessor to balance these two opposing factors and come up with an optimal sampling strategy for any given stream.

This process should be compatible with a randomized study of stream reaches as well as targeted, risk-assessment based approaches; again, the key point is that each assessment reach is sufficiently defined.

3.0 Assembling the Nutrient Assessment Data into a Decision Framework

Section 2.0 above discussed approaches used to identify appropriate assessment reaches. This section discusses how data that will have been collected from the assessment reach are to be assembled into a decision-making framework. The parameters and methods apply to wadeable streams. Non-wadeable waterbodies were listed back in **Table 1-1.**

[6] Wadeable streams are perennial as well as intermittent (ARM 17.30.602 [15]) streams in which large portions of the channel are wadeable during baseflow conditions. For the list of waterbody segments not considered wadeable (i.e., the large rivers), see **Table 1-1** above Derivation of the **Table 1-1** list is found in Flynn and Suplee (2010).

3.1 Overview of Useful Parameters for Carrying out Nutrient Impairment Assessments

Among the vast array of parameters that can be measured in a stream, we narrowed the list to those we believe are the best, readily-measured indicators of stream nitrogen and phosphorus enrichment (**Table 3-1**). Many of these parameters are discussed in Suplee *et al.* (2008), and are also discussed in detail in the appendices of this document.

Table 3-1. Parameters in Streams that are Considered Useful in Assessing Nutrient Enrichment

Parameter	How collected	Linkage to nutrient enrichment	Primary or secondary indicator*
Total nitrogen	Water sample	Total instream concentrations are indicative of the level of nutrients that are ultimately biologically available for autotrophic or heterotrophic uptake.	Primary
Total phosphorus	Water sample	Total instream concentrations are indicative of the level of nutrients that are ultimately biologically available for autotrophic or heterotrophic uptake.	Primary
Benthic algal biomass	Benthic samplings of stream bottom	Nutrients stimulate benthic algal growth in wadeable streams Benthic algal growth can develop to nuisance levels; nuisance algae level is known. Excess algal growth affects on DO have been documented.	Primary
Dissolved oxygen *delta* (daily max value minus the daily min value)	Instantaneous: By hand-held instrument, at dawn and in the late pm. Continuous monitoring: by deployed instrument	Nutrient enrichment stimulates autotrophic primary productivity and heterotrophic decomposition of organic material. Both of these in turn affect dissolved oxygen patterns in streams.	Primary
Diatom biometric (nutrient increaser taxa metric)	Benthic sampling of stream bottom	As primary producers, diatoms can be directly stimulated by increased availability of N and P. Diatom population structure has been found to vary in predictable ways with increasing nutrient enrichment.	Primary and Secondary
Macroinvertebrate biometric (Hilsenhoff Biotic Index, or HBI)	Kicknet sampling of stream bottom	A large number of macroinvertebrate taxa have been assigned a numeric value which represents each organism's tolerance to low dissolved oxygen/organic pollution. Resulting metric (HBI) found to significantly correlate to total nutrient concentrations in Montana streams.	Secondary
Biochemical oxygen demand (BOD_5)	Water sample; must be at laboratory within 48 hrs	High BOD can indicate presence of large quantities of dissolved and suspended organic matter, whose decomposition can produce a large DO demand. Can help determine if DO sags are caused by high primary productivity, high BOD, or both.	Secondary

Table 3-1. Parameters in Streams that are Considered Useful in Assessing Nutrient Enrichment

Parameter	How collected	Linkage to nutrient enrichment	Primary or secondary indicator*
Stream macrophyte species	By hand; field identification	Observed shift in dominance to a single macrophyte species in highly enriched prairie streams; loss of *Chara*.	Secondary

*Primary means the parameters is considered to be a very good indicator of nutrient enrichment. Secondary mean the parameter is considered a good indicator of nutrient enrichment, or helpful in identifying other factors affecting DO (e.g., BOD).

Note that **Table 3-1** contains physical and biological measurements. We support the long-held view in the WQPB that stream assessment is best carried out by looking at both data types together. The famous water-pollution biologist H.B Hynes said it best: “When the chemist and the biologist both work on the assessment of pollution they can discover much more together than either can alone” (Hynes, 1966).

The parameters in **Table 3-1** need to be arranged in a decision making framework in order to produce consistent decision outcomes (i.e., stream *is* impaired by nutrients, stream *is not* impaired by nutrients). **Figure 3-1** below outlines the process we recommend for this purpose.

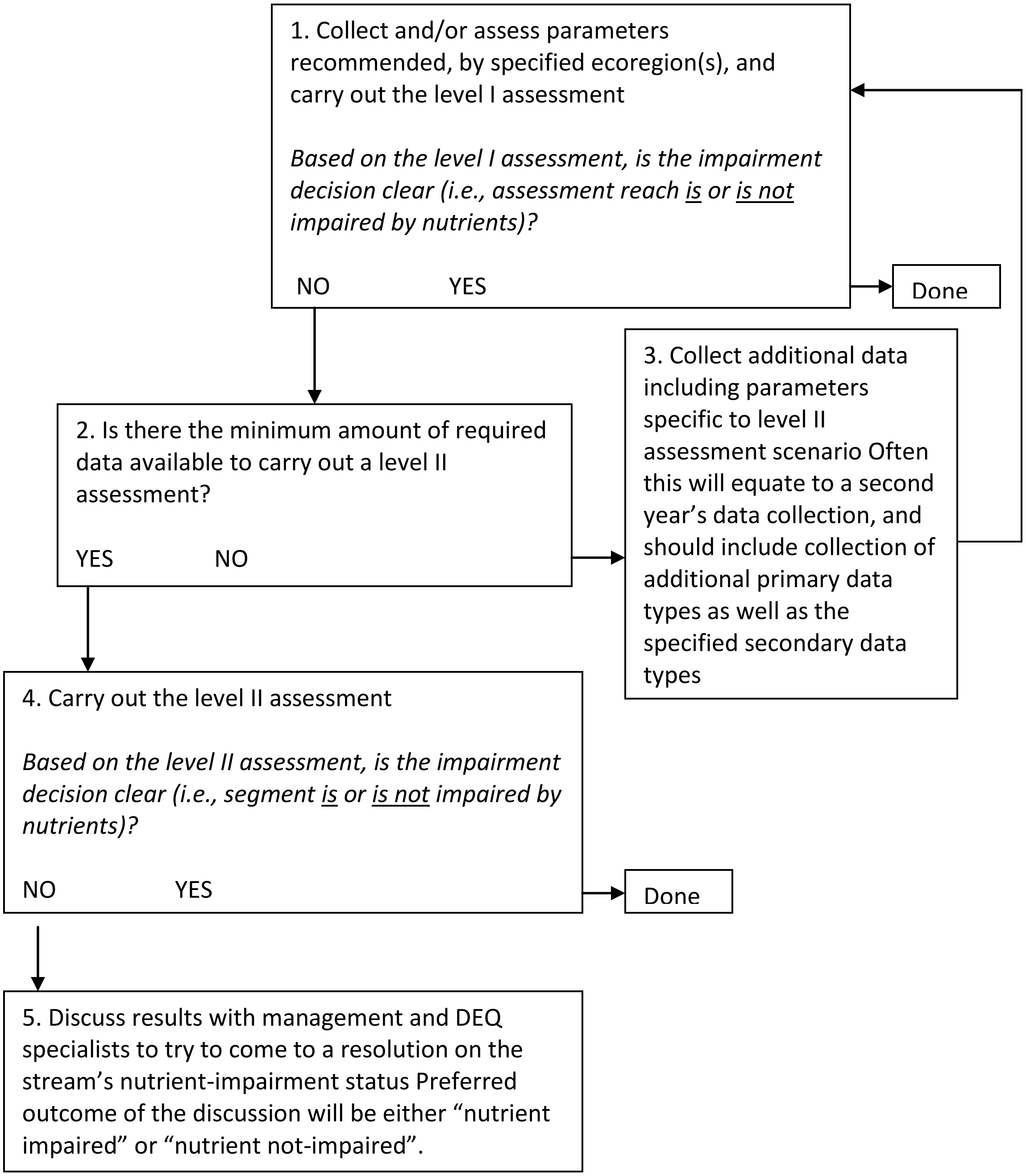

Figure 3-1. Flow-path for decision making using data parameters in Table 3-1.

If the level I assessment leads to an unclear decision, the assessor should then use the data (primary and secondary, if data sufficiency met) to carry out the level II assessment[7]. If a level I assessment is inconclusive and leads to a 2^{nd} year of data collection, always pass the now-larger dataset back through the level I assessment matrix first. It may result that that the conclusion is now clear, without having to go to level II. **NOTE:** Nothing in the approach shown in **Figure 3-1** precludes an assessor from collecting,

[7] The approach shown in **Figure3-1** closely parallels the decision framework of EPA's CALM guidance (U.S. Environmental Protection Agency, 2002); see **Figure 3.2**, page 3-10 of that document.

in a single field season, all data needed to complete a level I and a level II assessment. Situations may arise (e.g., land access issues) where this approach is preferable.

As can be seen, one notable aspect of the approach in **Figure 3-1** is that the data we have labeled "secondary" (**Table 3-1**) are brought into the decision framework only after the primary data have lead to an unclear conclusion. In effect, secondary data are being held to the side until the primary data have been played out to their fullest. The approach attempts to keep data combination scenarios to a minimum and decision making as simple as is reasonable (Occam's razor; "plurality should not be posited without necessity")[8].

The different combinations of results that can occur have been assembled in an Excel spreadsheet (**NtrntAssessFramework.xlsx).** In the spreadsheet, the user identifies the unique combination of results from their assessment reach, and then derives a conclusion. For each combination of results, the spreadsheet provides an outcome (i.e., nutrient impaired, not-nutrient impaired, unclear), and an explanation as to what is likely going on in the stream's ecology. **Different parameter sets are used in different geographic regions, therefore the user must use the tabs for the region applicable to their stream.** Regional tabs are further subdivided to correspond to a level I or level II assessment, per the approach shown in **Figure 3-1**. As an example, three result combinations for the mountain and transitional region are given in **Table 3-2**.

Table 3-2. Three combinations of results, and the conclusions that can be drawn from them, using the parameters listed in Table 3-1.
All three examples apply to streams of the mountain and transitional region of the state, and are from a level I assessment.

Scenario	Nutrient Binomial Test	Nutrient T-test	Benthic Algae	Diatom Increaser Taxa-Probability of Impairment (OPTIONAL)*	Resulting Decision	Further Sampling?
1	PASS	PASS	≤120 mg Chl *a*/m^2 or ≤35 g AFDW/m^2	<51%	Waterbody <u>is not</u> nutrient impaired. All indications show that the stream is in compliance.	No

[8] We did this for two reasons. First, we believe that in most cases some types of data are inherently better for nutrient-enrichment assessment than others, and if the decision can be made using those data alone, the assessment will be simpler and less expensive. Second, it reduces the total number of data-combination outcomes and, in turn, the number of scenario-by-scenario conclusions about impairment that have to be made. To illustrate, for any given data type for which there is a dichotomous outcome (i.e., result is <u>above</u> or <u>below</u> some threshold), the number of possible permutations is 2 raised to the number of data types. Three data types result in (2^3) = 8 possible data-combinations, and one must consider what each unique combination of results is saying about nutrient impairment. Five data types considered together already results in 32 unique combinations, and so on If not all of the additional data are as useful as the previous, it becomes questionable whether the additional work, cost, and complexity are warranted.

Table 3-2. Three combinations of results, and the conclusions that can be drawn from them, using the parameters listed in Table 3-1.
All three examples apply to streams of the mountain and transitional region of the state, and are from a level I assessment.

Scenario	Nutrient Binomial Test	Nutrient T-test	Benthic Algae	Diatom Increaser Taxa-Probability of Impairment (OPTIONAL)*	Resulting Decision	Further Sampling?
12	PASS	FAIL	>120 mg Chl *a*/m^2 or >35 g AFDW/m^2	>51%	Waterbody is nutrient impaired. Non-compliance with the T-test suggests that pulsed nutrient loads are allowing high algae biomass to be maintained via luxury uptake. Diatoms confirm enrichment finding.	No
16	FAIL	FAIL	>120 mg Chl *a*/m^2 or >35 g AFDW/m^2	>51%	Waterbody is nutrient impaired. All indicators show that the stream is not in compliance.	No

*However, if the data minima are available for diatom metric category, they must be used in the decision framework

Subsequent sections will provide detail on which assessment parameters apply where, which statistical tools are to be applied to which parameters, etc. The important point to note here is that the combinations of results you will encounter have been accounted for in the spreadsheet tool (**NtrntAssessFramework.xlsx)**.

Returning to **Figure 3-1**, the "Discuss results with management and DEQ specialists" outcome occurs when the level II assessment has still not resulted in a clear conclusion. This resolution step was suggested by Mark Bostrom (DEQ Bureau Chief) as a way to come to a conclusion without ending up in an endless do-loop. Details of this process remain to be worked out; a potentially useful framework for carrying out the determination has been developed by EPA (Cormier, et al., 2000; Cormier and Suter, II, 2008).

3.2 Details on the Use of Nitrogen and Phosphorus Concentration Data, and Other Measured Parameters, to Support Nutrient-Impairment Assessments

As noted above, different groups of parameters best apply to particular regions of the state. The applicable list of parameters, their impact thresholds, and the delineation of the regions are provided in **Section 4.0** (mountain and transitional streams) and **Section 5.0** (plains streams). In order to maintain temporal independence to the best degree possible, samples collected sequential at a site should be collected about **30 days** after the previous sampling. (There are exceptions to this; see individual parameter list in **Section 3.2.2** below.) Spatial independence of sites within an assessment reach can generally be established by following these guidelines:

- Sites (or short reaches equivalent to sites) should be located a minimum of 1 stream mile apart.

- Sites may be placed < 1 mile apart along the assessment reach if there is a flowing tributary confluencing with the segment between the two sites.
- Try to collect water samples starting at the downstream end of the assessment reach moving upstream, to avoid re-sampling the same water.
- Land use changes and land form changes should be considered and can be used to help define additional sampling sites within an assessment reach.

See **Section A.3** in **Appendix A** for the derivation of these guidelines.

Numeric nutrient criteria apply during summer baseflow[9], also referred to as the growing season. Start and end dates for the growing season vary by ecoregion (Suplee, et al., 2008); see **Table 3-3** below. These dates should be adhered to for collection of the other parameters in **Table 3-1** as well. However, a ten day window (plus/minus) on the Growing Season start and end dates is acceptable, in order to accommodate year-specific conditions (e.g., an early-ending spring runoff). The assessor should use their best professional judgment when deciding if early or later sampling is warranted.

Table 3-3. Start and Ending Dates for Three Seasons (Winter, Runoff and Growing), by Level III Ecoregion

Ecoregion Name	Start of Winter	End of Winter	Start of Runoff	End of Runoff	Start of Growing Season	End of Growing Season
Canadian Rockies	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Northern Rockies	Oct.1	March 31	April 1	June 30	July 1	Sept. 30
Idaho Batholith	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Middle Rockies	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Northwestern Glaciated Plains	Oct.1	March 14	March 15	June 15	June 16	Sept. 30
Northwestern Great Plains	Oct.1	Feb. 29	March 1	June 30	July 1	Sept. 30
Wyoming Basin	Oct.1	April 14	April 15	June 30	July 1	Sept. 30

3.2.1. Nitrogen and Phosphorus Data

The nitrogen and phosphorus criteria are <u>not</u> presented in this document. Readers should refer to Suplee et al. (2008) and its addendums to locate the nutrient concentration values. DEQ anticipates that the nutrient criteria will be adopted by the Board of Environmental Review. After adoption they will be referred to as "base numeric nutrient criteria", and will be housed in a new DEQ circular (DEQ-12). Please use the most updated versions of the criteria in all assessments. Check with Standards on status.

Nutrient (TN, TP) concentration data from an assessment reach are to be assessed collectively, i.e., all nutrient data collected along the reach are to be assessed together, using statistical tests. We recommend two statistical testing procedures to evaluate the nutrient dataset; the Exact Binomial Test and the One-Sample Student's T-test for the Mean. The rationale for using two statistical tests is in **Appendix A**. The tests are in two Excel spreadsheets and their use is described below.

To use the statistical tests, do the following:

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[9] Lakes generally require year-round nitrogen and phosphorus criteria if they are to be protected from cultural eutrophication This may in turn affect the time-of-application of nutrient standards in the near-field tributaries of those lakes.

- For new, un-listed stream segments, use the Excel spreadsheet tool named "MT-NoncomplianceTool.xls".
- For already-listed stream segments, use the Excel spreadsheet tool named "MT-ComplianceTool.xls"
- In both tools, for either test, set alpha to 0.25 (25%)[10]. For the Binomial set the critical exceedance rate to 0.2 (20%) in cell B6. The effect size (gray zone) should be 0.15 (15%) and is set as a function of the exceedance rate. So, in the "MT-NoncomplianceTool.xls" this means p2 should be set to 0.35 (i.e., enter 0.35 into cell B7), and in "MT-ComplianceTool.xls" p2 should be set to 0.05 (enter 0.05 in cell B7).

Both tests (Binomial, T-test) will produce a result (PASS, FAIL). For the Binomial, you need to compare the allowable number of exceedances shown by the test ("**e**s", found in column D) to the actual number of exceedances manifested by your dataset. For the T-test, you will need to enter the dataset into the spreadsheet along with the criterion concentration against which the data are being compared. If the assessment reach complies with a test, the result is PASS; if the assessment reach does not comply with a test, the result it FAIL.

Note: If a non 303(d)-listed nutrient species is the same element as a listed one (e.g., stream is listed for nitrate, but you are also assessing TN, and TN is not currently listed), use the "MT-ComplianceTool.xls" for the non-listed nutrient species as well.

3.2.1.1 Minimum Sample Size for Nitrogen and Phosphorus Data

In the vast majority of cases the assessor will be making the nutrient-impairment decision with a fairly small nutrient-concentration dataset (probably < 13 samples). Statistics derived from small datasets such as these are subject to a fair amount of uncertainty. For example, outcomes from the Binomial Test (compliant, non-compliant) will, for nutrient sample sizes around 13, have confidence levels of about 75% (i.e., alpha and beta error of about 25% each).

For assessment reaches, the target number of nutrient samples is 12 (new, un-listed stream segments) or 13 (already-listed stream segments). The rationale for these sample sizes is presented in **Appendix A**.

HOWEVER: Cases exist where a dataset smaller than 12 or 13 will provide a sufficiently clear result that further nutrient sampling is not warranted. At about 6 samples or less, beta error in the Binomial test can become unacceptably high (> 65%) and increasingly worse with smaller ***n.*** At 7-8 samples, however, there are cases where a certain number of exceedances would be extremely unlikely unless the stream's true exceedance rate was much in excess of 20%. Therefore, for sample sizes of 7, ≥ 4 exceedances can be considered FAIL for the Binomial test. If <4 exceedances are found, sampling should be resumed until the minimum of 12 or 13 is achieved. The T-test can also be used with 7 samples but its power is lower at this sample size. Please see the bullets in **Appendix A**, **Section A.5**.

Also, circumstances may arise where nutrient sampling that is planned over two field seasons may lead to a reduction in the necessary number of samples. For example, if at the end of year one ten (10) TN and TP samples have been collected from an assessment reach on an unlisted stream, and the number of exceedances in each dataset is one (1), it would not be necessary to collect the addition two samples

[10] Alpha, exceedance rate, and the gray zone can be changed via the input cells in the upper left hand corner of the spreadsheet.

(to achieve 12) the following year. This is because even if both of the subsequent samples were above the criteria, the decision outcome (assessment reach “attains”) would not be altered. Assessors should consider these types of situations at the end of each field season in order to best optimize work and cost.

Important Caveat: When the nutrient-concentration dataset is large (as defined below), the nutrient impairment decision should be made using nutrient concentrations alone.

- Large nutrient dataset for already-listed segments: 90 samples in the assessment reach
- Large nutrient dataset for unlisted segments: 50 samples in the assessment reach.

The large sample sizes were determined using the Binomial distribution with an alpha of 0.05 (95% confidence level) and a balance between alpha and beta error (i.e., beta is also about 0.05).

If the large sample sizes listed above are available, the assessor should generally forgo the use of parameters other than total nitrogen and phosphorus (i.e., those in **Table 3-1**) in nutrient-assessment decision making. Nutrient concentrations alone can be used to assess standards compliance, via the Binomial test only.

3.2.2 Minimum Sample Sizes for Other Parameters

The remaining parameters in **Table 3-1** (with the exception of BOD_5; more on it below) are effect variables, i.e. they are affected by changes in nutrient concentrations. Sample size requirements for each parameter are summarized below. Each result, from a sampling event and for a parameter, should normally be considered on its own merits when using the decision spreadsheet (NtrntAssessFramework.xls) and completing the assessment. Exceptions to this exist; see below. The parameters applicable to specific regions (mountain and transitional streams vs. plains streams) and the impact thresholds associated with those parameters are given **Section 4.0** and **Section 5.0**. ***Important Note*: Within their region of application, parameters shown below are required in order to carry out a level I assessment (see Figure 3-1). If a parameter is only required for a level II assessment, this will be indicated.**

Benthic Algal Biomass Samples (Chl *a* and AFDW): At least three (3) sampling events for benthic algal biomass are to be carried out in the assessment reach. These may include approved visual estimation methods. If more than one site is established in the assessment reach, disperse sampling effort across the different sites. Otherwise, assure that about 30 days have passed before sampling again at the same site.

Diatom Samples (Nutrient Increaser Taxa Metric): At least two (2) diatom sampling events are to be carried out in the assessment reach. If more than one site is established in the assessment reach, disperse sampling effort across the different sites. Otherwise, assure that about 30 days have passed before sampling again at the same site. **Note:** Diatom samples are required at level I in the plains, but are only required for a **level II** assessment in the **mountain and transitional region.** However, since there is no validated diatom increaser metrics for the Middle Rockies ecoregion, they are not required collection there.

Macroinvertebrate Samples (HBI Metric): At least two (2) macroinvertebrate sampling events are to be carried out in the assessment reach, unless the site is in the Middle Rockies ecoregion, in which case at least three (3) sampling events are to be carried out. If more than one site is established in the assessment reach, disperse sampling effort across the different sites. **If only one site is present is the assessment reach**, **or if you decide to collect across-time samples at several sites**, collect the across-time samples at the site(s) approximately 30 days apart. The across-time HBI scores from a site should then be averaged prior to comparison to the threshold. If you only have one site, you will only have one HBI value for the assessment reach to compare to the threshold. **Note:** Macroinvertebrate samples are only required for a **level II** assessment, and only in the **mountain and transitional region**.

Sampling to Determine Dissolved Oxygen Delta: At least three (3) DO sampling events (i.e., days) are to be carried out in the assessment reach. If more than one site is established in the assessment reach, disperse sampling effort across the different sites. **DO deltas fluctuate rapidly, and therefore you do not need to wait 30 days to collect subsequent DO data at a site.** For example, collection at a site over 3 contiguous days is acceptable. The more DO deltas that can be collected in the assessment reach, the better.

BOD_5: At least three (3) BOD sampling events are to be carried out in the assessment reach. If more than one site is established in the assessment reach, disperse sampling effort across the different sites. Samples for the standard 5-day biochemical oxygen demand (BOD_5) test are similar to nutrient samples, in that they are a stream-water measurement that can change rapidly, and BOD's affect on DO varies according to other factors (e.g., wind mixing). **Note:** BOD_5 samples are only required for a **level II** assessment, and only in the **plains region**.

Observation Data for Macrophytes and Benthic Algae: The Fish Cover/Other form, i.e. the component of the form pertaining to macrophytes and benthic algal growth, is to be filled out in accompaniment with each benthic algal biomass and/or diatom sampling event. It should also be filled out at each site in the assessment reach at least once each summer. If across a summer growth conditions have notably changed at the site, fill it out again.

3.2.3 Determine the Nutrient Most Likely to be Harming Use(s)

Normally both N and P, and potentially different species of N (e.g., nitrate, TN), will be simultaneously evaluated within an assessment reach. Cases will arise where the harm-to-use signal from one element or the other is clearly stronger, which will help streamline the assessment determination and subsequent work (e.g., TMDL development). An example is provided below.

Table 3-4. Simultaneous Review of Multiple Nutrients and Effect Variables

Assessment Reach	Nutrient	n	Binomial	T-test	Diatom Increaser Taxa	Benthic Chl *a*
Fred Cr reach 1	NO_3+NO_2	14	PASS	PASS	Exceeds criteria	Exceeds criteria
Fred Cr reach 1	TN	14	PASS	PASS	Exceeds criteria	Exceeds criteria
Fred Cr reach 1	TP	14	FAIL	FAIL	Exceeds criteria	Exceeds criteria

Total P results in **Table 3-4**, when run through the assessment process in the "NtrntAssessFramework.xlsx" spreadsheet, result in a clear "nutrient impaired" decision, at level I. This is largely driven by the two FAILS for the statistical tests (i.e., TP concentrations were very elevated). But because each nutrient type is assessed separately and TN is PASS for both statistical tests, the TN outcome is considered unclear and would, as a result, move to level II. As a result, the TN-impairment determination would be driven only by the biotic response variables and not by TN (see scenario 10, 'Mountain and Transitional' tab, in NtrntAssessFramework.xlsx). However, the most succinct conclusion (again, applying Occam's razor) is that the problem in this assessment reach is related to P, not N, as can be seen in **Table 3-4**, and only P should be listed. Arranging and reviewing the data as shown in **Table 3-4** should prevent unnecessary chasing of vague results for a nutrient when clearly the alternate nutrient is the issue.

Cases will arise where both nutrients will give mixed results within the two statistical tests, and therefore neither nutrient is clearly the culprit. In such cases, in accordance with the final outcome of the weight-of-evidence assessment, N and P should probably both be listed as probable causes.

3.2.4 Examples of Nutrient-impairment Decisions

Below are 3 assessment reach examples and their outcomes, to demonstrate the process.

(**1**) The assessment reach is in western MT and has 6 sampling sites. Each site has been sampled 2 times for nutrient concentrations (TN, TP) and once for benthic Chl *a.* **Action*:*** The nutrient samples (n = 12) are assessed, by type (TN or TP), using the two statistical tools, which will result in PASS or FAIL for each test. Both TP tests are FAIL, but the TN tests are both PASS. Each of the six Chl *a* sampling events (each comprised of 11 replicates which have been reduced to a sampling-event average) are independently compared to the criteria. One of them exceeds 120 mg Chl a/m^2, so declare Chl *a* as 'Exceeds Criteria' for the assessment reach. The data suggest a TP problem but not a TN problem, per methods in **Section 3.2.3** above. TP is listed as the cause, and further data collection and assessment for TN is not necessary.

(**2**) The assessment reach is on an eastern MT plains stream. There are 3 sampling sites where nutrients have been sampled 4 times and DO has been continuously monitored by deployed instrument for two summer months, at one site. **Action*:*** The nutrient samples (n = 12) are assessed, by type (TN or TP), using the two statistical tools, resulting in PASS or FAIL for each test. Both TN tests are FAIL, TP tests are mixed (1 PASS, 1 FAIL). Daily DO deltas from the long-term DO dataset should be calculated and compared to the DO delta threshold of 5.3 mg/L. Because this is a long-term dataset, close attention should be paid to the percent of DO deltas exceeding the threshold; if >10%, DO would be declared 'Exceeds Criteria' for the assessment reach[11]. Both TN and TP are suspected (N much more strongly) and should both be listed.

(**3**) The assessment reach is in western MT and the assessment has gone to a 2nd year of data collection. There are four sites. Nutrients have been collected at the 4 sites three times, benthic Chl *a*/AFDW once at each site, macroinvertebrate samples have been collected at 3 sites once each, and twice at the 4th site, and diatom samples have been collected at all 4 sites two times each. **Action*:*** All data from both years are first routed through the level I decision framework. The nutrient samples (n = 12) are assessed, by type (TN or TP), using the two statistical tools, which will result in PASS or FAIL for each test. Both TP are FAIL, and both TN tests are PASS Each of the four Chl *a* and AFDW sampling events (each comprised of 11 replicates which have been reduced to a sampling-event average) are independently compared to the criteria. One of them exceeds 35 g/m^2 AFDW, which is sufficient to declare Chl *a*/AFDW as 'Exceeds Criteria' for the assessment reach. Each macroinvertebrate HBI metric score where there is only one observation per site (three sites) is considered independently. For the 4th site, the two temporally-

[11] If DO deltas >5.3 mg/L comprise < 10% of the dataset, consider if the site has a strong presence of macrophytes or not. If macrophytes are very common, the site could be declared as 'Meets Criteria'. If macrophytes are not common, it could be declared 'Does Not Meet Criteria'. Consult Standards Section for further assistance.

collected HBI scores are averaged. One of the preceding macroinvertebrate HBI scores is >4, thus 'Exceeds Criteria' would be declared for the macroinvertebrate category for the assessment reach If the diatom increaser taxa scores (all 8) were each <50% probability of impairment, then the diatom metric score would be declared as 'Meets Criteria' for the assessment reach. The assessment reach is found to be impaired for TP at level I without having to use the macroinvertebrate results. For TN, the assessment could move to level II assessment and this would show nutrient impairment; but that outcome is driven purely by biometrics, which are sensitive to both nutrients. The overall dataset, per methods in **Section 3.2.3** above, suggests a TP problem but not a TN problem. TP is listed, and further data collection and assessment for TN is not needed.

3.2.5 Overwhelming Evidence of Nutrient Impairment-All Regions

Some circumstances related to excess nutrient pollution are severe enough that a rigorous data collection effort is not required. Photo documentation will suffice. Below are listed conditions that can be considered overwhelming evidence; these apply equally to wadeable streams across the state. These conditions are likely to be intertwined with organic pollution problems.

- Fish kills involving massive growths of senescing algae mats. These mats may be attached to the bottom or floating. Dissolved oxygen levels at dawn will likely be less than 1 mg/L.

- Filamentous algal growth covering the entire bottom from bank to bank and extending continuously for a substantial longitudinal distance (> 150m). Use the photographs below (**Figure 3-2** and **3-3**) as a guide. Don't confuse these conditions with sporadic, longitudinally-patchy growths of heavy filamentous growth, in between which there is lighter algal growth. The latter are not extreme enough to warrant overwhelming evidence, and should be sampled/assessed per the method earlier described.

Figure 3-2. Photographs of heavy, bank-to-bank and longitudinally continuous Cladophora growth Left photo is from (Sandgren, et al., 2004).

Photo courtesy Dr. Vicki Watson.
Figure 3-3. Massive *Cladophora* growth in the Clark Fork River, MT, 1984.
This nuisance alga is aptly named "blanket weed".

4.0 Nutrient Impairment Assessment Methodologies: Wadeable Streams in the Mountain and Transitional Region

The following subsections describe assessment methods best suited for use in mountainous streams and streams that transition between mountains and plains. Analysis shows that level IV and level III ecoregions are the most useful classification tool for defining nutrient zones (Varghese and Cleland, 2005), and nutrient criteria have been developed using ecoregions as the base zoning system (Suplee *et al.*, 2008). Consideration has also been given to the legal classification system for streams (B-1, C-3, etc.) which defines the streams' designated beneficial uses. There is a very high degree of correspondence between streams with salmonid fish among their beneficial uses (A-closed, A-1, B-1, B-2, C-1, C-2) and certain groups of ecoregions. Specifically, the mountainous level-III ecoregions (15, 16, 17, and 41) plus specified level-IV ecoregions along the Rocky Mountain front — Level IVs that are subunits of the level-III Northwestern Glaciated Plains (42) and Northwestern Great Plains (43) ecoregions — comprise a group well suited for assessment methodologies presented in this section. Four (4) additional level IV ecoregions (42l, 42n, 43o, 43t) that were not presented in Suplee *et al.* (2008) have been added to the group. These four level IV ecoregions are also transitional along the Rocky Mountain Front and comprise regions in which all or virtually all waterbodies are classified as supporting salmonid fishes among their beneficial uses. The regions are shown in **Table 4-1**.

Table 4-1. Ecoregions (levels III and IV) in which Assessment Methods in this Section Best Apply

Ecoregion Scale	Ecoregion Name	Ecoregion Number
Level III	Northern Rockies	15
Level III	Idaho Batholith	16
Level III	Middle Rockies	17
Level III	Canadian Rockies	41
Level IV	Sweetgrass Uplands	42l
Level IV	Milk River Pothole Upland	42n

Table 4-1. Ecoregions (levels III and IV) in which Assessment Methods in this Section Best Apply

Ecoregion Scale	Ecoregion Name	Ecoregion Number
Level IV	Rocky Mountain Front Foothill Potholes	42q
Level IV	Foothill Grassland	42r
Level IV	Unglaciated Montana High Plains	43o
Level IV	Non-calcareous Foothill Grassland	43s
Level IV	Shields-Smith Valleys	43t
Level IV	Limy Foothill Grassland	43u
Level IV	Pryor-Bighorn Foothills	43v

Note: The level IV ecoregion "Unglaciated Montana High Plains" (43o) has more than one polygon in Montana. Only the polygon located just south of Great Falls, MT should be considered part of the transitional streams group. Also, the level IV ecoregion "Foothill Grassland" (42r) has polygons associated with both the Middle Rockies *and* Canadian Rockies level III ecoregions. 42r polygons are associated with the level III ecoregion (either Middle Rockies or Canadian Rockies) against which they abut.

4.1. Assessment of Benthic Algal Growth

For wadeable streams, we recommend that site-average benthic algae densities of 120 mg Chl a/m^2 and 35 g AFDW/m^2 be used as thresholds (i.e., maximum allowable levels) to prevent impact to the fish and associated aquatic life uses (i.e., to maintain DO standards in DEQ-7), and the recreation use (ARM 17.30.637(1)(e)). Details on how these values were derived are in **Appendix B**.

Note: AFDW results from core samples should never be included in determining a site's average AFDW. The method measures organic material from the entire core sample, not just the surface where the algae are growing, and will therefore over-report AFDW.

Each sampling event result should be considered on its own merits when using the decision spreadsheet (NtrntAssessFramework.xlsx) and completing the assessment. That is, if 3 sampling events for benthic alga growth were undertaken and 1 of the Chl *a* averages exceeds the recommended threshold, then the conclusion for the assessment reach for the benthic algae category would be "exceeds 120 mg Chl a/m^2".

4.2 Assessment Using Biometrics

Biometrics based on diatom algae are stressor-specific (e.g., address nutrient pollution) and apply to specific regions. A diatom sample that indicates >51% probability of impairment by nutrients indicates the sample is from a site manifesting an excess nutrient problem. Details on how the diatom biometrics were developed and the thresholds derived are presented in the periphyton SOP (Montana Department of Environmental Quality, 2011b).

Always consider cautiously the results from samples collected very early and very late in the sampling season. Algae are a successional community, and if you sample too early, you will sample fewer 'pioneer' species and too late, you will start seeing the community as a whole die off - some taxa sooner than others. These changes can affect metric results.

Various biometrics based on macroinvertebrates were reviewed. We selected the Hilsenhoff Biotic Index (HBI) as the best tool for assessing nitrogen and phosphorus pollution problems. An HBI score of 4.0 should be used as the threshold (i.e., maximum allowable value) to prevent impact to fish and associated aquatic life uses. Details on how the biometrics were selected and the thresholds derived are presented in **Appendix B**.

Each sampling event result for a biometric should be considered on its own merits when using the decision spreadsheet (NtrntAssessFramework.xlsx) and completing the assessment. That is, if 2 sampling events for macroinvertebrates were undertaken and 1 of the results was an HBI score of 5.0, then the conclusion for the assessment reach for the macroinvertebrate category would be ">4.0" (i.e., exceeds).

5.0 Nutrient Impairment Assessment Methodologies: Wadeable Streams in the Plains Region

Table 5.1 below shows areas of the state in which the methods of this section best apply. Essentially, the methods apply to all of ecoregion 42 (Northwestern Glaciated Plains) and ecoregion 43 (Northwestern Great Plains) *except for* the level IV ecoregions along the Rocky Mountain Front which are being lumped with the mountainous ecoregions (see **Table 4-1**).

Table 5-1. Ecoregions (level III) in which Assessment Methods in this Section Best Apply
Some level IV ecoregions associated with the level IIIs shown are excluded; these are listed below each level III.

Ecoregion Scale	Ecoregion Name	Ecoregion Number
Level III	Northwestern Glaciated Plains	42
Level IV ecoregions of the Northwestern Glaciated Plains not in the Warm Water Fishery Class:		
Level IV	Sweetgrass Uplands	42l
Level IV	Milk River Pothole Upland	42n
Level IV	Rocky Mountain Front Foothill Potholes	42q
Level IV	Foothill Grassland	42r
Level III	Northwestern Great Plains	43
Level IV ecoregions of the Northwestern Great Plains not in the Warm Water Fishery Class:		
Level IV	Unglaciated Montana High Plains	43o
Level IV	Non-calcareous Foothill Grassland	43s
Level IV	Shields-Smith Valleys	43t
Level IV	Limy Foothill Grassland	43u
Level IV	Pryor-Bighorn Foothills	43v

Note: The level IV ecoregion "Unglaciated Montana High Plains" (43o) has more than one polygon in Montana. Only the polygon located just south of Great Falls, MT is excluded from the Warm Water Fishery Class.

5.1 Assessment Using Biometrics

Biometrics based on diatom algae are stressor-specific (e.g., address nutrient pollution) and apply to specific regions. A diatom sample that indicates **>51%** probability of impairment by nutrients indicates the sample is from a site manifesting a nutrient problem. Details on how the diatom biometrics were developed are presented in the periphyton SOP (Montana Department of Environmental Quality, 2011b).

Each biometric sampling event result should be considered on its own merits when using the decision spreadsheet (NtrntAssessFramework.xlsx) and completing the assessment. That is, if 2 sampling events for diatoms were undertaken and 1 of the results was “65% probability of impairment by nutrients”, then the conclusion for the assessment reach for the diatom category would be “>51%,” (i.e., exceeds).

Note: Diatom biometrics for the plains region have an inherently high false negative rate (62%; i.e., the chance that the metric declares a truly nutrient-impacted site as having no nutrient impact). This fact is given consideration, in that the resulting decisions in the spreadsheet (NtrntAssessFramework.xlsx) lean somewhat to the protective side.

Always consider cautiously the results from samples collected very early and very late in the sampling season. Algae are a successional community, and if you sample too early, you will sample fewer 'pioneer' species and too late, you will start seeing the community as a whole die off - some taxa sooner than others. These changes can affect metric results.

5.2 Assessment Using the Difference Between the Daily Maximum Dissolved Oxygen Concentration and the Daily Minimum Dissolved Oxygen Concentration (Delta)

We recommend that the magnitude of the daily DO concentration change (daily maximum minus the daily minimum, or *delta*) be used to assess plains streams. Elevated daily DO delta values indicate high productivity and the potential for DO standards exceedances (per DEQ-7) that would impact fish and aquatic life. We suggest that a DO delta of 5.3 mg/L be used as the threshold. **Assessors need not wait 30 days to take subsequent DO measurements at a site; each DO sampling event may be considered on its own merits.** Details on how the DO threshold was identified are provided in **Appendix C**.

Each DO sampling event result should be considered on its own merits when using the decision spreadsheet (NtrntAssessFramework.xlsx) and completing the assessment. That is, if 5 sampling events for DO delta were undertaken and 1 of the DO deltas exceeds the recommended threshold, then the conclusion for the assessment reach for the DO delta category would be “exceeds 5.3 mg/L”. Further consideration may be needed if the data were collected long-term[12]. **Note:** Do deltas in the plains region have an inherently high false negative rate (63%; i.e., the chance that the DO deltas indicate that a truly

[12] If DO deltas >5.3 mg/L comprise < 10% of the dataset collected using an instrument deployed at least 14 days, consider if the site has a strong presence of macrophytes or not. If macrophytes are very common, the site could be declared as ‘Meets Criteria’. If macrophytes are not common, it could be declared ‘Does Not Meet Criteria’. Consult Standards Section for further assistance.

nutrient-impacted site has no nutrient impact). This fact is given consideration in that the resulting decisions in the spreadsheet (NtrntAssessFramework.xlsx) lean somewhat to the protective side.

5.2.1 Deploying a Continuous DO Monitoring Device in Wadeable Plains Streams

If a continuous monitoring device is to be deployed (e.g., YSI 6600 sonde), we recommend that at least one (1) full day of data be collected to properly calculate a daily DO delta. The day of deployment and the day of retrieval are usually truncated, and so at least one full day in between assures the necessary data are collected. We recommend a fifteen minute time step for monitoring, as that has worked well in our experience and provides good data resolution. Initial calibration, as well as drift from calibration—which is determined at the time the unit is retrieved— should be documented per the project's QAPP and or SAP.

5.2.2 Instantaneous DO Monitoring in Wadeable Plains Streams

Daily DO minimum and maximum concentrations each need to be obtained, and can be collected using a hand-held instrument. The daily DO minimum needs to be collected starting in the pre-dawn hours, up to as late as 8:00 am. The daily DO maximum will *usually* occur between 2:30 pm and 5:00 pm; observations can be collected every 15-30 minutes during this time to identify the highest value. Continue monitoring after 5:00 pm if observations are still climbing. Further details on how these time frames were identified are provided in **Appendix C**.

The YSI 85 instrument has a 50 reading, manual-entry memory which can be used for collecting DO maximums. With the unit on and the sensor properly deployed, depress the ENTER button for two seconds to record an instantaneous observation. Data may be downloaded later.

For the purpose of calculating DO delta, at least 3 DO sampling events (i.e., 3 different days) should be taken in each assessment reach; if collected at the same site, they do <u>not</u> need to be collected 30 days apart (e.g., 3 days in a row is OK).

5.2.3 BOD_5

We recommend that biochemical oxygen demand, or BOD_5 (also called just BOD), be used to assess plains streams at level II. We recommend a threshold of 8.0 mg/L be used. Each BOD sampling event result should be considered on its own merits when using the decision spreadsheet (NtrntAssessFramework.xlsx) and completing the assessment. That is, if 3 sampling events for BOD were undertaken and 1 of the BODs exceeds the recommended threshold, then the conclusion for the assessment reach for the BOD category would be ">8.0 mg/L" (exceeds).

5.2.4 Algal and Macrophyte Indicators of Nutrient Enrichment in Wadeable Plains Streams

The Fish Cover/other form (see periphyton SOP, (Montana Department of Environmental Quality, 2011b)) should be filled out when assessing plains streams. Although not required to fill out the form, we recommend that the dominant macrophytes be identified, which will help with your assessment back in the office using the information below.

In the Northwester Glaciated Plains ecoregion, streambed cover by filamentous algae should generally be less than 30% for a single sampling event and less than 25% for the summertime average (Suplee, 2004). These data can be collected visually using the Fish Cover/Other form, which is provided in the periphyton SOP (Montana Department of Environmental Quality, 2011b). (Although a somewhat tangential issue, the presence of a healthy and widely distributed macrophyte community should be taken as indicative that the stream has a reasonable level of morphologic stability; stream instability has been found to be a major factor in controlling algae and macrophyte dynamics in prairie streams [Suplee, 2004]).

Throughout the plains region, attention should be paid to the types of macrophyte species present. We have observed that northern watermilfoil (simultaneously known as *Myriophyllum exalbescens, Myriophyllum sibiricum, and Myriophyllum spicatum L. var exalbescens* (Muenscher, 1944). DiTomaso and Healy (2003) is extremely common throughout the Northwestern Glaciated Plains and Northwestern Great Plains ecoregions, as has been observed by others (Klarich, 1982). However, in stream sites where high nutrient enrichment is occurring, we have observed northern watermilfoil's (and other macrophyte's) near-complete replacement by coontail (aka hornwort), *Ceratophyllum demersum*[13] Coontail is a rootless, free floating macrophyte—though it can anchor itself to bottom substrates via specialized buried stems—that can proliferate in streams which are being heavily loaded with nutrients (DiTomaso and Healy, 2003). In this it is similar to floating and benthic algae in that it relies on water-column nutrients for growth, because it does not take up nutrients from the sediment via roots, as other macrophytes do. Choking mats of coontail, or its presence to the exclusion of other macrophytes, should be taken as a strong indicator of nutrient over enrichment. Close-up and panoramic photos should be taken to record the extent of the problem, and aide identification of the plants in-office.

Finally, we documented during the Box Elder Creek dosing study that *Chara* spp. (commonly called stonewort or muskgrass) were greatly depressed in number in the nutrient-dosed reaches compared to the Control reach, and also compared to the pre-dosing period. *Chara* spp. are a branched form of algae, are an important component of natural aquatic ecosystems (DiTomaso and Healy, 2003), and are often associated with clean water.

6.0 Acknowledgements

We would like to thank Dr. Vicki Watson and her students for the many years of work she has carried out in support of DEQ projects. These projects (e.g., the reference streams project) have provided the basic data required to develop many of the guidelines in this document. We would also like to thank all of the DEQ staff and field crews who, over the years, have collected mounds of data that also support this work. Thanks to the Carter County Conservation District for the cooperation and support on the nutrient dosing study. Finally, we want to thank the contractors whose excellent work helped build the foundation on which this document stands: Arun Varghese and Josh Cleland (ICF International); Dr. Loren Bahls (*Hannaea*); Mark Teply (Cramer Fish Sciences); Wease Bollman (*Rhithron* Associates); and Dr. Jeroen Gerritsen and Dr. Lei Zheng (Tetra Tech).

[13] Coontail and watermilfoil can readily be distinguished in the field with a good macrophyte guidebook and a hand-held magnifying glass.

APPENDIX A. STATISTICAL CONSIDERATIONS

A.1.0 INTRODUCTION

The numeric nutrient criteria addressed in this appendix are not intended to be ideal standards, i.e., "no sample shall exceed" values. As such, appropriate inferential statistical tests, assumptions about stream sampling frames, etc. must be developed so that the criteria can be correctly applied. This appendix outlines these statistical considerations and provides rationales for the various approaches used. It also provides precautionary points where certain assumptions depart from more conservative statistical thinking, and discusses how improper sampling design has the potential to mislead a conclusion made about a stream's condition. The key issues addressed herein are:

- Sampling frames, populations, and sampling units for streams, and associated assumptions and precautions
- Consideration of what constitutes our best description of sample independence in streams (spatial and temporal), and associated assumptions and precautions
- Determination of appropriate critical exceedance rates for nutrients (nitrogen and phosphorus)
- Statistical testing procedures and accompanying decision rules
- Minimum sample sizes

A.2.0 SAMPLE FRAME, POPULATION, AND SAMPLING UNITS

All studies involving statistical evaluations of data require that a sample frame, population, and sampling unit be defined. Streams are particularly poor entities for establishing these parameters because streams are an interconnected network rather than discreet entities. Nevertheless, streams are the entities to be sampled so some effort must be made to segregate them into definable units. For the purposes of determining compliance with numeric nutrient criteria, we define the following:

- **Sample Frame:** A wadeable[14] stream segment listed in the Assessment Data Base (ADB) (DEQ 2009, and updates) OR a sub-segment of an ADB stream segment. These segments are referred to here as an "assessment reach".
- **Population:** All the water flowing through the assessment reach during the time period when the numeric nutrient criteria apply, and the surface area of the stream bottom over which the water flows.

[14] Wadeable streams are perennial and intermittent streams in which large portions of the channel are wadeable during baseflow conditions. For the list of waterbody segments not considered wadeable (i.e., the large rivers), see Flynn and Suplee (2010).

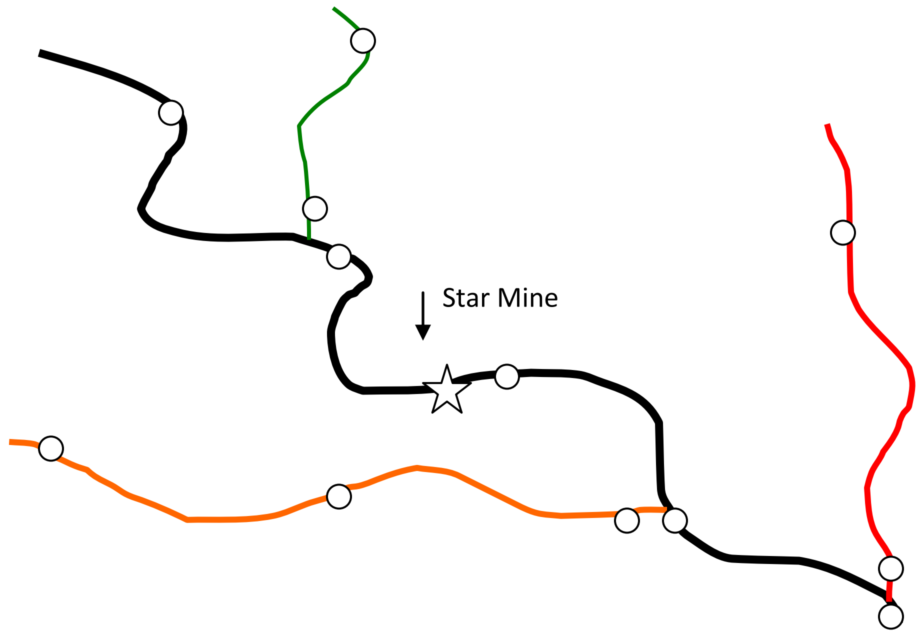
- **<u>Sampling Unit:</u>** A sample collected from the assessment reach that is largely independent of other samples collected within the assessment reach and collected during the time when the numeric nutrient criteria apply.

Assumptions: Each assessment reach (ADB segment or sub-segment) will be made up of a series of sampling sites, or a series of very short study reaches that are essentially sites (**Figure A2-1**). The minimum number of sites on an assessment reach is provided in DEQ SOPs (Montana Department of Environmental Quality, 2005). **Figure A2-1** illustrates the variety of ADB segments that may be found; segment lengths can vary tremendously. For purposes of determining compliance with numeric nutrient standards using statistical methods, it is usually assumed that (1) pollution sources are evenly dispersed along the reach, (2) sampling sites are randomly located along the reach, and (3) each sample is independent of the others (Spatial and temporal independence guidelines for sites are addressed in **Section A.3** below).

It some cases, ADB segments may have pollution problems (hotspots) concentrated only in a particular part of the stream, say, the last 5 stream miles. In such cases, it may not make sense to view the original ADB segment as the best possible sampling frame. That is, it would be better to further stratify the sample frame and, thus, the population of interest .This will prevent distortion of results caused by mixing together, for common analysis, data from the relatively un-impacted sub-segment with data from the impacted sub-segment. For example, in **Figure A2-1** it might be prudent to consider the sub-segment upstream of the Star Mine as a sampling frame apart from the sub-segment below the mine. Stratification is common in studies employing purely random sampling, where it is referred to as stratified random sampling (Cochran, 1977). Stratification allows maximal precision of estimates for minimal sampling effort (Norris, et al., 1992). **The assessor carrying out the analysis on an ADB segment will have to judge if further stratification is warranted.** If it is warranted, then sampling requirements, described above and further detailed below, would apply to *each* of the new sub-segments (aka assessment reaches), individually.

Precautionary Considerations: Pollution sources are rarely evenly-dispersed along stream segments, violating assumption 1 above. And purely random sampling is usually not practical due to stream access issues, etc. Targeting only the known or potential pollution “hotspots” — even within an assessment reach that has been broken out from a larger ADB segment — could over represent the hotspots and distort the statistical tests. Sampling and analysis plans (SAPs) should proceed with goal-oriented sampling (U.S.Environmental Protection Agency, 2000) that works towards striking an equitable balance between the number of hotspot sites and the number of un- or minimally impacted sites *within the defined assessment reach.* That is, the aggregate of collected samples should be representative (U.S. Environmental Protection Agency, 2002) of the assessment reach as a whole. Advanced knowledge and expertise of the field will be needed to accomplish this (Norris, et al., 1992), and modifications to the assessment reach boundaries can be made on-the-fly during field work, if deemed necessary. It is possible to sub-segment a stream reach to the point where, for a particular assessment reach, there really is little left but hotspots; if this is the case, then the hotspots *are* representative of the assessment reach. As a general rule, it is better to lump than split to avoid unnecessary sampling and administrative work. **The requirement to create reasonably uniform assessment reaches is inherently in conflict with the need to “lump” for the purpose of keeping assessments as simple as possible. Judgment is needed to balance these two opposed factors and come up with an optimal sampling strategy.**

Although this quasi-systematic approach is not a substitute for truly random sampling it will, if carried out properly, achieve good sample interspersion and representativeness. For further discussion of randomization *vs.* interspersion approaches, see page 196 of Hurlbert (1984).



Example sampling sites (hollow dots) are shown along each segment.

Figure A2-1. Four different stream reaches (shown by different colors), each representing 1 sampling frame (ADB stream segment)

A.3.0 Determining Sample Independence

According to definitions in Hurlbert (1984), much sampling carried out by DEQ on individual streams tends to violate spatial and temporal independence assumptions and results in pseudoreplication. For example, samples collected over time at a site can be serially correlated, which precludes temporal independence (Hurlbert, 1984). However, the statistical views advocated by Hurlbert are not universally supported; contrary opinions on the matter can be found in the literature (Stewart-Oaten and Murdock, 1986; Stewart-Oaten, et al., 1992; Osenberg, et al., 1994) and have led to what one journal referred to as a “healthy debate” (*Ecological Applications*, volume 4, No. 1, 1994). In general, more needs to be known about detection of non-independence and the frequency with which temporally independent samples can be collected (Underwood, 1994).

Time-series collected samples from a site may be used in inferential statistical testing, if used cautiously; this requires that one assumes that actual trends in time are identical in magnitude and direction for all the sites across the study (Norris and Georges, 1993). Osenberg *et al.* (1994) examine time-series serial correlation of physical and biological measurements in a BACI (Before-After-Control-Impact) study and conclude that, in the marine environment they study, sampling can occur at a site every 60 days without yielding substantial serial correlation.

DEQ recognizes the issue of temporal pseudoreplication, but also needs to be practical about the reality of sampling streams which, by their very nature, make collection of independent samples difficult In DEQ's reference project (Suplee, et al., 2005), 30 days has generally been used as a minimum time span between sampling events at a site to infer temporal independence of water samples. This time span was based on the experiential observation that, during the brief Montana summer, substantial changes in flow, temperature, and vegetation (both riparian and instream) occur from month to month, changes that would likely effect water quality. But Stewart-Oaten *et al.* (1986) recommend that the assumption of temporal independence be tested, rather than assumed. The Durbin-Watson test statistic is widely used to check for time-series serial correlation. Stream sites with monthly nutrient sampling during the summer were available in Montana, and some of these sites were tested using the Durbin-Watson statistic Results are shown in **Table A3-1** below.

Table A3-1. Durbin-Watson Values for Time-series Collected Nutrient Samples at Selected Sites

All Samples were Collected Approximately 30 Days apart Nutrients Showing Probable Time-series Serial Correlation (95% Confidence Level) are Highlighted.						
					Nutrient	
Stream Site	**Months Sampled**	**Time Range**	**n**	**Total N**	**Total P**	**NO_{2+3}**
Rock Creek Site 2	June, July, Aug, Sept	2001-2004	12	1.18	1.43	2.3
Clark Fork R. at Deer Lodge (site 9)	July, Aug, Sept	1998-2006	25	1.81	1.78	1.68
Clark Fork R. above Little Blackfoot R. (site 10)	July, Aug, Sept	1998-2006	26	2.01	1.57	1.46
Clark Fork R. above Flathead R. (site 25)	July, Aug, Sept	1998-2006	26	1.76	1.21	2.08

In general, Durbin-Watson values around 2 mean there is no serial correlation, whereas values greater than approximately 2.5 or less than about 1.5 lead one to suspect negative or positive serial correlation, respectively (Neter, et al., 1989; Ott, 1993). What can be concluded from this limited analysis? Most nutrients did not show serial correlation, and one of the three that did is borderline cases (statistic =1.43, but power of test very low). Overall, it appears that serial correlation is present in nutrient samples collected a month apart, but the effect is very weak It is evident that 30-day separated water samples can provide a fairly high degree of independence for nutrients.

DEQ is aware that spatial independence is also a concern. Water flows from upstream to downstream, consequently influencing the spatial independence of downstream sampling sites. No generally applicable spatial minimums were found as of this writing. U.S.EPA guidance (USEPA, 2002) generally glosses over the topic of spatial independence in streams.

To address spatial independence, we tested a Montana dataset. We used the pre-dosing baseline data collected as part of the Box Elder Creek nutrient dosing study (Montana Department of Environmental Quality, 2010a). We found that total nutrient samples collected within hours of one another at two sites located 0.73 stream miles apart were not spatially correlated. We compared nutrient samples collected from the Low Dose site to those collected on the same day at the High Dose site which is 0.73 miles downstream. Box Elder Creek is perennial and was flowing during all sampling events. No tributary intervenes between the sites. Samples were collected within 1-2 hours of one another, during the summer index period. We only considered samples collected *prior to* nutrient dosing, as these are comparable to what one would encounter during routine stream sampling/assessment. Using the Rank von Neumann test (U.S.Environmental Protection Agency, 2006), we found that there was no serial correlation for total N or total P (i.e., we could not reject the null hypothesis "no serial correlation"), at an alpha of 0.05. There was serial correlation for Soluble Reactive Phosphate (SRP). We were unable to assess soluble N as there were too many non-detects in the datasets, which led to too many rank-ties; too many rank-ties precludes proper statistical evaluation (Gilbert, 1987).

Spatial independence can therefore be established (albeit as rules of thumb) for total nutrients as a minimum of about 1 mile between two sites. Other factors leading to spatial independence include a tributary confluencing on a stream between two sampling sites, or if major land form or land use changes occur along the reach (Montana Department of Environmental Quality, 2007; Montana Department of Environmental Quality, 2011a).

Giving consideration to our findings, below are guidelines for establishing independence of samples collected within an assessment reach:

- Sites (or short reaches equivalent to sites) should be located a minimum of 1 stream mile apart.
- Sites may be placed < 1 mile apart along the assessment reach if there is a flowing tributary confluencing with the segment between the two sites.
- Try to collect water samples starting at the downstream end of the assessment reach moving upstream, to avoid re-sampling the same water.
- Land use changes and land form changes should be considered and can be used to help define (1) breaks between assessment reaches and/or (2) additional sampling sites within an assessment reach.
- Samples collected at the same site (or short reach) should be collected 30 days apart.

Total nutrient samples that meet the above conditions may generally be considered spatially and temporally independent for the purposes of determining compliance with the nutrient criteria. As such, they may be used in inferential statistical analyses and to make conclusions about the assessment reach in question.

Precautionary Considerations: The last bullet above (temporal independence resulting from approximate 30-day time spans) is not applicable for some bioassemblage samples (e.g., macroinvertebrates, fish). These organism populations operate on different (longer) time scales from water samples and diatoms and may show considerable year-to-year stability. Please see **Section 9.0** of Suplee (2004)) and Bramblett *et al.* (2005) for more details on temporal patterns of these biological assemblages. Diatom populations tend to shift quickly, within 1-5 weeks, in response to environmental changes (LaVoie, et al., 2008). Thus, this rate of change is sufficient to be able to consider diatom sampling events spaced 30 days apart as being largely independent of one another.

A.4.0 Selection of Inferential Statistical Tests, Confidence Levels, and Associated Decision Rules

A.4.1 Rationale for Using Two Inferential Statistical Tests to Help Determine Compliance with Nutrient Standards

Exhaustive reviews of the pros and cons of statistical tests available for determining compliance with numeric standards have already been published (U.S. Environmental Protection Agency, 2002). For brevity, rather than revisit all the detailed considerations put forward in those documents, recommendations are provided herein concerning what where judged to be the most applicable tests. These recommendations are then followed by a series of decision rules that allow the user to apply the tests in tandem. For purposes of compliance with numeric nutrient criteria, two tests should be used; the Exact Binomial Test and the One-Sample Student's t-test for the Mean.

- Exact Binomial Test: This test assumes data are dichotomous in nature (i.e., only two possible outcomes). For compliance with a criterion this reduces to (1) samples that exceed the criterion and (2) samples that do not exceed the criterion. If confidence levels, power, and exceedance rates (more on these below) are established upfront, minimum sample sizes can also be determined. The main disadvantage of the test is that it is blind to exceedance magnitude; that is, it takes no account of whether a sample exceeds the criterion by 1% or 1,000%.

- One-Sample Student's t-test for the Mean: This test does not assume the data take on a dichotomous relationship relative to the criterion. The test compares the mean of the samples in question to the criterion. The desired confidence levels in the test are established upfront. But unlike the Exact Binomial Test, it *is* greatly influenced by high values and outliers which can skew the dataset mean relative to the bulk of the other samples in the dataset. It is also influenced by the proportion of non-detects in the dataset[15].

The Exact Binomial Test is useful for determining sample sizes, and is not influenced by large numbers of non-detects in the dataset. In fact, if the magnitude of nutrient criterion exceedances was irrelevant, then the Exact Binomial Test could probably be used by itself. But this is not the case; one must consider the issue of luxury nutrient uptake by algae.

One of the main purposes of establishing nutrient criteria is to control excess algae growth and its effects on water quality. Many benthic and water-column algae have the ability to take up the non-limiting nutrient, be that N or P, in excess of immediate need and utilize it for growth later (luxury nutrient uptake; (Elrifi and Turpin, 1985; Portielje and Lijklema, 1994; Stevenson and Stoermer, 1982). If extracellular nutrient concentrations then decline in the water, growth can still be maintained on intracellular stores (Droop, 1973; Rhee, 1973). Therefore, pulsed loading events of nutrients to streams may allow algae to carry out luxury nutrient uptake which can sustain growth for several cell generations well after the pulse has ended.

[15] For the purposes of using the t-test, users should initially convert all non-detects to 50% of the reported detection limit (USEPA, 2006). If >> than 15% of the dataset will be affected, consult Standards Section.

Luxury nutrients uptake is a kinetics phenomenon dependent on the physiological condition of the algae, duration and magnitude of the nutrient pulse, etc.; complex factors not easily addressed by a simple t-test. But the t-test can help assess the *potential* for luxury nutrient uptake because pulsed loads of elevated nutrient concentrations, if captured during sampling, would increase the dataset mean and would show that mean water quality has exceeded the criterion; this is useful information not provided by the Exact Binomial Test.

Each test possesses strengths the other does not. Therefore, we recommended that the t-test be used in tandem with the Exact Binomial Test via a series of decision rules (**Section 3.0,** main document).

A.4.2 Form of the Null Hypothesis, Alpha, Beta, Effect Size, and Critical Exceedance Rate

All of the factors listed in **Section A.4.2's** title are interrelated and influence one another in statistical hypothesis testing. Again, rather than reiterate here the mass of discussion devoted to these topics already covered elsewhere ((U.S. Environmental Protection Agency, 2002; California Environmental Protection Agency State Water Resources Control Board, 2004), we will simply state what we concluded to be the best statistical parameters (form of null hypothesis, alpha, beta, etc.) associated with the two tests (Exact Binomial and One-Sample Student's t-test for the Mean), and provide further explanation where warranted.

A.4.2.1 Form of the Null Hypothesis for the Statistical Tests

For Streams Already on the 303(d) List:

- Null Hypothesis: Waterbody is not in compliance with numeric nutrient standards
- Alternative Hypothesis: Waterbody is in compliance with numeric nutrient standards

For Streams Not on the 303(d) List:

- Null Hypothesis: Waterbody is in compliance with numeric nutrient standards
- Alternative Hypothesis: Waterbody is not in compliance with numeric nutrient standards

In effect, this is a "guilty until proven innocent" approach for streams already considered to have water quality problems, and an "innocent until proven guilty" approach for newly assessed streams California uses the same approach (California Environmental Protection Agency State Water Resources Control Board, 2004).

A.4.2.2 Alpha, Beta, Effect Size

In statistical testing alpha, beta, effect size, and critical exceedance rate interact, and changes in one affect changes in the others. In environmental compliance work, there are strong arguments for attempting to balance type I (alpha) and type II (beta) errors; in doing so, it is important to consider the form of the null hypothesis and the implications for making one error or the other. Basically, each type of error has ramifications; one type of error leads to degradation of the environment, the other type of error leads to unnecessary expenditures on the part of the regulated entity. Working towards balancing type I and II errors is a process which inherently recognizes the consequences of each error type (California Environmental Protection Agency State Water Resources Control Board, 2004, Page 52, Appendix C; Mapstone, 1995, Page 178; Schroeter, et al., 1993; U.S. Environmental Protection Agency, 2002). *Given that* working towards balancing type I and type II errors is a valuable endeavor, here are general recommendations for the parameters to be input into statistical tests for nutrients in wadeable streams:

- Alpha should be about 0.25 or less (equates to ≥ 75% confidence level), depending on the form of the null hypothesis and its implications.

- Beta should be about 0.3 or less (equates to ≥ 70% power), and will vary according to the samples size (more on sample size minimum in **Section 5.0**).

- Effect size (gray zone) should be set at 0.15, per USEPA (2002).

In the statistical spreadsheet tools that accompany this technical appendix ("MT-NoncomplianceTool.xls" and "MT-ComplianceTool.xls"), one or the other file is used depending upon whether you are dealing with a new, unlisted stream (use "MT-NoncomplianceTool.xls") *vs.* a 303(d) listed stream (use "MT-ComplianceTool.xls"). You will be able to set alpha, critical exceedance rate (p), and effect size (p2) in the Exact Binomial Test in both of the files. The program will then return various sample sizes, their associated beta values, and the maximum number of exceedances allowed while still remaining in compliance with the criterion.

For the One-Sample Student's T-test, you must input alpha and the nutrient criterion in mg/L. The One-Sample Student's T-tests will then provide a result indicating if the statistical test can or cannot confirm the alternative hypothesis. (The alternative hypothesis will reverse, according to whether you are using the tool for a listed or for a new, unlisted stream).

A.4.2.3 Critical Exceedance Rate

Critical Exceedance Rate: *An estimate of the actual proportion of samples that exceed an applicable water quality criterion. When more than this proportion exceeds the criterion, the standard is not attained (i.e., stream is not in compliance with standard).*

Among the four statistical parameters critical to the Exact Binomial Test—alpha, beta, effect size, and exceedance rate—exceedance rate needs some kind of empirical ground-truthing to assure its validity. The implications of different alpha and beta errors can be understood relative to the form of the null hypothesis, while the effect size (gray zone) is not knowable *a priori*, and is therefore assumed; we recommend an effect size of 0.15 per EPA (U.S. Environmental Protection Agency, 2002). In contrast, an exceedance rate can be estimated using lines of reasoning, empirical evidence and literature values. The considerations used to estimate an exceedance rate for numeric nutrient standards were (**1**) recommended exceedance rates from EPA (U.S. Environmental Protection Agency, 2002) and (**2**) long-term benthic algae and nutrient relationships on the Clark Fork River, MT (Consideration (**1**) and (**2**) are further detailed below.). We recommend:

- A critical exceedance rate for compliance with numeric nutrient standards be set at 0.2 (20%)

Below are our two major considerations leading to the selection of the 20% exceedance rate.

(1) EPA recommends that, for a number of different polluting substances (e.g., fecal bacteria, conventional pollutants, toxic trace metals, etc.), criteria exceedance rates be set between 0.1 and 0.25

(10 to 25%) to protect beneficial uses (Environmental Research Laboratory-Duluth, 1997; U.S. Environmental Protection Agency, 2002).

(2) The analytical approach described in **2.1** below was undertaken in June 2008, and only considered Clark Fork River data through 2006. Subsequent data collection (through 2009) and a somewhat different approach to ascertaining an acceptable exceedance rate allowed us to update this analysis, as provided in **2.2.** Both analyses (that from 2008 [**2.1**], and the work done in 2011[**2.2**]) arrive at the same basic conclusion, and both are presented here. If readers are already familiar with the work in **2.1,** we recommend you skip to **2.2**.

(2.1)*The following analysis was completed in June 2008*.
Introduction: Numeric nutrient (TN and TP) and benthic algae (mg Chl *a*/m^2) standards have been in place on most of the Clark Fork River in Montana for about 6 years. A systematic collection of nutrient and algae data has been ongoing since 1998. At a number of sites both algae and nutrient data have been collected multiple times each year for nearly 10 years. These data lent themselves well to empirically deriving a numeric nutrient exceedance rate because some river sites almost always exceed the algae standards, while others do not. The question became:

> *Do sites on the Clark Fork River that routinely exceed the numeric algae standards exceed the river's established numeric nutrient (TN and TP) standards more frequently than sites that do not exceed the numeric algae standards?*

Benthic algae levels in excess of 150 mg Chl *a*/m^2 (maximum) are not to be exceeded during the summer (ARM 17.30.631). Maximum in this case does not refer to a single high repeat measure from a Clark Fork River site; it refers to the mean value of a series of repeat measures (n = 15 to 20) that are collected at a site *during a particular sampling event.* Clark Fork River sites are usually sampled several times throughout the summer. It has been noted for some years that, during the summer, some sites are usually above the algae standards, while others are not. TN and TP standards were established on the Clark Fork River (ARM 17.30.631) and, if ultimately met, should keep benthic algae below the nuisance threshold described above. However, an exceedance rate was never explicitly established in the regulations In carrying out the exceedance rate determination described herein, it is assumed that the magnitude of the TN and TP criteria on the Clark Fork River were accurately determined, and therefore any exceedance rate drawn from this analysis is meaningful.

Methods: Benthic algae and TN and TP concentration data where concurrently available for seven Clark Fork River sites from 1998-2006. Data were restricted to the time period June 30th to October 1st to generally comply with the summer growing season for this ecoregion (Suplee, et al., 2007)and the regulatory timeframe in ARM 17.30.631. Every benthic Chl *a* measurement from a site (n = 15-20 per sampling event) collected over time was treated as a repeat measure. This resulted in a grand total of 285 to 333 repeat measures of Chl *a* at each site for the period 1998-2006. A grand benthic Chl *a* mean was calculated for a site by averaging all the repeat measures collected between June 30th and Oct 1st for all available years. Nutrient data collected at the corresponding sites during the same time frames where similarly compiled. At each site nutrients were collected as a single grab sample and, as a consequence, there were fewer data (43 to 78 N or P samples per site). Total N data were not collected; however, Total Kjeldahl Nitrogen (TKN) and NO_{2+3} were. Therefore, for each site, individual Total N concentrations were calculated by summing the TKN and NO_{2+3} sample results collected simultaneously during a sampling event.

Next, the Clark Fork River TN and TP criteria concentrations were matched to their corresponding values in the nutrient cumulative frequency distributions for each site, and the associated percentile was recorded. For example, the TN criterion for the Clark Fork River is 0.3 mg/L, and it resulted that at site 9.0 (Clark Fork at Deer Lodge) 0.3 mg TN/L corresponded to the 23rd percentile of site 9.0's cumulative TN frequency distribution. This process was carried out for all 7 sites for both TN and TP. There is a break at the Blackfoot River confluence where the Clark Fork's upstream TP criterion (0.02 mg/L) differs from that below (0.039 mg/L); each TP criterion was applied as appropriate for a site's location along the river.

Results: Table A4-1 shows the results for 3 sites that, over the 1998-2006 time period, did not exceed the Clark Fork River's benthic algal biomass criteria. For this group of sites the nutrient criteria exceedance rate (both TN and TP) was, on average, about 8%. That is, nutrient samples whose concentrations exceed the standards occur only about 8% of the time at these sites. **Table A4-2** shows three sites that *did* exceed the benthic algae standard; for this group of sites, the nutrient criteria exceedance rate was, on average, about 58%. Sites in **Table A4-1** (did not exceed algae standard) had a range of exceedance rates (TN and TP) from 0.1%-24%, and sites in **Table A4-2** (exceed algae standard) had a range of exceedance rates from 27.7% to 88%. The remaining site examined (Site 12; Clark Fork River at Bonita), which is not presented in **Tables A4.1** or A**4.2**, had a mean algae density (144 mg Chl *a*/m^2) so close to the algae standard it was considered borderline. Site 12's exceedance rate was 30.8% for TN, 68% for TP.

Table A4-1 Sites on the Clark Fork River (CFR) Not Exceeding the Maximum Benthic Algae Standard (Growing Season, 1998-2006)

			Percentile in Site's Nutrient Frequency Distribution Matching CFR Standard		Criteria Exceedance Rate (%)	
Clark Fork River Site #	**Site Name**	**Long-term Benthic Algal Biomass (mg Chl *a*/m^2, growing season) Mean [median]**	**TN**	**TP**	**TN**	**TP**
15.5	Clark Fork above Missoula	96 [80]	90th	95th	10.2%	5.4%
22	Clark Fork at Huson	72 [52]	76th	96th	24.0%	3.8%
25	Clark Fork above Flathead	35 [20]	100th	99th	0.1%	1.5%
					Grand Mean:	**7.5%**
					Grand Median:	**4.6%**
					Maximum:	**24.0%**
					Minimum:	**0.1%**

Table A4-2. Sites on the Clark Fork River (CFR) Consistently Exceeding the Maximum Benthic Algae Standard (Growing Season, 1998-2006).

			Percentile in Site's Nutrient Frequency Distribution Matching CFR Standards		**Criteria Exceedance Rate (%)**	
Clark Fork River Site #	**Site Name**	**Long-term Benthic Algal Biomass (mg Chl *a*/m^2, growing season) Mean [median]**	**TN**	**TP**	**TN**	**TP**
9	Clark Fork at Deer Lodge	180 [147]	23rd	50th	77.0%	50.0%
10	Clark Fork above Little Blackfoot River	163 [117]	48th	12th	52.0%	88.0%
18	Cark Fork at Shuffields	197 [181]	50th	72nd	50.4%	27.7%
					Grand Mean:	**57.5%**
					Grand Median:	**51.2%**
					Maximum:	**88.0%**
					Minimum:	**27.7%**

Discussion: The main assumption of this analysis was that the magnitudes of the Clark Fork River nutrient criteria, which were established as standards for the river, are correct. That is, if the nutrient standards are achieved, then summertime algae levels should be kept below the established nuisance thresholds. It was assumed that, as has previously been shown, both N and P co-limit in the Clark Fork River (Lohman and Priscu, 1992); (Dodds, et al., 1997). It was further assumed that the algae standard (150 mg Chl *a*/m^2, site mean per sampling event) will protect beneficial uses. Regarding the later, research completed since the Clark Fork River standards were adopted in 2002 show that 150 mg Chl *a*/m^2 (site mean) is identified as a nuisance threshold by the Montana public majority (Suplee, et al., 2009). If all these assumptions hold true, then reasonable exceedance rates for the 9 year dataset can be derived and used as a case study. It would have been ideal to have a true population of data (rather than a subset of data for a single river over a specific time period) with which to carry out this analysis. But such data are not readily available, and the long-term dataset examined here will have to serve as a proxy.

Comparison of Clark Fork River sites 15.5, 22, and 25 (don't exceed algae standard; **Table A4-1**) *vs.* 9, 10, and 18 (do exceed algae standard; **Table A4-2**) show a clear separation in the consistency of compliance with the river's numeric nutrient standards. It is clear from **Table A4-2** that if the exceedance rate is about 50% then nuisance algae growth will almost certainly occur. But when the exceedance rate is ca. 5-10%, nuisance algae is unlikely to occur (**Table A4-1.**) For purposes of estimating a protective nutrient criteria exceedance rate, the range of exceedance rates from these site groups needs to be considered as well. Note that an exceedance rate *of as much as* 24% does not result in excess benthic algae in some cases (site 22; **Table A4-1**). On the other hand, notice that an exceedance rate *of as little as* 27.7% can result in non-compliance with the algae standard (site 18; **Table A4-2**). Thus, an exceedance rate around 25% probably represent a threshold; if about 25% of the dataset exceeds the nutrient criteria, then there are roughly equal odds that the site could have nuisance algae (or not). This is partially supported by the fact that the single site with borderline algae conditions (site 12, Clark Fork River at Bonita; 144 mg Chl *a*/m^2) had a TN exceedance rate of 30.8%.

Conclusion: These analyses show that over a 9 year period (1998-2006) sites on the Clark Fork River that have consistently exceeded the nuisance algae standard (150 mg Chl *a*/m^2, summertime max) have TN and TP exceedance rates with a central tendency around 54%. On the other hand, sites that did not exceed the benthic algae standards had TP and TN exceedance rates with a central tendency around 6%. Within each group (sites that do not exceed algae standards, those that do; **Tables A4-1** and **A4-2**), individual sites had exceedance rates as high as or as low as about 25%. This suggests that 25% may be an exceedance rate threshold where the ability to assure compliance with the algae standard becomes tenuous. Given that about 50% is certainly too high of an exceedance rate and will not protect beneficial uses, approximately 10% is probably too restrictive, and 25% is borderline, it is recommended that a nutrient exceedance rate be set to 20%.

(2.2) 2011 Analysis.

The 12-year (1998-2009) nutrient and algae dataset for the Clark Fork River was very large, and was first reduced prior to statistical analyses. Data reduction followed the following general pattern: At any given site (e.g., CFRPO-12), for any given year (e.g., 2005), and for any given parameter (e.g., TP concentration), the data were reduced to a monthly mean for each summer month (June, July, August, or September). First, quality control duplicates collected on the same day were reduced to a mean (TN data was not analyzed directly until 2009 and so, for 1998-2008 data, TN is the sum of TKN and NO_{2+3} samples collected simultaneously during a sampling event). Next, the mean of all individual days where sampling occurred within a month was calculated, resulting in a monthly mean. Nutrient sampling effort varied considerably from site-to-site and from year-to-year, and we did not want heavily sampled months or years to be over-represented in the dataset in the final analysis. In the manner we reduced the data, therefore, each monthly value carries equivalent weight, with some summer months being better characterized (i.e., sampled more days) than others.

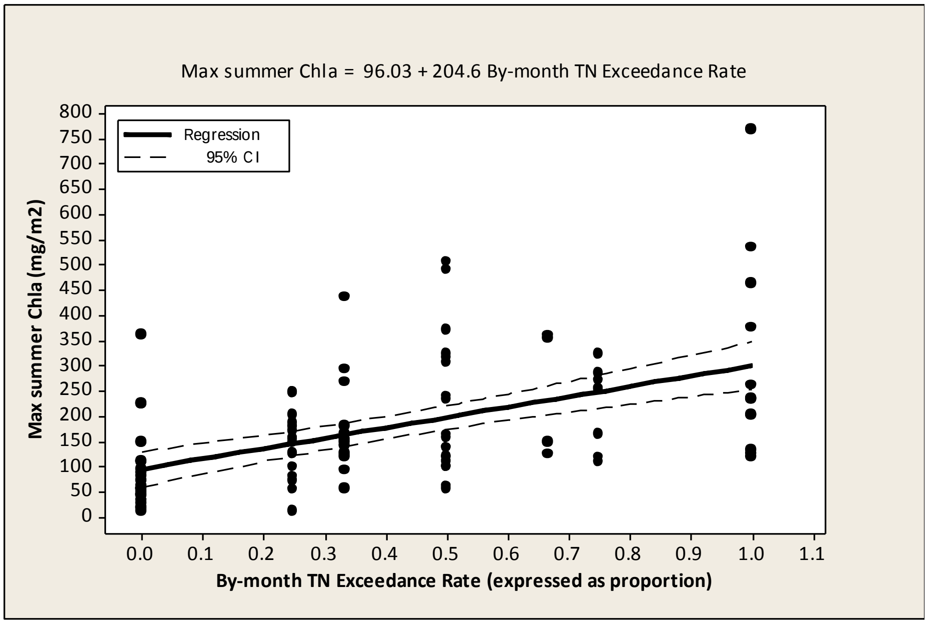
For benthic algae samples, up to 20 spatially-dispersed replicates were collected at a site during any given sampling event. Algae sampling events occurred only once a month. Thus, for a given site/year/month, the benthic algae mean calculated was the value used.

We next determined if each mean nutrient concentration, computed on a month-by-month basis, was above or below the Clark Fork River's applicable standards (TP or TN). This was only carried out for sites and times which had corresponding benthic algae samples. Then, we determined the proportion of months during a summer, at a site, that exceeded the river's nutrient criteria. For example, if a site in 2008 was sampled in June, July, August, and September, and June and August exceeded the TN standard, the TN exceedance rate for summer '08 would be 0.5 (50%). Each exceedance rate was then associated with its corresponding "Max Summer Chl *a*" value (exceedance rate as X, Max Summer Chl *a* as Y). Max Summer Chl *a* is the highest mean monthly Chl *a* value encountered during the summer at a site, per ARM 17.30.631. TN or TP data that were collected *after* the Max Summer Chl *a* event occurred were not included (e.g., if the Max Summer Chl *a* occurred in August, we did not include in the analysis the September TN or TP data for that site/year). Finally, least squares regressions (with 95% confidence intervals) were run for TN exceedance rate vs. Max Summer Chl *a* and TP exceedance rate vs. Max Summer Chl *a,* combining all sites and years together. The results are shown on the next page in **Figure A4-1.**

Regression statistics for both regressions were significant ($p << 0.01$). Using the line equations shown in **Figure A4-1**, 150 mg Chl *a*/m^2 (i.e., the maximum allowable benthic Chl *a* level for a summer; ARM 17.30.631) equates to a 26% exceedance rate of the TN standard and a 31% exceedance of the TP standard. The equivalent exceedance rates corresponding to the upper 95% confidence intervals (which are more conservative) are about 11% and about 5% for TN and TP, respectively.

These Clark Fork River data demonstrate that, across 10 sites with 12 year's worth of monitoring, there is a significant, definable relationship between benthic algal growth and the frequency of exceedance of the river's nutrient standards. That is, sites which frequently exceed the nutrient standards have higher levels of benthic algae. Sites that experience greater than about 25-30% exceedance of the nutrient standards will develop nuisance benthic algal growth, i.e., growth equal to or greater than 150 mg Chl *a*/m^2.

The analytical approach taken in 2008 (**2.1** above) was more coarse than what we have done here, in that it lumped all data by site and then looked to see how often that site—over the long haul—exceeded the nutrient standards. This analysis, in contrast, looks at each site and each summer as an individual event, and then collectively evaluates all the data together, regardless of location along the river (**Figure A4-1**). Interestingly, the overall results between the earlier analysis and the current one are largely the same, in spite of the different analytical approaches. If we continue to assume that the nutrient standards on the Clark Fork River are largely correct in magnitude, then this latest analysis indicates we would want to keep exceedance rates of the applicable nutrient standards between 5-31%, if we want to keep benthic algae below nuisance levels. Since these results correspond nicely to the earlier analysis, we continue to recommend that nutrient criteria exceedance rates be set at 20%.



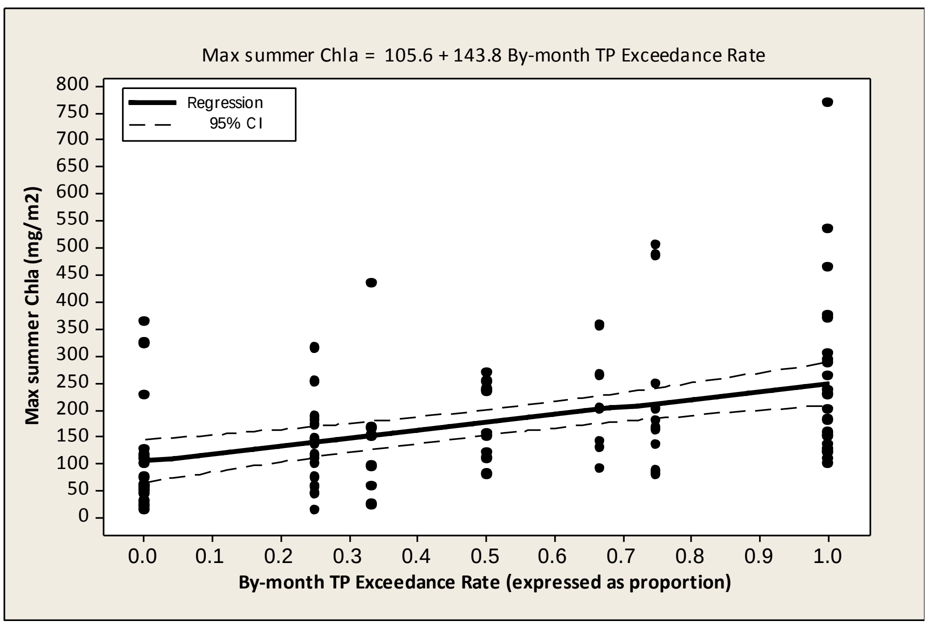


Figure A4-1. Least squares regression for TN exceedance rate vs. Max Summer Chl *a* (upper panel) and TP exceedance rate vs. Max Summer Chl *a* (lower panel), for ten Clark Fork River monitoring sites (1998-2009). Dotted lines are the 95% confidence intervals. Both regressions are significant (p << 0.01).

A.5.0 MINIMUM NUMBER OF NUTRIENT SAMPLES

The final consideration is minimum nutrient sample size. A "nutrient" sample refers to a nutrient type, such as TP or TN Sample sizes apply to each nutrient type, and not to the total number of nutrient samples collected from a stream segment. So, if 7 TN and 7 TP samples were collected from a segment they would not represent 14 samples, but rather 7 of TN and 7 of TP. There is extensive discussion of determining appropriate sample size on a study-by-study basis in USEPA (2002). However, the recommendations made here for determining compliance with the numeric nutrient standards are meant to apply generally to all Montana wadeable streams, mainly for purposes of 303(d) listing/delisting. *Please note that these sample size minimums do not apply to biological samples (e.g. benthic algae or diatoms) that may be collected concurrently with the nutrient samples.*

For unlisted streams, those for which the form of the null hypothesis is "complies with standard" (**Section A.4.2.1**), the implication for making a type II (beta) error is that a truly non-compliant stream segment would be incorrectly declared compliant. This is a scenario DEQ wants to minimize, and so the probability of such an outcome should be reduced well below 50%, i.e., well below that of a coin flip. The Exact Binomial Test in the accompanying spreadsheet tool can be used to estimate minimal sample sizes. In the test it can be seen that if alpha (type I) error is set to 0.25, exceedance rate (p) to 0.2, and effect size/gray zone (p2) to 0.15 (entered value = 0.35), then a Beta (type II) error of 0.35 is achieved with 12 samples. (Note in the spreadsheet that introducing lower and lower alpha values causes beta error to increase and, therefore, many more samples are needed to try to balance alpha and beta errors.) Twelve samples is about as low an **n** that can be used and still have roughly balanced (0.25 *vs*. 0.35) alpha and beta errors that are each well below 50%.

For listed streams, a similar approach is used. Listed streams are those for which the form of the null hypothesis is "does not comply with standard". In this scenario, the implication of making a type I (alpha) error is that a truly non-compliance stream segment is incorrectly declared compliant; again, this is a scenario. DEQ wants to minimize Setting the alpha to 0.25, exceedance rate (p) to 0.2, and effect size/gray zone (p2) to 0.15 (entered value = 0.05), a beta error of 0.14 (14%) can be achieved with 13 samples. This is a reasonable balance of type I and II errors, and provides a total sampling effort about the same as that for unlisted streams. Given these considerations, it is recommended that:

- For new, unlisted stream segments, a minimum of 12 independent samples for any given nutrient be collected for compliance determination.

- For 303(d)-listed stream segments that already have one 1 nutrient criteria exceedance for a given nutrient, a minimum of 13 independent samples (this total can include newly collected as well as previously collected samples) should be used for compliance determination.

- For listed streams with 13-18 total samples that already have 2 or more exceedances for a given nutrient, the default conclusion is that the stream segment has failed the Exact Binomial Test (no further sampling required at this time). Run the t-test as well and incorporate results in decision matrix.

- For listed streams that have > 18 samples for a given nutrient, set alpha to 0.25, exceedance rate to 0.2 and effect size to 0.15 in the Exact Binomial Test, and determine if the reach is (or is not) in compliance with the Exact Binomial Test. Carry out the same for the T-test.

- If a very large dataset (> 300 samples) is available for a particular stream, then lower type I (alpha) and type II (beta) error can be achieved with higher confidence in the results. Use the special feature of the Exact Binomial Test to help define these confidence levels. Confer with Standards Section if needed.

APPENDIX B. DETAILS ON ASSESSMENT METHODOLOGIES FOR WADEABLE MOUNTAIN AND TRANSITIONAL STREAMS

The following provides the rationales used for the selection of the assessment tools. It also provides the rationales for selected impact thresholds. This information is summarized in **Section 4.0** of the main document.

B.1.0 ASSESSMENT OF BENTHIC ALGAL GROWTH

An evaluation of statistical uncertainty in the site averages calculated using DEQ's standard procedure for collecting and analyzing benthic Chl *a* is detailed in **Appendix A** of the Chl *a* SOP (Montana Department of Environmental Quality, 2011c). To summarize: if a benthic Chl *a* sampling event has followed the SOP, DEQ is confident that—for a typical wadeable stream— at least 80% of the time the measured Chl *a* average calculated will be within ±30% of the true average. Given this known variability, decision points pertaining to benthic algae growth and harm-to-uses have been developed, and are further detailed in **Sections B.1.1** and **B.1.2** below

B.1.1 BENTHIC ALGAL CHL *A* LEVELS AND THE RECREATION USE

It is reasonable that, once a site's true average algal level exceeds about 150 mg Chl a/m^2, impairment to the recreational use has occurred. This is shown in **Figure B1-1**. But we also have to account for the uncertainty around the Chl *a* measurement. Shown are the three photographs that bracket the acceptable-unacceptable threshold, per Suplee *et al.* (2009) each with their interval widths (based on the 20 Chl *a* replicates associated with each photo) calculated at the 90% confidence level. Once algae levels have reached the lower bound of photo E (photo E's lower confidence bound = 169 mg Chl a/m^2), the acceptability threshold has already been exceeded. This is because the public majority finds the algae level shown in the photo to be highly undesirable. The gray zone in **Figure B1-1** represents the zone where public acceptability rapidly transitions from "OK" to "Not OK". Going forward, any measured average Chl *a* value DEQ believes could plausibly be as high as 165 mg Chl a/m^2 (in the gray zone, and at upper confidence bound of photo F, but still below the lower confidence bound of photo E) should be considered an exceedance.

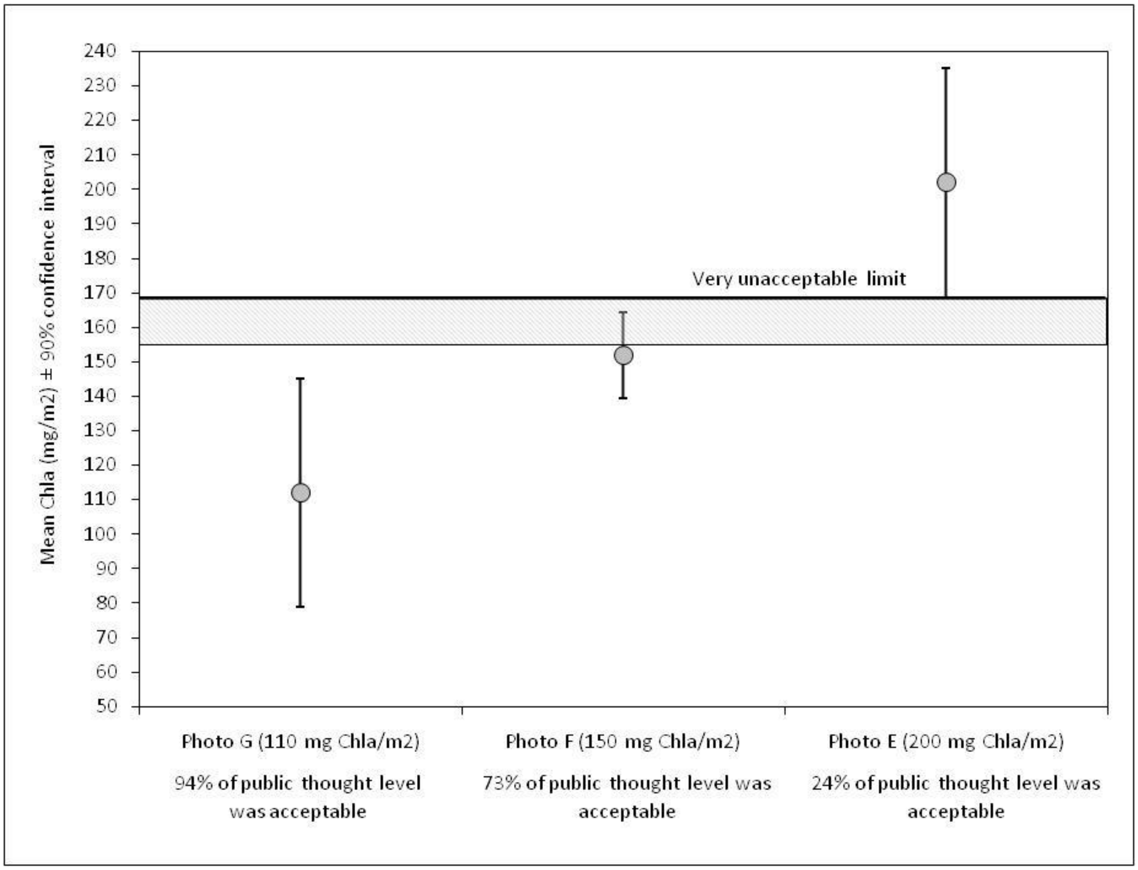


Figure B1-1 Averages (Gray Dots) and 90% Confidence Bands for the Three Photos Bracketing the Acceptable-unacceptable Threshold, per Suplee et al. (2009) The gray band shows the algae level range across which public opinion rapidly shifts from acceptable to unacceptable.

Returning to DEQ's algae-sampling protocol, the average that is calculated for any given sampling event has a definable interval width and, when the upper bound of that interval reaches about 165 mg Chl *a*/m^2, an impairment at that site should be considered to have occurred. Using the approach outlined in Appendix A of DEQ (Montana Department of Environmental Quality, 2011c) and given that ***n*** = 11, Coefficient of Variation (CV) = 73%, DEQ can be 80% certain that the true benthic algae average may be as high as 165 mg Chl *a*/m^2 when the measured average is ≥ 129 mg Chl *a*/m^2. Therefore, any sampling event for which the measured benthic algal Chl *a* average is **≥ 129 mg Chl *a*/m^2** should be considered an exceedance, and in violation of ARM 17.30.637(1)(e).

B.1.2 Benthic Algal Chl *a* Levels Relative to Late-season Dissolved Oxygen Problems, and Potential Impacts to Fish and Associated Aquatic Life

In 2009, DEQ commenced a BACIP (Before After Control Impact Paired) design, whole-stream nutrient addition study to better understand the exact way stream changes are manifested due to nitrogen and phosphorus pollution. We wanted to understand the relationship between these changes and stream beneficial uses (Montana Department of Environmental Quality, 2010a)[16]. In summer 2010 soluble nitrogen and phosphorus were added to a reach (High Dose reach) of the stream, and major changes occurred as a result. One of the most interesting findings was the temporal manner in which stream dissolved oxygen (DO) problems occurred, and the relationship of those DO levels to measured benthic algae levels.

After nutrient additions began in early August 2010 and then continued throughout summer 2010, DO standards that protect fish (1 Day minima in DEQ-7;(Montana Department of Environmental Quality, 2010b)) were never exceeded — nor even approached —in the High Dose reach. This was true in spite of large daily DO swings (**Figure B1-2**)[17]. Relative to the upstream Control reach, DO increased dramatically during the day, but did not at night fall much below the Control reach values (**Figure B1-2**). Benthic algal production (growth) exceeded respiration throughout most of the summer and this, in conjunction with adequate re-aeration due to the stream's flow, likely prevented nighttime DO levels from dropping below the DO standards. However, in fall, the growing season's accumulated algal growth began to senesce en masse and we observed large amounts of decaying algae on the stream bottom in early October. The decaying algae induced a high oxygen demand which was concentrated near the bottom, in affect acting like a sediment oxygen demand (SOD), and which in turn led to exceedances of the DO standards. (The YSI in **Figure B1-2** was monitoring DO about 20 cm off the stream bottom, in a run.) The DO standards exceedances all occurred late in the season, after robust benthic algae growth had ended.

It should be noted that two other YSIs (one deployed upstream and another further downstream of the one in **Figure B1-2**, thus bracketing the High Dose reach) simultaneously recorded DO concentrations, none of which violated DO standards, even in October (data not shown). This was apparently due to longitudinal changes in stream morphology (e.g., width/depth relationships) and their affect on stream re-aeration, and dead algae accumulation on the bottom. We calculated that for DO to decline from what was measured by the YSI just upstream of the High Dose reach to that recorded in early October in **Figure B1-2** would require a SOD higher than any we could locate in the literature[18]. This suggests that DO concentrations were not uniform from stream surface to bottom, but rather, a bottom-to-surface DO gradient likely existed. Our analyses further indicated that DO was probably near zero near the bottom, and then near saturation at the surface. These findings suggest that DO problems of this nature can be both longitudinally and vertically patchy along the stream channel.

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[16] Although this study was carried out in a C-3 warm water fishery stream, the stream's key characteristics relative to algal growth make it a reasonable comparison to western Montana gravel-bottom streams, as we have done here. The stream has a gravel dominated substrate, perennial flow of about 4-6 CFS in summer, a water surface slope of 0.4%, and riffles are common throughout, as are pools. The dominant filamentous algae that grew during the study was *Cladophora*, which is also commonly found in western MT streams.

[17] Routine QC checks of the instrument, including mid-project calibration, routine instrument cleaning, etc. was undertaken The low DO values measured are considered valid measurements.

[18] Stream water biochemical oxygen demand (BOD) samples were also collected in early October at the site. They were non-detect, thus the DO consumption had to be coming from the decaying algal material on the bottom.

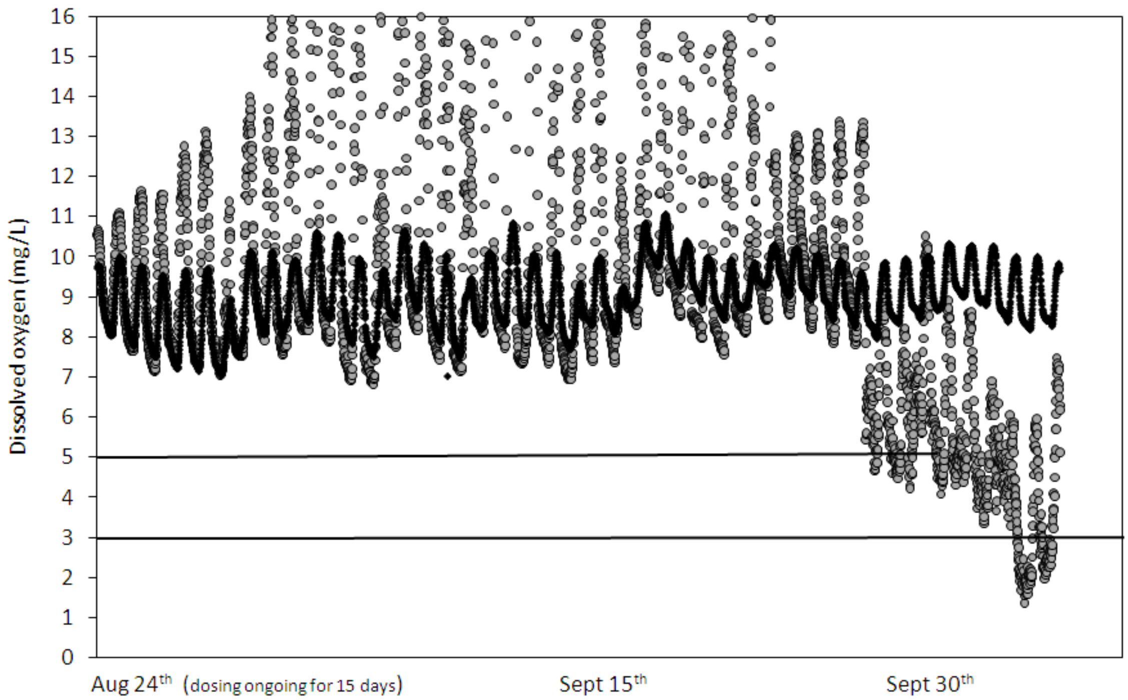


Figure B1-2. Temporal Changes in Dissolved Oxygen (DO) Levels in the High Dose Study Reach as Measured by Deployed YSI Instrument, 2010. Gray dots are DO observations at the High Dose Reach, collected at 15 min intervals. For comparison, the black line oscillating around 9 mg/L is the DO measured by another YSI in the upstream Control reach, which did not receive nutrient additions. The horizontal black lines are (upper) the adult fish and (lower) juvenile fish DO standards for this stream.

Returning to the High Dose reach's benthic-algal growth, benthic algae Chl *a* levels at the end of the growing season reached 127 mg Chl a/m^2 (**Table B1-1**). It is apparent from **Figure B1-2** that this level of benthic algae was sufficient to induce DO violations along the channel when the algae died and decomposed en masse. The implication of this finding is that the late-season average benthic algae we measured (127 mg Chl a/m^2) has the potential, in wadeable streams, to cause DO standards exceedances in the post-summer period, probably in late September or October. Although this study was carried out in a C-3 warm water fishery stream, the stream's late season water temperatures (which ranged from about 12-16 ^{0}C) are comparable to what is observed in typical western Montana gravel bottom streams at that time of the year. While it is true that water temperature strongly affects DO concentration, we would not expect western Montana streams manifesting similar algal densities to be able to compensate (i.e., maintain DO above standards) due to their having cooler water temperatures, as their temperatures are often about the same at that time of the year.

Table B1-1. Benthic Algae Density Measured at the High Dose Study Site, 2010

Sampling Date	Reach average benthic-algae density (mg Chl a/m^2)	Chl *a* replicates' CV(%)*	Reach average AFDW density (g/m^2)	AFDW replicates' CV(%)*
August 26, 2010	**111**	123	**26**	93
September 8, 2010	**116**	97	**34**	45
September 22, 2010	**87**	82	**37**	61
October 6, 2010	**127**	90	**33**	73

* 11 replicates were collected for each sampling event. Less than 11 were used to calculate reach average AFDW because core samples, if collected, are not included in the calculation of average reachwide AFDW density.

As mention at the beginning of **Section B.1**, there is a definable level of uncertainty around any given benthic Chl *a* average, but there is no way to know if the values in **Table B1-1** are at the low end, high end, or in the middle of that range. We'll assume here that the 127 mg Chl a/m^2 measured is, in fact, accurate. Thus, to be protective and assure that DO problems that could harm fish and associated aquatic life are prevented from occurring, we recommend that when a site's average benthic Chl *a* exceeds **120 mg Chl a/m^2** it is too high, and should therefore be considered an impact to fish and associated aquatic life.

B.1.3 Benthic Algal AFDW Levels and Harm-to-use Thresholds

Ash Free Dry Weight (AFDW) collected from natural stream-sediment surfaces is a useful measurement for estimating algal biomass. The laboratory method basically oxidizes and reports back the mass of all organic material in the sample (American Public Health Association, 1998). It is useful in that it provides an additional means of assessing accumulated algal biomass independent of Chl *a.* Chl *a* levels tend to be highest during peak growth, and then decline later as the Chl *a* molecules degrade as the algae senesce (Stevenson, et al., 1996). If an assessor samples a stream late in the season, they may find fairly low Chl *a* values in spite of the presence of a large biovolume of algal material. Thus, a site that may truly have an excess algae problem could potentially be assessed as unimpaired simply due to the fact that the samples were collected late in the season.

For this reason, we recommend that AFDW be determined for all samples when Chl *a* is collected. AFDW can be determined from the same sample in a subsequent analysis that follows the Chl *a* analysis Site average. AFDW can be determined from individual replicates, or as a weighted average.

Note: AFDW results from core samples should never be included in determining a site's average AFDW. The method measures organic material from the entire core sample, not just the surface where the algae are growing, and will therefore over-report AFDW.

DEQ has not collected AFDW using the 11-transect method long enough to be able to carry out the type of statistical uncertainty calculations used for Chl *a* (Appendix A of (Montana Department of Environmental Quality, 2011c). However, there are good estimates of what comprises too much algal AFDW. In Suplee et al. (2009), the threshold Chl *a* level of 150 mg/m^2 corresponds to 36 g $AFDW/m^2$ In New Zealand, extensive analysis of algal AFDW resulted in a recommendation of 35 g $AFDW/m^2$ as the maximum level for gravel/cobble streams, to protect recreation use (Biggs, 2000). Note in **Table B1-1** above that the late season AFDW corresponding to 127 mg Chl a/m^2 (the Chl *a* level linked to the late-season DO problems) is 33 g/m^2 Long -term monitoring in the Clark Fork River (1998-2009) shows that the average summer AFDW at sites that do not develop nuisance algae (i.e., they are consistently <150 mg Chl a/m^2) ranged from 17 to 48 g $AFDW/m^2$ (mean: 27 g $AFDW/m^2$). Given the values presented, we recommend that site average AFDW (i.e., mean of the 11 replicates collected at a site, replicates being only templates or hoops) should be no greater than **35 g $AFDW/m^2$.** This value should be protective of both fish and aquatic life and recreation uses.

B.1.4 Some Additional Considerations Regarding Benthic Algae Sampling

Recently, DEQ has instituted an economization practice that consolidates all hoop, core, or template samples from a sampling event together, so that only three (at most) Chl *a* samples need to be analyzed, instead of eleven. While unquestionably thrifty, the ability to determine the replicates' variance has been lost. All of the Chl *a* confidence calculations discussed in **Section B.1.1** assume that a sampling event will manifest a typical replicate CV of 73%, but this is an assumption In cases where it is very important to truly know the replicates' CV, the replicates should each be analyzed separately..

Cases may also arise where an entity is not satisfied with the level of confidence or interval widths DEQ has presented here. Collecting 11 samples in a stream reach is already a time consuming and expensive procedure, and we consider the confidence level (80%) and interval width (± 30% of the mean) to be satisfactory for algae sampling. If an entity (regulated or otherwise) desires higher levels of precision, than it is our recommendation that the financial cost to achieve those levels fall to the entity.

If more precision is wanted, how many more algae samples should be collected? Long term sampling of benthic Chl *a* by Dr. Vicki Watson on the Clark Fork River shows that with about 20 replicates, one can be 90% confident that the measured average Chl *a* is within ± 20% of the true average. For benthic algae sampling, which is inherently noisy, this is a fairly high degree of confidence. For DEQ's wadeable stream method, this would involve placing 20 transects instead of 11 along a site, with algae collection occurring at each of the 20 transects using the systematic approach (R, L, C, repeat) described in the SOP.

B.2 Assessment Using Biometrics

DEQ has used diatom-algae assemblages and macroinvertebrate assemblages for many years to make assessment of stream water quality and condition. Some of these metrics are being incorporated into the process for assessing excess nitrogen and phosphorus pollution. Details of each are given in the **Sections B.2.1** and **B.2.2** below.

B.2.1 Biometrics Based on Diatom Algae

DEQ has been using benthic diatoms to assess water quality since the 1970s. Earlier approaches used diagnostic and descriptive biometrics based on quasi-universal ecological attributes of diatom species and observed structural characteristics of benthic diatom associations (Bahls, et al., 2008). The current approach (initiated in 2004) uses regional classification, stream reference sites, a priori knowledge of stressors in streams, and discriminant function analysis to identify "increaser" taxa that respond to specific stressors and in a predictable way (Teply and Bahls, 2006; Bahls, et al., 2008; Tepley and Bahls, 2005). The metrics were specifically developed to indicate the likelihood of nitrogen and phosphorus impairment, have been developed for many regions of the state, and can function properly in the presence of other major pollutants (Teply, 2010a; Teply, 2010b). Please see the periphyton SOP (Montana Department of Environmental Quality, 2011b) for details. Each sample will provide the probability of a nutrient problem, such as in this example:

> *This indicates that the sample represents a stream that has about a 65% percent probability of being impaired due to nutrients (nitrogen or phosphorus) under 303(d) guidelines. This probability is based on past evidence of taxa associated with nutrient-impaired streams in the Northern/Canadian Rockies Stream Group. Nutrient Increaser Taxa do not discriminate other causes of impairment and this result does not*

indicate whether the stream may or may not be impaired due to other causes.

Diatom nutrient-increaser metrics are available for the Northern and Canadian Rockies ecoregions, the Idaho Batholith ecoregion, and a series of level-IV ecoregions that predominate along the Rocky Mountain Front (i.e., mountain-to-plains transitional zones). **Note: There is currently no validated nutrient-increaser model for use in the Middle Rockies ecoregion.** As of this writing, a sample that indicates **>51%** probability of impairment by nutrients should be considered to indicate the sample is from a site with excess nutrient problems. Findings based on diatom samples are not, however, stand alone, and need to be incorporated with other data per the decision framework described in **Section 3.0** of the main document.

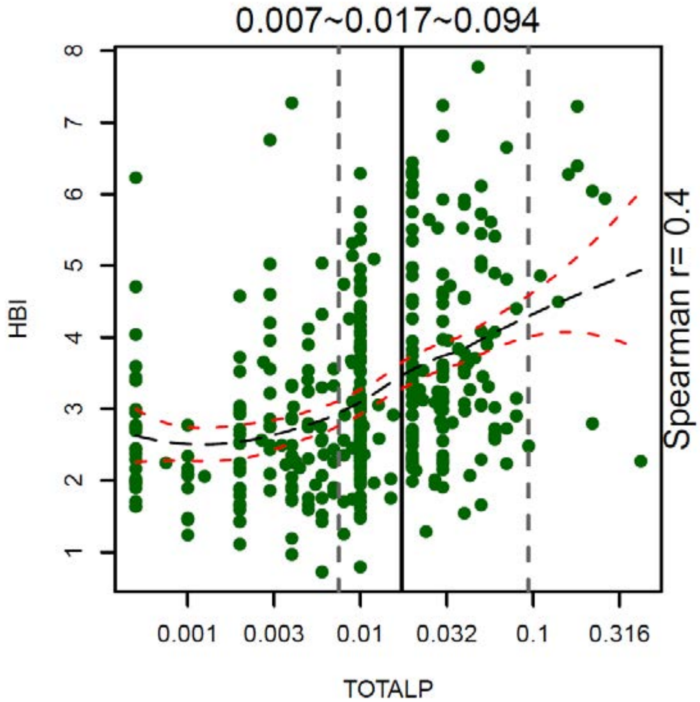
B.2.2 Biometrics Based on Macroinvertebrates

Benthic macroinvertebrates have been used for a long time as indicators of stream water quality (e.g., (Hilsenhoff, 1987; Barbour, et al., 1999). Recently, DEQ and EPA carried out a correlation analysis using Montana data to examine the relationship between stream nutrient concentrations and benthic macroinvertebrate metrics (Tetra Tech Inc., 2010). Among the metrics, one (Hilsenhoff Biotic Index, or HBI) is sufficiently well understood and showed a sufficiently patterned response to stream nutrient gradients in Montana's mountainous regions that we believe it can be used as a secondary response variable to help assess nutrient impacts. How the metric will be incorporated with other effect variables was discussed in **Section 3.0.** We here define a biological threshold for the HBI metric, giving consideration to the fact that almost all mountainous streams in Montana are to be maintained suitable for "growth and propagation of salmonid fishes and associated aquatic life" (A-Closed, A-1, B-1, C-1 classes), or "growth and marginal propagation of salmonid fishes and associated aquatic life" (B-2, C-2 classes).

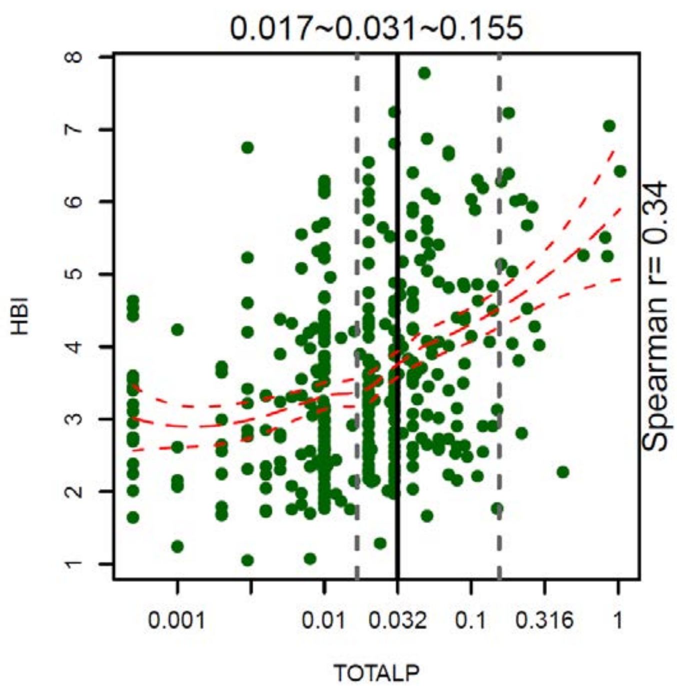
B.2.2.1 Hilsenhoff Biotic Index

HBI is based on tolerance values. A large number of macroinvertebrate taxa have been assigned a numeric value which represents the organism's tolerance to organic pollution (Barbour et al., 1999). HBI is then calculated as a weighted average tolerance value of all individuals in a sample. Higher index values indicate increasing tolerance to pollution.

Figure B2-1(A) shows the HBI *vs.* TP correlation in mountainous-region streams (Tetra Tech Inc., 2010). The data are from the "Mountains" site class (a.k.a. Mountains bioregion)(Montana Department of Environmental Quality, 2006). The Mountains Bioregion comprises stream sites whose catchments are mainly in the Middle Rockies, Canadian Rockies, Northern Rockies, and Idaho Batholith ecoregions and where elevation is greater than 1700 m, precipitation is greater than 700 mm/year, and annual mean daily maximum temperature is < 11^{o}C. Also shown is the same data, but this time aggregated simply by level III ecoregion rather than bioregion (**Figure B2-1(B)**); note the very similar patterns. This indicates that ecoregions and bioregions work about equally well as geospatial frameworks to segregate macroinvertebrate data for the purpose of correlation to stream nutrient concentrations.



A. Mountains bioregion.



B Middle Rockies ecoregion.

Figure B2-1. HBI metric vs. TP.

The relationship between HBI and TN is shown below (**Figure B2-2**), aggregated by ecoregions (only Middle Rockies ecoregion shown). As for TP *vs*. HBI, there is a noisy but discernable (and significant) relationship between nutrients and the HBI score.

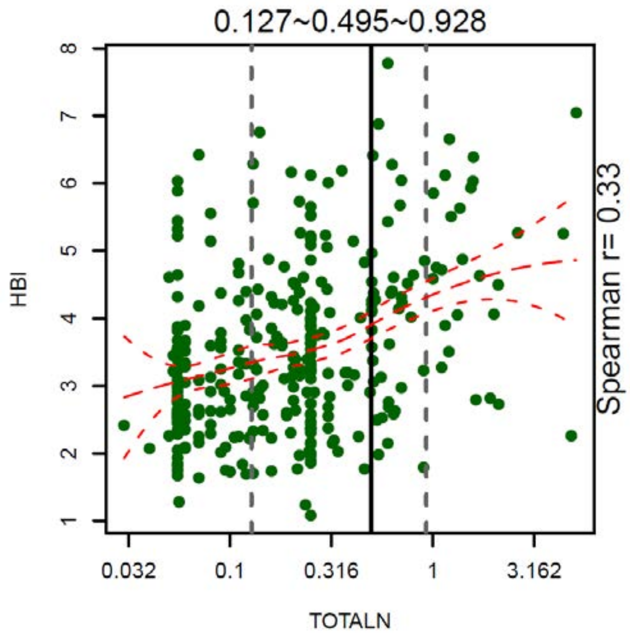


Figure B2-2. HBI Macroinvertebrate Metric vs. Stream TN Concentration Data are aggregated by ecoregions (Middle Rockies ecoregion shown).

Several components of **Figures B2-1 (A) and (B)** and **Figure B2-2** require explanation. Change-point analysis (Qian, et al., 2003) shows the statistically-derived point in the dataset where a shift, or threshold, in Y has occurred relative to X. In the figures, it is the vertical black line, bracketed to the right and left by its 90% confidence interval as dashed grey lines. This change-point, ± its 90% confidence interval, is shown as a numeric value at the top of each figure. The curving dashed lines running left to right are the locally-weighted regression line (LOWESS) and associated 90% confidence limit as dashed red lines. The Spearman's *rho* correlation value (Conover, 1999) between nutrient and metric is shown on the right side of each figure.

B.2.2.2 Interpreting the Macroinvertebrate Metric Correlations to Nutrients

Although **Figure B2-1** and **Figure B2-2** demonstrate significant correlations (parametric least-squares regression), they show large amounts of scatter. Numerous factors contribute to this scatter. For example, macroinvertebrates are separated from the direct effects of nutrient increases by one trophic level (i.e., nutrients directly influence aquatic plants and algae, and changes in plant species and biomass in turn influences macroinvertebrates). This is why they are considered secondary data, per the **Section 3.0** decision framework. In addition, environmental factors (natural and human-caused) other than nutrients influence macroinvertebrate populations, adding to the scatter in the relationships between the metrics and the nutrients. This is especially true for these data, which have been compiled over relatively large spatial areas and incorporate many different streams sampled over a long period of time (> 15 years). In spite of the scatter, there are patterns that can be discerned. Note, for example, that when TP is greater than 0.15 mg/L in **Figure B2-1(B)**, the likelihood of a stream having an HBI score <4 is very low.

The following section provides more detail on how **Figure B2-1** and **Figure B2-2** (and related data) were used to interpret the macroinvetebrate metrics relative to nutrients.

B.2.2.3 Patterns Observed Between HBI and Change-points, and Previously Used DEQ Thresholds

DEQ has in the past used thresholds for stream impairment using the HBI metric (Bukantis, 1998). Mountainous and intermountain valley regions used HBI values of 3.0 and 4.0, respectively, as the threshold values between full support and impairment of aquatic life (Bukantis, 1998). Hilsenhoff (1987) notes that the transition between Very Good water quality (slight organic pollution) and Good water quality (some organic pollution) is an HBI score of 4.5. How do the thresholds relate to the data seen in **Figures B2-1** and **Figure B2-2**? Results are shown below in **Table B2-1,** and include data from the Middle Rockies, the Mountains Bioregion, and the Low Valleys Bioregion.

Table B2-1. Nutrient Concentrations on the LOWESS Regression Line Corresponding to Specified Thresholds.
(Statistical Changepoint or Macroinvertebrate Metric Value) for **Figures B2-1, B2-2**, and Similar Graphs. Data are grouped by bioregions and ecoregions. Only Middle Rockies ecoregion shown due to insufficient data on other ecoregions.

			Corresponding Nutrient Concentration	
MT Bioregion	**Parameter**	**Metric Threshold Value**	**TP (mg/L)**	**TN (mg/l)**
Mountains bioregion	**Changepoint**		**0.02**	**0.46**
Mountains bioregion	HBI	4.0	0.07	1.15
Mountains bioregion	HBI	4.5	0.19	beyond graph
Low Valleys bioregion	**Changepoint**		**0.04**	**0.32**
Low Valleys bioregion	HBI	4.0	0.04	0.32
Low Valleys bioregion	HBI	4.5	0.19	1.35
Middle Rockies Ecoregion	**Changepoint**		**0.03**	**0.50**
Middle Rockies Ecoregion	HBI	4.0	0.04	0.55
Middle Rockies Ecoregion	HBI	4.5	0.16	1.7

One observation that can be made about **Table B2-1** is that, in any given region, HBI scores of 4.5 can correspond to nutrient concentrations much higher than the corresponding changepoint concentrations, whereas nutrient concentrations matching an HBI of 4.0 usually match fairly closely to the changepoint concentrations. These results suggest that an HBI score of 4.0 is a meaningful threshold relative to stream nutrient concentrations in western Montana streams. This conclusion stems from the fact that an HBI of 4.0 has previously been recommended as a threshold by DEQ for this region (Bukantis 1998), and that the data show a statistically-significant threshold (i.e., a change in biological structure relative to nutrients) at an HBI of 4.0. An HBI score of 4.0 is also meaningful from the perspective of water quality protection, as scores of 4.5 indicates transition into conditions where some organic pollution is already noted (Hilsenhoff, 1987). Thus, if elevated nutrients are suspected and an HBI score of >4 is encountered, there is a good chance that excess nutrients are causing the problem.

APPENDIX C. DETAILS ON ASSESSMENT METHODOLOGIES FOR WADEABLE PLAINS STREAMS

Benthic algae levels discussed in **Appendix B**, **Section B.1.1** are appropriate for the mountainous and transitional (mountain-to-plains) region of the western part of the state. Many eastern Montana plains streams are ecologically different from western Montana streams, and therefore the results from the public perception algae survey (Suplee et al., 2009) should probably not be universally applied to them. Montana plains streams often become intermittent, are generally low gradient, commonly have mud bottoms and are often turbid, and frequently have substantial macrophyte populations. It is not uncommon in these streams to see macrophytes intermixed with filamentous algae and floating masses of green algae; these conditions are even occasionally observed in plains streams minimally impacted by people (i.e., plains reference streams). These situations make measurement of benthic algal biomass in plains streams difficult and complicated. Further, our analysis shows that 4% of the sampling-event averages from plains reference streams have benthic algae >150 mg Chl *a*/m^2, whereas none of the sampling-event averages for benthic algae in western Montana reference streams even approach this value (e.g., the highest sampling-event average for a western Montana reference site was 76 mg Chl *a*/m^2). These findings, taken together, suggest that benthic algae measurement is probably not the best assessment tool for determining impairment for plains region streams. As such, we recommend that benthic algal biomass not be used to assess plains streams.

The following sections discuss the assessment tools we believe are more appropriate for assessing wadeable streams of the plains region.

C.1.0 ASSESSMENT USING DIATOM ALGAE BIOMETRICS

Nutrient-increaser diatom metrics have been developed for plains wadeable streams using the same methods as the diatom metrics presented in **Appendix B.** The metrics will indicate the likelihood of nitrogen and phosphorus impairment, and can function properly in the presence of other common pollutants such as sediment and metals (Teply, 2010a; Teply, 2010b). Please see DEQ's periphyton SOP (Montana Department of Environmental Quality, 2011b) for details. Each periphyton sample will provide the probability of a nutrient problem, such as in this example:

> *This indicates that the sample represents a stream that has about a 25% percent probability of being impaired due to nutrients (nitrogen or phosphorus) under 303(d) guidelines. This probability is based on past evidence of taxa associated with nutrient-impaired streams in the Warm-water Stream Group. Nutrient Increaser Taxa do not discriminate other causes of impairment and this result does not indicate whether the stream may or may not be impaired due to other causes.*

As of this writing, a sample that indicates **>51%** probability of impairment by nutrients should be considered to indicate the sample is from a site with excess nutrient problems. Findings based on diatom samples are not, however, stand alone, and need to be incorporated with other data per the decision framework described in **Section 3.0** of the main document.

C.2.0 ASSESSMENT USING THE DIFFERENCE BETWEEN THE DAILY MAXIMUM DISSOLVED OXYGEN CONCENTRATION AND THE DAILY MINIMUM DISSOLVED OXYGEN CONCENTRATION (DELTA)

We initially considered using DEQ's DO standards for routine assessment of plains streams. But examination of long-term DO datasets, including those from the plains-stream dosing study (**Appendix B, Section B.1.2**), showed that DO standards are a fairly insensitive way to assess nutrient impacts in plains streams. We found only a low number of instances where streams that we know have excess nutrient impacts consistently violated the DO standards, and some nutrient-impacted stream sites never violated the DO standards at all (at least during summer and early fall).

We found that streams that have high daily DO **delta** (i.e., the daily maximum DO minus the daily minimum DO) may *eventually* manifest DO standards violations late in the year, when the algae die and decompose en masse (see **Appendix B**, **Section B.1.2**). To address this, the DO monitoring-period could be extended (e.g., to the end of October or early November), but this is not always practical given the unpredictable onset of winter and its affect on road access, retrieval of deployed instruments, etc. In lieu of extending the monitoring season, measurement of summer and early fall DO deltas has great potential as an assessment tool. Others have found that DO delta is related to harm to aquatic life. In Minnesota, strong positive correlations are found between the percent tolerant fish and the magnitude of the DO deltas. At DO deltas <4.5 mg/L, tolerant fish are usually <10% of the total fish population, but when DO deltas are > 4.5 mg/L tolerant fish become a substantial proportion of the population. Conversely, sensitive fish exhibit a wide range of values at DO deltas <4 mg/L, but above 4.5 mg/L they decline to 10% or less of the fish population(Minnesota Pollution Control Agency, 2010). The state of Minnesota is recommending that, for the northern plains regions at its southern end, measured DO deltas should not exceed 5.0 mg/L.

DO delta is also shown to be lower in reference streams. In Tennessee, the maximum DO delta value reported in a wadeable reference stream is 4.0 mg/L, whereas about 45% of impacted streams assessed have measured DO deltas greater than 4.0 mg/L (Arnwine and Sparks, 2003).

We calculated DO deltas for Montana plains <u>reference</u> streams. There were a total of 177 day's worth of delta values from the Box Elder Creek Control reach (Montana Department of Environmental Quality, 2010a) and the Little Beaver Creek Reference Site, collected in 2009 and 2010. 90% of the daily DO deltas from these reference sites were less than 5.3 mg/L. The single highest DO delta measured was 6.6 mg/L, from Little Beaver Cr, which is influenced by the presence of macrophytes. (Also, on about 10% of occasions, DO at the Little Beaver Cr reference site dropped just below the juvenile fish standard of 5.0 mg DO/L; however, it never got close to the adult fish DO standard of 3.0 mg/L.)

No fish data were collected contemporaneously with the reference site DO data, however fish populations have been evaluated in both of these streams (Bramblett, et al., 2005) at alternative reference sites (BoxElder_382_W and LittleBe_410_W) not far downstream. These alternative reference sites had among their fish populations substantial proportions of sensitive/intolerant species, especially Box Elder Creek in the Little Beaver Creek site 17% of the fish captured were considered sensitive/intolerant. Sensitive/intolerant species are typically the first species to disappear due to chemical and physical perturbations (e.g., low DO)(Barbour, et al., 1999). Assuming DO patterns at the alternative reference sites are roughly comparable to those which we monitored, the fish data suggest that a healthy fishery is being maintained in spite of occasionally high DO deltas and occasional exceedances of the juvenile fish criterion[19].

We employed change-point analysis (Qian, et al., 2003) to help identify any DO delta thresholds. We had 11 plains-stream locations, in both reference (Suplee, et al., 2005) and non-reference condition, which had continuous instrument-measured DO data (**Table C2-1**). These sites comprised both perennial and intermittent streams. Delta values were calculated, resulting in over 550 days of DO delta values. Each location was assigned a rating (1 through 4) representing our BPJ assessment of how strongly it was impacted by nutrients, and these ratings were associated with the corresponding DO delta values. The ratings used were: 1 = no known nutrient impact; 2 = low nutrient impact; 3 = medium nutrient impact; 4 = high nutrient impact. In quite a few cases the ratings could be very accurately assigned, as some sites were reference sites and some were part of the nutrient dosing study (e.g., all DO deltas associated with the High-dose reach were assigned a rating of 4). Change-point analysis was then run on the dataset with delta values on the X axis and their corresponding rating scores on the Y A highly significant ($p << 0.001$) change-point was identified at 6.0 mg/L (the 90% confidence interval for the change-point was 5.5 mg/L to 6.6 mg/L). Essentially, analysis showed that in moving from sites rated 3 to sites rated 4, the magnitude of the DO deltas ramped up dramatically, with the threshold of this change occurring at 6.0 mg/L.

Table C2-1. Long-term DO Monitoring Sites in Plains Streams

Station ID	Continuous Data Time Range	DO observations time-step (min)
Y26BOXEC08-upstream	Aug 25 to Sept 30, 2010	15
Y26BOXEC08-downstream	Aug 24 to Sept 30, 2010	15
Y26BOXEC04	July 26 to Sept 26, 2009 AND July 19 to Oct 7, 2010	15
Y26BOXEC09-upstream	Aug 25 to Sept 30, 2010	15
Y26BOXEC09-downstream	Aug 11 to Sept 30, 2010	15
Y27LBVRC02	Aug 30 to Sept 25, 2008 AND Aug 29 to Oct 8, 2010	15
Y27LBVRC04	Aug 30 to Sept 24, 2008 AND July 29 to Oct 8, 2010	15
M22CTWDC03	July 22 to July 24, 2003	30
M22BSPRC10	Aug 17 to Aug 20, 2003	30
Y27LBVRC12	Aug 30 to Sept 25, 2008	15
Y27LBVRC01	July 28 to Sept 24, 2009	15

We then estimated false-positive and false-negative rates, and made comparisons to the reference data, using the datasets above. Data were aggregated to create two basic groups (ratings 1, 2 = nutrient un-impacted; ratings 3, 4 = nutrient impacted), and 65 observations were then randomly drawn for false positive/negative analysis (35 from the un-impacted group, 30 from impacted group). Results are shown in **C2-2** below.

[19] Dr. Robert Bramblett (MSU fishery biologist; personal communication, March 11, 2010) provided species counts and IBI scores for the two sites Box Elder Creek's score was quite good (77) Dr. Bramblett noted that the Little Beaver Creek site's IBI was being reduced (score 55) by the presence of northern pike; this non-native predatory fish plays a large role in reducing metric scores in the Bramblett IBI He noted that the site's habitat was simple and northern pike were crowded in with their prey, but otherwise, the Little Beaver Creek site seemed healthy.

Table C2-2. Statistics Associated with DO Delta Threshold

DO Delta Threshold	% of all reference-site deltas > threshold	Estimated false positive rate*	Estimated false negative rate†
≥ 6.0	3%	9%	77%
≥ 5.3	**10%**	**23%**	**63%**
≥ 5.0	15%	26%	60%
≥ 4.0	26%	54%	53%

* The probability that a truly un-impacted site is found to have a DO delta value great than the threshold.
† The probability that a truly impacted site is found to have a DO delta value less than the threshold

The change-point threshold (DO delta of 6.0 mg/L) is probably too high for assessment, as it has a particularly high false-negative rate (77%; **Table C2-2**). Other DO thresholds between 6.0 and 4.0 mg/L were also evaluated. A DO delta threshold of 4.0 mg/L provides good balance between alpha and beta error, but its ability to determine impact is no better than a coin flip. It also allows far too many of the deltas from the reference sites to be exceedances (in fact, almost all observations from one reference site are >4.0 mg/L). The reality is, sites with excess nutrient problems do not manifest high DO deltas every single day throughout the summer, due to the vagaries of clouds, weather, and wind, which means that as an assessment tool DO delta will inherently have high false-negative rates. We selected 5.3 mg/L as the threshold for these reasons:

- It has false positive and negative rates comparable to what was found for the diatom-based nutrient increaser metrics applicable to this region(Teply, 2010b)
- It is very close to the lower bound of the 90% confidence interval (i.e., DO delta of 5.5 mg/L) of the change-point, as determined from the change-point analysis
- It keeps the proportion of reference-site data exceeding the threshold to no more than 10%
- It is in fairly good agreement with the threshold recommended by Minnesota for their plains region (i.e., DO delta of 5.0 mg/L)to protect fish and aquatic life

C.2.1 Instantaneous DO Monitoring in Wadeable Plains Streams

DEQ assessment can continue to rely on instantaneous measurements of DO. The following guidelines are recommended for instantaneous DO data collection.

When to Measure, Minimum: Without question the best time to measure the lowest daily DO is at dawn. DO in streams and standing waters is usually at its daily low just before sunrise (e.g., (Odum, 1956; Teply, 2010b; Boyd, et al., 1978; Madenjian, et al., 1987; Quinn and Gilliland, 1989). DO measurements in streams at other times of the day usually cannot give a reliable estimation of the nighttime low, especially in plains streams, because numerous other factors (e.g., wind speed and direction, air temperature, the stream's sediment oxygen demand, presence/absence of aquatic macrophytes) play a role in the rate of DO decline per unit time. Simple models incorporating various environmental factors have been used to estimate dawn DO in aquaculture ponds (e.g., (Boyd, et al., 1978; Madenjian, et al., 1987), and more sophisticated models can simulate diel DO cycles in streams and rivers (e.g., QUAL2K; (Chapra, et al., 2008). However, numerous input variables are required to run these models, making these impractical approaches for routine stream DO assessment. Therefore, we recommend dawn DO measurements be taken. We examined a number of plains stream diel DO plots, including both reference and non-reference sites, and found that **the most appropriate time window for capturing the DO daily minima is between dawn (or pre-dawn) and 8:00 am.** This time frame should be adhered to when sampling during the summer growing season (June 16th to September 30th).

The assessor should make notes of weather conditions at the time (approximate wind speed and direction, cloud cover).

When to Measure, Maximum: The DO concentration maximum in flowing streams usually occur after solar noon, commonly around 4:00 pm. The combined effects of plant/algae respiration, plant/algae primary production (which peaks around solar noon), and flow and re-aeration influences on DO saturation result in the typical sinusoidal DO patterns observed each day (Odum, 1956; Chapra and Di Toro, 1991). We examined the continuous recordings of DO for plains stream (both flowing and intermittent), and found that most daily DO peaks occurred between 2:30 pm and 5:00 pm. Time of year appeared to have no discernable effect, and the exact timing of the DO peak seemed more influenced by local factors (probably clouds, and wind velocity and direction). **We recommend that monitoring for the DO maximum occur between 2:30 pm and 5:00 pm.** Measurements need not be taken continuously during that period; checking stream DO every 15-30 minutes should be sufficient to catch the peak. You may need to stay somewhat beyond 5:00 pm if the values are still climbing. DEQ's main hand-held instrument for DO measurement is the YSI 85. This instrument has a 50 reading, manual-entry memory which can be used for collecting daily DO maximums. Set the instrument up in situ and then leave it on between 2:30 pm and 5:00 pm; record readings every 15-30 minutes by depressing the ENTER button for two seconds. Data may be downloaded later.

For the purpose of calculating DO delta, at least 3 DO sampling events should be taken in each assessment reach. Temporal independence of DO measurements is not a concern, since DO delta can be quite variable on a day-to-day basis. Each sampling event DO delta can considered on its own merits. Therefore, there is no reason to wait for 30 days to collect a subsequent DO measurement at a site. The assessor may collect DO data each day while they are in the area. This will also help increase the number of sampling events collected from an assessment reach.

C.3.0 Biochemical Oxygen Demand

Biochemical oxygen demand is one of the oldest water quality assessment tools, first recommended for use by the English Royal Commission on Sewage Disposal in the early 1900s (Hynes, 1966). It is a standardized test carried out over 5 days that measures the amount of putrescible material in water, which consumes oxygen as it decomposes. It is also one of the required measurements for wastewater nationally under the National Secondary Treatment Regulations (40 CFR part 133).

Montana has no standard for ambient BOD_5 in streams (although wastewater facility effluent and mixing zones are held to BOD_5 requirements). Nevertheless, the following guidelines for BOD_5 are commonly followed in many parts of the world:

- 1-2 mg BOD_5/L: Very clean water, little biodegradable waste
- 3-5 mg BOD_5/L: Moderately clean water, some biodegradable waste
- 6-9 mg BOD_5/L: Many bacteria, much biodegradable matter
- ≥10 mg BOD_5/L: Very bad, large amounts of biodegradable wastes in the water

The method used at the DPHHS Environmental Laboratory currently has a detection limit of 4 mg BOD_5/L (which coincides with the Royal Commission's recommendation that 4 mg BOD_5/L not be exceeded; Hynes, 1966). In plains streams we have found that otherwise healthy streams (i.e., reference sites) can have values in the 6-9 range fairly often. We recommend a value of 8.0 mg/L as a threshold for concern in plains streams.

APPENDIX D - REFERENCES

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The nutrient stoichiometry of benthic microalgal growth: Redfield proportions are optimal

Abstract—Cellular nutrient ratios are often applied as indicators of nutrient limitation in phytoplankton studies, especially the so-called Redfield ratio. For periphyton, similar data are scarce. We investigated the changes in cellular C : N : P stoichiometry of benthic microalgae in response to different levels and types of nutrient limitation and a variety of abiotic conditions in laboratory experiments with natural inocula. C : N ratios increased with decreasing growth rate, irrespective of the limiting nutrient. At the highest growth rates, the C : N ratio ranged uniformly around 7.5. N : P ratios $<$13 indicated N limitation, while N : P ratios $>$ 22 indicated P limitation. Under P limitation, the C : P ratios increased at low growth rate and varied around 130 at highest growth rates. For a medium with balanced supply of N and P, an optimal stoichiometric ratio of C : N : P = 119 : 17 : 1 could be deduced for benthic microalgae, which is slightly higher than the Redfield ratio (106 : 16 : 1) considered typical for optimally growing phytoplankton. The optimal ratio was stable against changes in abiotic conditions. In conclusion, cellular nutrient ratios are proposed as an indicator for nutrient status in periphyton.

The chemical composition of oceanic seston is known to be relatively constant at a C : N : P ratio of 106 : 16 : 1 (Redfield 1958; cf. Copin-Montegut and Copin-Montegut 1983).

This biogeochemical ratio became widely known as the "Redfield ratio" and was subsequently physiologically interpreted for aquatic organisms. Droop (1974, 1975) investigated the nutrient content of phytoplankton within different limitation scenarios and developed the cell quota growth model. Internal nutrient ratios are equivalent to carbon-based cell quotas. Nitrogen and phosphorus supply supporting maximum growth rate was shown to lead to phytoplankton stoichiometry resembling the Redfield ratio (Goldman et al. 1979; Elrifi and Turpin 1985), and the internal nutrient ratios were proposed as an indicator of algal nutrient status (Healey and Hendzel 1980; Flynn 1990). Despite some criticism (Ryther and Dunstan 1971; Tett et al. 1985), biomass stoichiometry has been widely applied to assess nutrient supply to phytoplankton in marine (Paasche and Erga 1988; Burkhardt and Riebesell 1997) and freshwater studies (Sommer 1991*a;* Hecky et al. 1993). It should be noted that C : N : P ratios close to the Redfield ratio do not indicate the absence of light limitation (Tett et al. 1985). They just indicate that neither N nor P are limiting factors of growth (Goldman 1986).

The use of nutrient ratios as an index of nitrogen or phosphorus limitation was also recommended for benthic microalgae (Borchardt 1996), because of the close phylogenetic relationship between benthic and pelagic microalgae. However, in benthic studies, it has seldom been applied to date (Engle and Melack 1993; Rosemond 1993; Rosemond et al. 1993; Hillebrand and Sommer 1997). The infrequent use may be because the relationship between cellular stoichiometry and growth rate for marine microphytobenthos has not yet been tested experimentally. Kahlert (1998) recently reviewed literature data from freshwater periphyton and found that C : N : P ratios are a reliable tool for the assessment of the nutrient status of benthic algae, proposing an optimum ratio of 158 : 18 : 1.

In the present study, we wanted to answer the following questions: (1) Is there a consistent relationship between benthic microalgal growth rates and cellular stoichiometry, and (2) Is this relation independent of abiotic conditions?

To investigate the response of internal nutrient ratios to changes in nutrient regimes, we used a semicontinuous dilution of culture media combined with sampling in intervals. We used natural inocula of algae to simplify comparison to natural assemblages. The algae for the inocula were scraped from an artificial substrate in the Kiel Fjord, Western Baltic Sea, 10 d before the experiments were started. The algae were cultured in unenriched filtered seawater under the same abiotic conditions as the experimental treatments (Table 1). At the beginning of the experiments, 1 ml of the inoculum was added to each treatment. The experiments were conducted in flat-bottom, transparent, polystyrene culture flasks with 30 ml total medium content. The algae grew as a biofilm on the bottom of the flasks, which were shaken once daily. This biofilm was a dense monolayer of cells lacking an overstory. It can therefore be assumed that CO_2 was available in excess.

The media used in the experiments consisted of organism-free–filtered seawater (0.2-μm cellulose-acetate filters) from the same location, enriched with nutrients and trace metals. The balanced medium (designated V) contained 80 μmol liter^{-1} N (as $NaNO_3$), 80 μmol liter^{-1} Si (as $Na_2O_3Si \times 5H_2O$), and 5 μmol liter^{-1} P (as $Na_2HPO_4 \times 2H_2O$), resulting in a medium N : P ratio of 16. For the N-limited medium (designated N_{lim}), the nitrogen concentration was reduced to 10 μmol liter^{-1}. For the Si-limited (Si_{lim}) media, Si was reduced to 10 μmol liter^{-1}; for the P-limited medium (P_{lim}), no phosphate was added, resulting in 1.0 μmol liter^{-1} P. The experiments were conducted in autumn 1997 and spring 1998. In the autumn experiment, four different media were applied, and the treatments consisted of an alteration of dilution rate (Table 1). In spring 1998, three different media were used, and the temperature was altered (Table 1). The two experiments differed furthermore in the taxonomic composition of the inoculum and in the light intensity, which was measured with a LiCor LI 189 (Table 1). Each treatment was conducted in triplicate, resulting in 24 cultures in autumn (four media $\times$ two dilution rates $\times$ three replicates) and 27 cultures in spring (three media $\times$ three temperatures $\times$ three replicates).

Thrice a week, the algae were counted alive employing an inverted microscope (Leitz DMIRB) at $\times$630 magnification. Up to 400 cells were counted per sample. To compare

Table 1. Treatments in limitation experiments with natural algal inocula. The table lists the code of the experiment, the type of dilution rate, the duration of the experiment, the light intensity, the temperature, and the media used.

Code	Dilution rate (d^{-1})	Time	Light (μmol m^{-2} s^{-1})	Temp. (°C)	Applied media
H (high dilution)	0.5	22 Sep–10 Nov 97	21	14	V, N_{lim}, P_{lim}, Si_{lim}
L (low dilution)	0.07	Ditto	21	14	V, N_{lim}, P_{lim}, Si_{lim}
C (cold temp.)	0.5	19 Jan–9 Mar 98	35, 8	5	V, N_{lim}, P_{lim}
M (medium temp.)	0.5	Ditto	34, 7	14	V, N_{lim}, P_{lim}
W (warm temp.)	0.5	Ditto	32, 2	19	V, N_{lim}, P_{lim}

the different species, which span several size classes, biovolume was calculated by fitting appropriate geometric models (Hillebrand et al. submitted). In addition, thrice a week, a sample was taken with a sterile pipette from the bottom of the flask. The sample volume corresponded to the daily dilution, i.e., 15 ml for treatments C, M, W, and H (for codes, *see Table 1*) and 5 ml for treatment L. The samples were divided and filtered on precombusted Whatman GF/C filters for analyses of particulate CN and P, respectively. Particulate phosphate was determined as orthophosphate after a combined digestion with heat and acid (Hillebrand and Sommer 1997). Since this analysis needs a high amount of material, the three replicates of one treatment had to be pooled. Particulate carbon and nitrogen were measured with a Fisons CN-Analyzer (NA 1500N).

The experiments were divided into two phases. After 28 d, the daily medium dilution was stopped to enforce a stronger limitation by decreasing the supply with new media. Sampling was carried out once a week. After sampling, the total volume of the cultures was filled up to 30 ml again. The L treatment was stopped at day 28, but the other experiments were conducted until day 49.

The daily growth rates μ were calculated from the total biovolume of each replicate according to Eq. 1:

$$\mu = \frac{\ln B_2 - \ln B_1}{t_2 - t_1} \quad (1)$$

including

$$B = \text{biovolume.}$$

$$t = \text{time.}$$

The ratios of C : N, N : P, and C : P, respectively, were calculated on a molar basis. They were compared to the positive daily growth rates by a three-parameter exponential equation (Eq. 2). The use of ratios in regression analysis is not with-

Table 2. Fit of exponential function (Eq. 2) to C : N, C : P, or N : P ratio dependent on μ. The table lists treatments and media, ratio for which the regression was calculated, number of data, coefficient of determination r^2, and parameter estimates, as well as the F-ratio between regression explained mean squares and residual mean square (Sokal and Rohlf 1995). Significance level: * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$; ns, not significant. n.r., no regression (estimation procedure did not converge).

Treatment	Med	Ratio	n	r^2	a	b	c	$F_{(2;n-3)}$
All	All	C : N	322	0.1669	7.5 ± 0.7	1.8 ± 0.1	−6.8 ± 2.6	32.04***
All	V	C : N	102	0.3692	7.2 ± 0.6	1.9 ± 0.1	−9.9 ± 4.2	28.97***
All	P_{lim}	C : N	98	0.362	5.9 ± 0.9	1.8 ± 0.2	−5.0 ± 2.3	26.96***
All	N_{lim}	C : N	96	0.1118	7.5 ± 2.6	2.0 ± 0.2	−3.1 ± 2.5	5.78**
All	Sl_{lim}	C : N	26	0.1952	8.5 ± 0.6	1.8 ± 0.6	−27.5 ± 18.3	10.18**
C	All	C : N	77	0.3847	4.4 ± 2.4	2.2 ± 0.2	−3.3 ± 1.9	23.14***
M	All	C : N	63	0.2253	6.8 ± 2.3	2.3 ± 0.3	−6.0 ± 4.3	7.99**
W	All	C : N	63	0.2675	7.6 ± 0.8	2.6 ± 0.3	−26.2 ± 11.2	10.96**
H	All	C : N	89	0.1796	8.4 ± 0.4	1.3 ± 0.2	−12.1 ± 7.3	9.42***
L	All	C : N	30	n.r.	—	—	—	—
All	All	C : P	122	0.0655	143.6 ± 48.5	5.1 ± 0.3	−8.2 ± 10.6	4.17*
All	V	C : P	39	0.2380	119.0 ± 50.7	5.1 ± 0.4	−5.1 ± 4.2	3.28*
All	P_{lim}	C : P	37	0.4306	139.2 ± 47.8	6.1 ± 0.2	−17.5 ± 8.0	12.85**
All	N_{lim}	C : P	36	n.r.	—	—	—	—
All	Sl_{lim}	C : P	10	0.4067	51.2 ± 53.8	4.9 ± 0.5	−8.6 ± 10.4	2.40 ns
C	All	C : P	23	0.2076	80.3 ± 61.9	5.0 ± 0.5	−5.8 ± 6.9	3.54 ns
M	All	C : P	20	0.2618	73.3 ± 120.4	5.7 ± 0.4	−5.4 ± 6.0	2.23 ns
W	All	C : P	21	0.1655	116.7 ± 27.4	4.6 ± 0.6	−20.7 ± 22.0	1.79 ns
H	All	C : P	26	n.r.	—	—	—	—
L	All	C : P	18	0.1366	216.1 ± 105.0	5.8 ± 0.7	−15.6 ± 15.0	1.17 ns
All	P_{lim}	N : P	37	0.2668	20.9 ± 2.6	3.1 ± 0.3	−24.7 ± 14.6	6.50**

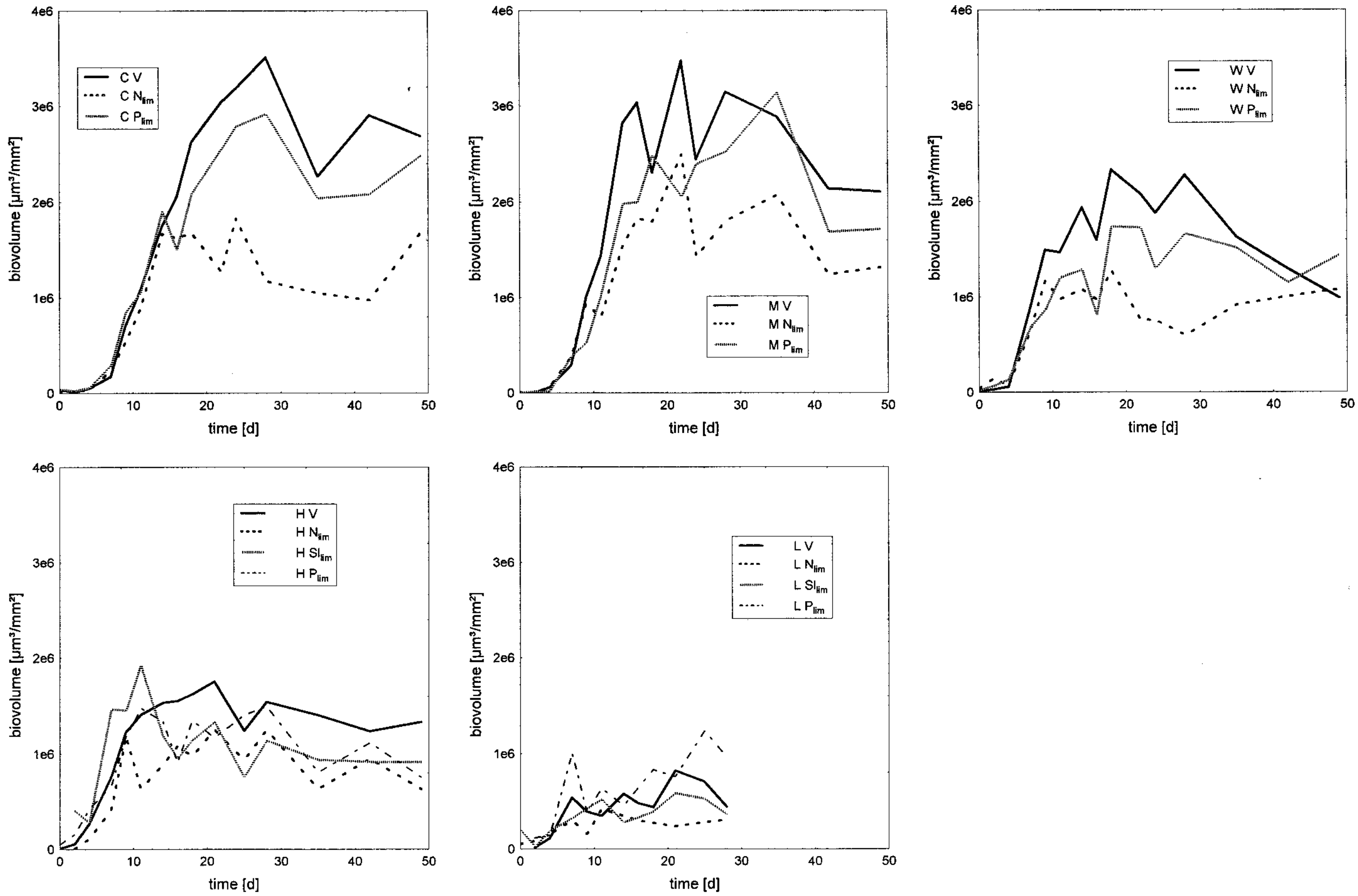

Fig. 1. Development of mean of total biovolume ($\mu m^3/mm^2$) of benthic microalgae over time for all treatments and media of experiments from autumn and spring.

out difficulties (Sokal and Rohlf 1995), but normal distribution of the data of all ratio variables was affirmed ($P > 0.05$, Kolmogorov-Smirnov test, Statistica 5.1).

$$R = a + e^{(b+c*\mu)} \quad (2)$$

with

R = molar ratio of C : N, N : P, and C : P, respectively.

μ = daily growth rate.

a, b, c = parameters.

With a negative estimate for c, the resulting curve decreases and approaches asymptotically to a horizontal line with $y = a$. Therefore, parameter a can be taken as an estimate of the optimal ratio. The validity of the regression model was estimated by an analysis of variance comparing the model-explained variance with the residual variance (Sokal and Rohlf 1995). The presence of a global convergence minimum in the regression procedure was affirmed by using different software and different estimation procedures (Statistica 5.1 and Statgraphics 6.1).

Total biovolume of the treatments followed a sigmoidal curve in all cases, explaining 87.03% ± 12.62 (mean ± SD) of the variance (Fig. 1). Final biovolumes were generally higher at lower temperature and lower in the N_{lim} treatments. Moreover, the spring experiments resulted in higher final biovolume than the autumn experiment; the latter again was divided in lower biovolume in L treatments compared to H treatments (Fig. 1). Daily growth rates increased in the beginning of the experiment and decreased afterwards, varying around zero after attainment of the carrying capacity (Fig. 2). In the autumn experiments, 30 species were present, representing the Bacillariophyceae, Chlorophyceae, and cyanobacteria. In spring, 30 species represented the Bacillariophyceae, cyanobacteria, and Rhodophyceae. Almost all treatments were dominated by diatoms (unicellular, chain-forming, and tube-dwelling), whereas nonheterocystous cyanobacteria became codominant at 19°C temperatures and high N content.

C : N ratios increased with time (significant regression slopes over time, $P < 0.05$, except for LV, LN_{lim}, and HSI_{lim}), up to 20 in P_{lim}, up to 23 in V, and up to 45 in N_{lim} cultures, respectively. This increase reflected the increasing strength of nutrient limitation. The ratio of C : N decreased exponentially with increasing growth rate (Fig. 3; Table 2), independent of treatment and limiting nutrient. The regression model described the relationship between C : N ratios and growth rates significantly (F-ratio, $P < 0.01$; Table 2) for all media and treatments, except treatment L. The optimal ratio at high growth rates, estimated by parameter a of the regression, is

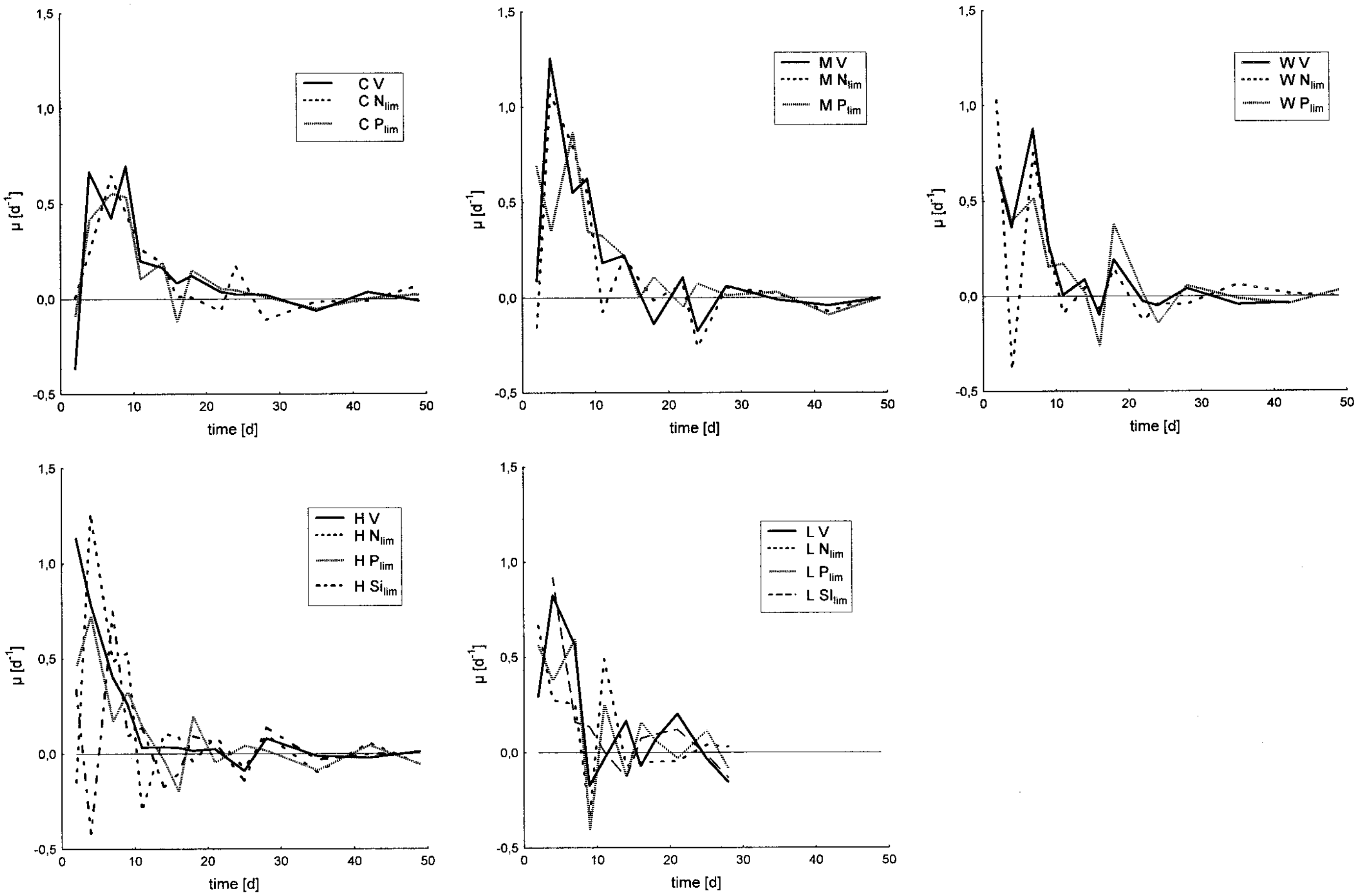

Fig. 2. Mean of daily growth rate (Eq. 1) in relation to time.

plotted for the different treatments and media in Fig. 4, compared to their standard error (except for the L treatment). The estimates of the optimal C : N ratios showed widely overlapping confidence intervals (Fig. 4). A difference between the optimal C : N ratios of different treatments could therefore not be sustained; thus, C : N ratios increased irrespective of the limiting nutrient.

The three-parameter exponential regression on C : P data was significant at $P < 0.01$, only for the P_{lim} media, and at $P < 0.05$ also for V and the pooled data (Table 2). In an Si- or N-limited situation, there was no consistent relation between C : P ratios and growth rate. Only under P_{lim} conditions did the N : P ratio increase exponentially with decreasing growth rate (Table 2). N : P ratios in the stationary phase ($\mu < 0.1$) were significantly different when comparing different media ($P < 0.05$; Kruskal-Wallis analysis of variance [ANOVA] on ranks; all pairwise multiple comparison by Dunn's method, Sigma Stat) but not between different abiotic conditions (Fig. 5). The mean of the N : P ratio was significantly lower at N_{lim} (10.69) compared to P_{lim} (33.62), while the cellular N : P was intermediate at balanced medium N : P (for V 20.58, for Si_{lim} 14.22).

Conclusively, these results show that high C : N ratios were due to nutrient limitation in general, while C : P and N : P ratios increased significantly only with P limitation.

Our data support the view that biomass stoichiometry is an indicator of nutrient status for benthic microalgae as for phytoplankton, for which it was developed experimentally and conceptually (Droop 1974, 1975; Healey 1978; Healey and Hendzel 1980). Similar to our results, the C : P and N : P ratios of pelagic microalgae increased with decreasing growth rate under P limitation, while the C : N ratio increased with decreasing growth rate under P and N limitation. Moreover, the increase of C : N at low growth rates was higher under N limitation than under P limitation in our data and in phytoplankton studies (Perry 1976; Sakshaug and Holm-Hansen 1977; Goldman et al. 1979; Healey and Hendzel 1980). Under N limitation, the C : P and N : P ratios were found to decrease with decreasing growth rates (Elrifi and Turpin 1985). It can be concluded from these data that C : N ratios are applicable as general indicators of limitation, while C : P and N : P ratios allow indication of P vs. N limitation.

The criticism concerning the indicator value of the nutrient ratios pointed mainly at light-limited conditions (Tett et al. 1985; Wynne and Rhee 1986), leading to the conclusion that light limitation was not reflected by biomass stoichiometry. In agreement with Goldman (1986), the optimal ratios emerging from our autumn experiment with lower light intensities did not differ from the ratios from the spring experiment (Fig. 4). This contradicts Wynne and Rhee (1986), who described changes in optimum N : P ratios in planktonic algae caused by changes in light intensity and wavelength.

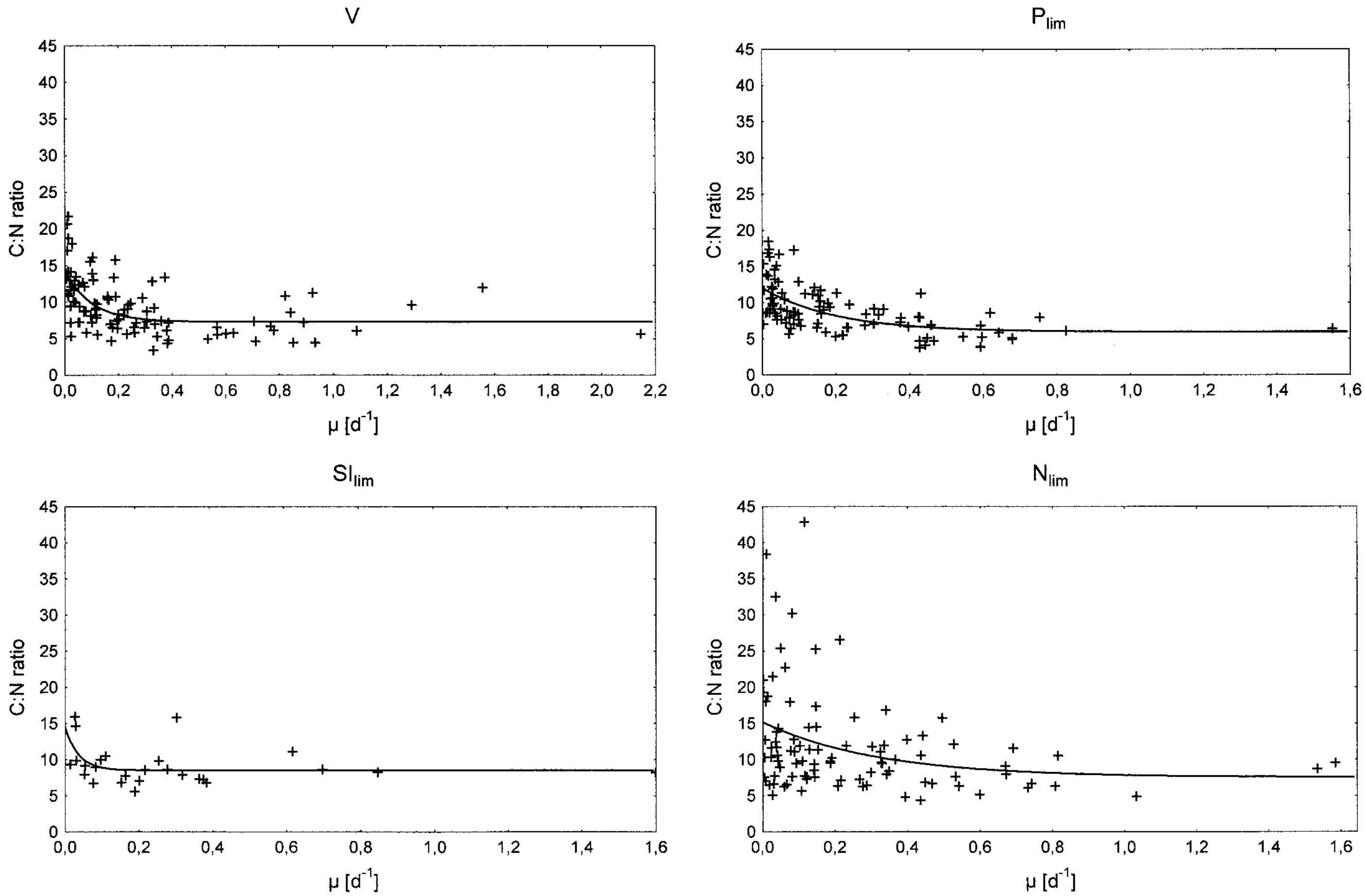

Fig. 3. C:N ratios of benthic microalgae dependent on growth rate, plotted for different media. Lines represent nonlinear fits of Eq. 2. Parameter estimates are presented in Table 2.

However, they calculated the optimum N:P ratio from minimal cell quotas, which refer to the stationary (i.e., limited) phase of cultures ($\mu = 0$). This is a contradiction to the concept of optimal ratios ($\mu = \mu_{max}$), as becomes evident, if we approach our data in the same way. The minimal cell quotas (q_0) of N and P can be calculated from Eq. 2 by substituting $\mu = 0$ and inserting the estimates of *a* and *b* from Table 2. The mean ratio of q_0N:q_0P was 18.56 for all treatments, whereas the ratio for P$_{lim}$ was 48.8. The latter ratio reflects the nutrient-limited situation and not an optimal ratio. Besides, these calculations show that cell quotas of benthic microalgae (ranges of q_0 are 0.060–0.083 mol N mol^{-1}C and 0.002–0.008 mol P mol^{-1}C, respectively) are within the same order of magnitude as those reported from freshwater phytoplankton (Sommer 1991*a*,*b*).

Based on the work of Healey and Hendzel (1980), Hecky et al. (1993) suggested limits of indicating values for phytoplankton nutrient ratios. Moderate N limitation was indicated by C:N > 8.3 and severe limitation > 14.6, whereas moderate P limitation was indicated by C:P > 129 and severe limitation by C:P > 258 and N:P > 22.

From our data, an optimal ratio of fast-growing periphyton can be determined in analogy to the optimal phytoplankton ratios, based on the estimates of *a* (Eq. 2) for the balanced experiments (medium V). This results in a C:N:P ratio of 119:17:1. However, it seems to be more useful to give ranges of ratios in nonlimited conditions. These ranges were calculated from the estimates for the different media (V, P$_{lim}$, N$_{lim}$, and Si$_{lim}$; Table 2), adding the standard error to the maximum estimate and subtracting it from the minimum. The optimum ranges are 5–10 for C:N and 90–185 for C:P. N:P ratios between 13 and 22 indicate balance between nitrogen and phosphate. These ranges allow the establishment of C:N:P ratios as indicators of nutrient limitation (Fig. 6). With an N:P ratio < 13 and a C:N ratio > 10, the periphyton can be assigned N limited. With an N:P ratio > 22 and a C:P ratio > 180, the microbenthic assemblage is P limited.

These ratios are more similar to the Redfield ratio and the limits given by Hecky et al. (1993) for phytoplankton than to the optimal ratio proposed for freshwater periphyton by Kahlert (1998). Based on a literature survey, she found an optimum ratio for freshwater periphyton slightly higher than the Redfield ratio (C:N:P = 158:18:1) and proposed much higher indicators of limitation than Hecky et al. (1993): C:P > 369 and N:P > 32 for P limitation and C:N > 11 and N:P < 12 for N limitation. The reason for the discrepancy is probably because macrophytes (e.g., *Cladophora*) were included in the data survey, which are known to differ in their C:N:P ratios. Atkinson and Smith (1983) found a median of C:N:P of 550:30:1 in a comparison of marine

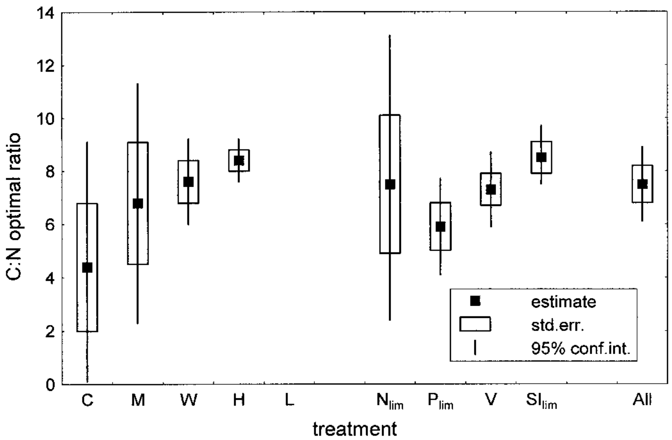

Fig. 4. Optimal C : N ratios of different treatments (M–L) and media (N_{lim}–Si_{lim}). Optimal C : N ratios are presented as estimate of parameter *a* (Eq. 2), its standard error, and confidence interval. For treatment L, no optimal ratio could be fitted. ''All'' represents the optimal ratio fitted on all data.

macroalgae, which are supposed to contain more structural carbon. In their data, however, the Redfield ratio posed a lower limit of the found ratios, which may be reached only in very nutrient-rich localities like Antarctic waters (Weykam et al. 1996). C : N : P ratios were reported from studies dealing with microphytobenthos before, but they were reported without relating them to growth rates. The ratios were in the range reported here (Reuter et al. 1986; Engle and Melack 1993; Rosemond 1993; Rosemond et al. 1993; Turner et al. 1994; Vymazal and Richardson 1995; Hillebrand and Sommer 1997). Therefore, the limits for nutrient ratios proposed by us can be applied for periphyton dominated by diatoms and cyanobacteria.

A note has to be added in proof: Biomass stoichiometry of natural benthic communities may be influenced by a high proportion of detritus or by carbon limitation, both of which were excluded in our experiments. The influence of detritus was found to be less than expected for phytoplankton (Healey and Hendzel 1980), but it has to be evaluated for periphyton (cf.

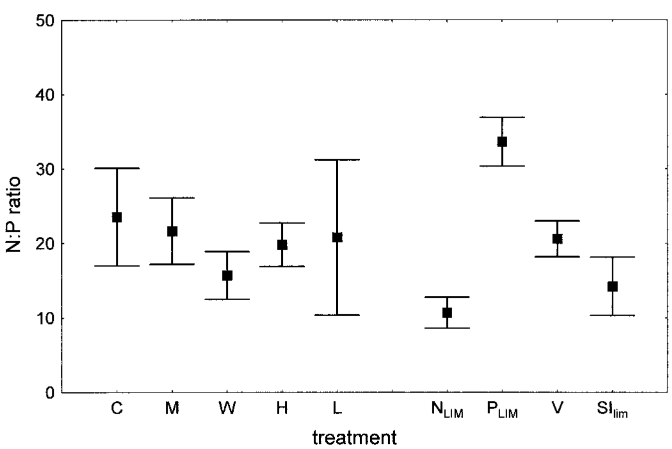

Fig. 5. N : P ratios of stagnant cultures of different treatments and nutrient content, depicted as mean ± SE of all sample dates with $\mu < 0.1$.

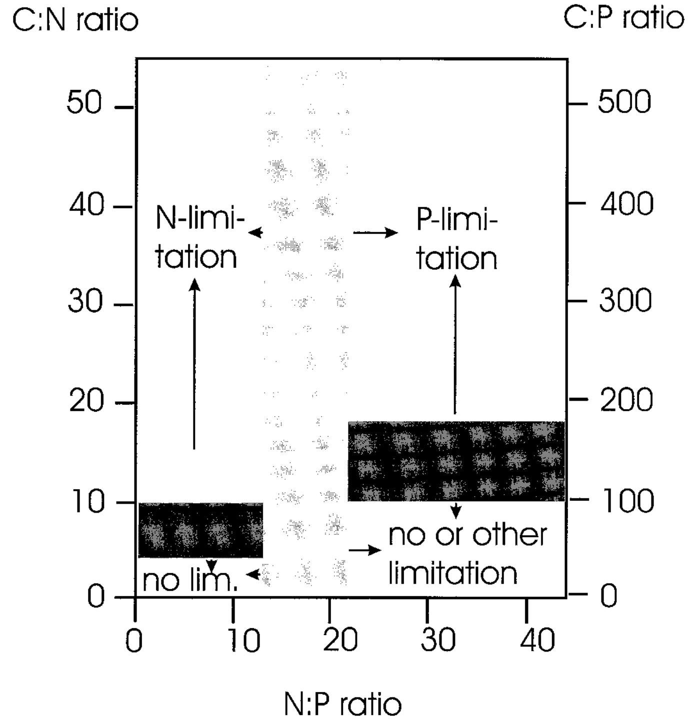

Fig. 6. Schematic diagram on the use of nutrient ratios as indicator of nitrogen or phosphorus limitation. Shaded bars represent the range of optimal ratios. Further explanation is given in the text.

Kahlert 1998). Carbon limitation was shown to change C : N : P ratios in a marine pelagic diatom, by decreasing C : P and N : P ratios and increasing C : N ratios (Burkhardt and Riebesell 1997). Furthermore, it should be noted that in dense biofilms, the algae could have different access to nutrients. This depends on whether these stem from the water column or from sediment pore water, so that emerging nutrient stoichiometry may be the mean of a gradient.

In conclusion, cellular nutrient ratios are a useful approach for the detection of nitrogen or phosphorus limitation in benthic microalgae as well as in phytoplankton. The following constraints should be observed: The cellular C : P ratio is an index of P limitation, the cellular C : N ratio indicates limitation in general, and the N : P ratio distinguishes between N or P limitation.

Helmut Hillebrand
Ulrich Sommer

Institut für Meereskunde
Abteilung Meeresbotanik
Düsternbrooker Weg 20
D-24105 Kiel
Germany

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ENVIRONMENTAL QUALITY

Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met by Entities in the Private Sector in 2011/2012

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Prepared by:
Water Quality Planning Bureau, Water Quality Standards Section
Montana Department of Environmental Quality
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

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ACRONYMS

Acronym	Definition
AFDW	Ash Free Dry Weight
AWTF	Advanced Water Treatment Facility
BAT	Best Available Technology
BBER	Bureau of Business and Economic Research
BNR	Biological Nutrient Removal
BOD	Biochemical Oxygen Demand
BPD	Barrels Per Day
CAFO	Concentrated (or Confined) Animal Feed Operations
DEQ	Department of Environmental Quality (Montana)
EBNR	Enhances Biological Nutrient Reduction
EPA	Environmental Protection Agency (US)
FO	Forward Osmosis
GDP	Gross Domestic Product
GHG	Green House Gas
IR	Integrated Report
MCA	Montana Code Annotated
MCF	Million Cubic Feet
ME-RO	Microfiltration-Reverse Osmosis (ME-RO)
MF	MicrofiltrationF
MGD	Million Gallons Per Day
MPDES	Montana Pollutant Discharge Elimination System
MT DEQ	Montana Department of Environmental Quality
MW	Megawatts
NAICS	North American Industry Classification System
NF	Nanofiltration
NPDES	National Pollutant Discharge Elimination System
OG	Oil and Gas
REC	Reclamation Division (DEQ)
RO	Reverse Osmosis
SB	Senate Bill
SMC	Stillwater Mining Company
TDS	Total Dissolved Solids
TKN	Total Kjeldahl Nitrogen
TMDL	Total Maximum Daily Load
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
ULPRO	Ultra-Low Pressure Reverse Osmosis (ULPRO)
UV	Ultraviolet
WRP	Watershed Restoration Plan
WTF	Water Treatment Facility
WWTP	Wastewater Treatment Plant

EXECUTIVE SUMMARY

This document quantifies the costs to affected Montana businesses of meeting the base numeric nutrient standards today, given the current state of treatment technology and the current economic status of the state. This paper demonstrates the substantial economic and social impacts of meeting nutrient criteria to 50 or so affected businesses in Montana and to those who depend upon the businesses for jobs, purchases, commodities, secondary spending, etc. It also looks at widespread effects on the state of Montana as a whole. This document provides the analyses and conclusions of the Montana Department of Environmental Quality's (DEQ) supporting the statute language (§75-5-313[5], MCA (2011) that all private dischargers are, at the present time, eligible for a general variance from the nutrient standards based on "Substantial and Widespread" economic impacts.

The EPA's Interim Economic Guidance for Water Quality Standards (U.S. Environmental Protection, 1995) offers steps that can be taken to determine substantial and widespread impacts of water quality standards on both public wastewater treatment plants and on private businesses. The guidance for public wastewater treatment plants is fairly straightforward, and was used by DEQ to demonstrate substantial and widespread impacts on the municipalities having to meet standards for those plants. The private guidance is not as straightforward and does not provide direct thresholds for the 'substantial' determination, as does the public guidance.

Therefore, this demonstration takes parts of the EPA Guidance and makes it part of a larger evaluation for assessing substantial and widespread impacts for private businesses in Montana. For the purposes of this demonstration, 'substantial impacts' refers to financial and other impacts on affected businesses, and 'widespread impacts' refers to ripple effects within Montana from the business impacts. The widespread impacts were looked at both locally (e.g. the effect of a business closing on the town it resides in) and statewide (e.g. overall impacts on Montana taxes, energy supply). The major steps for this evaluation included the following: 1) define the businesses that would be affected, 2) define both the current treatment level for nutrients and the applicable criteria for each business, 3) estimate the costs of meeting the applicable base numeric criteria for each affected business, 4) estimate the financial impacts of these costs on the businesses themselves, and 5) estimate the widespread ripple effects from the business impacts.

This demonstration is based upon best available information as it relates to each major step of the analysis. In addition, a sensitivity analysis was made around the estimated costs. It is DEQ's best professional judgment that the resulting costs of requiring immediate compliance with the base numeric nutrient criteria would result in substantial costs beyond what individual firms can internalize, and that this in turn would lead to adverse widespread impacts in Montana.

BACKGROUND

The Montana Department of Environmental Quality (DEQ) began developing numeric nutrient standards for state surface waters in 2001. A field pilot study was undertaken from 2001-2003 to identify and refine approaches for developing the criteria in the plains region of the state. Work from 2003-2008 focused on the selection of an appropriate zoning system by which the criteria would be applied, collection of data from reference streams to help with criteria derivation, and identification of harm-to-use thresholds for beneficial water uses that nutrients affect. During this same period DEQ undertook a focused data collection to support the QUAL2K water-quality model which was then used to develop numeric nutrient criteria for a large river (lower Yellowstone). In addition, DEQ collected data to support lake nutrient standards (this work in ongoing, as are other field projects intended to further refine the flowing-water criteria).

In 2008, DEQ released draft nutrient criteria for wadeable streams (Suplee, et al., 2008) and presented these to stakeholders. DEQ has subsequently refined the process by which wadeable stream criteria are derived, and is in the process of preparing those as of this writing; draft values are shown below (**Table 1**) along with draft criteria for the lower Yellowstone River. In **Table 1** and throughout this analysis, the N stands for nitrogen and the P for phosphorus. While stakeholders understand that the criteria were derived based on sound science and reflect societal values that are protective of the designated water uses, the proposed criteria are stringent (**Table 1**). As a result, the stakeholder community has been concerned about what their permit limits will be as well as the opportunities for temporary variances from the stringent limits. Many permitted businesses discharging into wadeable streams do not have instream dilution and would be required to meet the nutrient criteria end-of-pipe. This likely includes businesses on the Yellowstone River between Laurel and Billings, which are assumed to have to meet end-of-pipe standards.

Table 1. Montana Draft Nutrient Criteria

Level III Ecoregion	**Period When Criteria Apply**	**Parameter**		
		Total P (mg/L)	**Total N (mg/L)**	**Benthic Algae Criteria**
Northern Rockies	July 1 -Sept. 30	0.025	0.3	120 mg Chl *a*/m^2 (36 g AFDW/m^2)
Canadian Rockies	July 1 -Sept. 30	0.025	0.3	120 mg Chl *a*/m^2 (36 g AFDW/m^2)
Middle Rockies	July 1 -Sept. 30	0.030	0.3	120 mg Chl *a*/m^2 (36 g AFDW/m^2)
Idaho Batholith	July 1 -Sept. 30	0.030	0.3	120 mg Chl *a*/m^2 (36 g AFDW/m^2)
Northwestern Glaciated Plains	June 16-Sept. 30	0.12	1.1	n/a
Northwestern Great Plains, Wyoming Basin	July 1 -Sept. 30	0.12	1.0	n/a
Yellowstone River (Bighorn R. confluence to Powder R. confluence)	Aug 1 -Oct 31	0.09	0.70	Nutrient concentrations based on limiting pH impacts
Yellowstone River (Powder R. confluence to stateline)	Aug 1 -Oct 31	0.14	1.0	Nutrient concentrations based on limiting nuisance algal growth

(Suplee, et al., 2008)

Due to the difficulty of meeting the draft numeric nutrient criteria at the present time, Montana Senate Bill 367 was signed by Governor Schweitzer on April 21, 2011. Senate Bill 367 (now (§75-5-313, MCA) authorizes individual, general, and alternative variances. Under the general variance limits established in §75-5-313, MCA, permit limits would be established based on 1 mg/L TP and 10 mg/L TN for facilities discharging $\geq$ 1 MGD, or 2 mg/L TP and 15 mg/L TN for facilities discharging $\leq$ 1 MGD. Facilities with lagoons would be capped at their current nutrient load. Over the next 20 years, as treatment technology improves and costs come down, more stringent nutrient levels for the general variance will likely be required. As mentioned above, the present document provides DEQ's demonstration supporting the statute language that all private dischargers are, at the present time, eligible for a general variance from the nutrient standards based on "Substantial and Widespread" economic impacts.

MONTANA'S PRIVATE BUSINESSES

Out of the thousands of businesses in Montana, about 50 were identified as ones that would be affected by the nutrient criteria. Included were businesses that have a discharge permit into state waters, and are not otherwise hooked up to a municipal system. Therefore, the numeric nutrient water quality standards only apply to business entities that have a surface water discharge permit.

Of the approximately 75 private businesses with a Montana Pollutant Discharge Elimination System permit (MPDES), not all are subject to the numeric nutrient water quality standards. There are some private dischargers that would not have a reasonable potential to exceed the nutrient water quality standards because either the discharged wastewater does not contain either TP or TN, because the discharge is only non-contact cooling water or other process wastewater, or because nutrients are not a parameter of concern. The cost analysis began with a list of 74 MPDES permit numbers. Of those, 2 were for the CAFO general permit, 5 were considered terminated and sent to archives, 4 had not yet been issued, 2 were pending, and 2 others were excluded because it was hard to say what their operation consisted of. Of the remaining 59 permits within the analysis, 6 were considered to not have nutrients in their effluent, one was moving to a non-discharging system and one is a draft permit for a proposed facility. That left 51 MPDES permits subject to nutrient criteria that could be used for this demonstration.

The 51 businesses range from very large companies, employing over 1,000 people (e.g., Stillwater/East Boulder Mine), to very small, family owned businesses (e.g., Sleeping Buffalo Hot Springs). These businesses are in the following sectors:

- Metal mining (6)
- Coal Mining (9)
- Electric Generation (3)
- Oil and gas production (5)
- Refineries (4)
- Manufacturing including talc, silicon, cement, and chemicals (13)
- 'Other businesses' including hot springs, train yards, health care, sugar processing plants, livestock, and a boys and girls ranch (11)

These businesses tend to be located near Montana's seven large towns, with the largest number being in or near Billings. However, some are located in remote areas and the affected businesses are spread geographically across the state. The largest affected businesses are in the central and south central portions of Montana. The majority of businesses on the list are core Montana industries that generally pay higher than average wages, and in certain cases, supply crucial economic goods to Montanans and others out of state. The most crucial of these to the overall functioning of the Montana economy are the three affected refineries in or near Billings. They provide almost all of Montana's liquid petroleum products as well as about 50% of Spokane's and 30% of North Dakota's (Montana Department of Environmental Quality, 2012). In addition, the Stillwater mine, consisting of two primary mines, is one of the only sources of palladium and platinum in North America (although we are focusing on Montana impacts in this demonstration, rather than North American impacts). In addition, Montana's coal resources supply over 60% of Montana's electricity generation (100% of its coal-based electricity generation) and supply coal to more than 10 other states for the purpose of electricity generation.

CURRENT TREATMENT LEVELS FOR NUTRIENT REMOVAL AND APPLICABLE CRITERIA FOR EACH BUSINESS

Data Gathering

DEQ and a contractor examined the wastewater permits and the Statement of Basis for each of the 51 affected businesses. These records are located within DEQ's Permitting Division. Within each permit, DEQ collected the following information where available:

- Current level of water treatment technology
- Measured effluent data from the business (including current nutrient levels)
- Name and status of the receiving stream
- The dilution potential for the effluent given the receiving stream

From this data, DEQ and the contractor calculated the applicable nutrient criteria for each of the businesses depending upon their location in Montana, dilution potential, etc. The nutrient criteria that a particular business would have to meet would depend on how much dilution the receiving stream has for their effluent, and where the business is located in state (the specific ecoregion). For most businesses, nutrient effluent levels were not available, so DEQ used a method for many businesses to 'back out' current nutrient effluent levels. For most businesses, current effluent level (including TN and TP) was determined by the description of the current treatment system included in their NPDES permits and supplemented by past monitoring data included in their most recent permit.

To estimate costs to each business of meeting nutrient criteria, DEQ relied on a study that looked at costs associated with removing nutrients from wastewater at 5 different levels of treatment (**Table 2**; (Falk, et al., 2011) (**Appendix C**). The DRAFT Interim WERF study "*Finding the Balance Between Wastewater Treatment Nutrient Removal and Sustainability, Considering Capital and Operating Costs, Energy, Air and Water Quality and More*" (Falk, et al., 2011) looked at different levels of nutrient treatment ranging from minimal nutrient treatment (level 1) to a treatment that is close to Montana's base nutrient criteria (level 5). WERF Level 1 treatment does not directly treat N and P, whereas Level 2 treatment is about the same as the general variance levels in Montana SB 367 mentioned above. The levels of treatment and associated costs in the WERF study are presented in **Table 2**. The technologies used at each level are presented in **Table 3**.

Table 2. Effluent Quality and Associated Treatment Costs in the Interim WERF study (Falk, et al., 2011; Tetra Tech, 2011)

Level	Description	Capital Cost (million dollars per 1 MGD design flow)	Operations Cost (dollars per day per 1 MGD actual flow)
Level 1	No N and P removal	9.3	250
Level 2	1 mg/l TP; 8 mg/l TN	12.7	350
Level 3	0.1-0.3 mg/l TP; 4-8 mg/l TN	14.4	640
Level 4	<0.1 mg/l TP; 3 mg/l TN	15.3	880
Level 5	<0.01 mg/l TP; 1 mg/l TN	21.8	1370
Level 5/100% RO	<0.01 mg/TP; <1 mg/l TN	28.3	1860

Table 3. Unit Processes per Treatment Level in WERF Study (Falk, et al., 2011)

Level	Liquid Treatment	Solids Treatment	Comment
1	Primary Clarifier Activated Sludge Disinfection Dechlorination	Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Conventional Activated Sludge for BOD/TSS removal
2	Primary Clarifier Activated Sludge Alum (optional) Disinfection Dechlorination	Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Nitrification/Denitrification and Biological Phosphorus Removal
3	Primary Clarifier Activated Sludge Methanol (optional) Alum (filtration) Filtration Disinfection Dechlorination	Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Nitrification/Denitrification and Biological Phosphorus Removal and Filtration
4	Primary Clarifier Activated Sludge Methanol (optional) Alum/Polymer (Enhanced Settling) Enhanced Settling Filtration Disinfection Dechlorination	Fermentation Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Nitrification/Denitrification and Biological Phosphorus Removal, High Rate Clarification and Denitrification Filtration
5	Primary Clarifier Activated Sludge Methanol (optional) Alum/Polymer (Enhanced Settling) Enhanced Settling Filtration Microfiltration Reverse Osmosis Disinfection Dechlorination	Gravity Belt Thickener Anaerobic Digestion with Cogen Centrifugation	Nitrification/Denitrification and Biological Phosphorus Removal, High Rate Clarification, Denitrification Filtration, and MF/RO on about Half the Flow

The current treatment level distribution for those Montana dischargers with sufficient information to make a determination of level of treatment (32 of the 51 businesses) was 47% (WERF) level 1, 3% Level 2, 22% Level 3, 19% level 4, and 9% level 5 (without RO, or with RO as a backup). For businesses where the information in their permit was not adequate to make a determination, DEQ assigned treatment level 3 as a conservative estimate for the analysis (in order to lessen the chance of overestimating costs and impacts to businesses—assuming an already high level of nutrient treatment being done by businesses lessens the cost estimate of them having to meet base numeric criteria).

COSTS TO PRIVATE BUSINESSES OF HAVING TO MEET BASE NUTRIENT CRITERIA

Several tables were created that estimate the cost for each affected business of meeting base numeric nutrient criteria. **Appendix A** presents two Excel spreadsheets developed to calculate the annualized capital and operations and maintenance costs (O&M) associated with meeting the base numeric nutrient standards for each of the 51 businesses (where flow data were available). Capital and O&M costs for attaining nutrient standards were estimated from the DRAFT Interim WERF study (Falk, et al., 2011). **Appendix B** documents key underlying assumptions applied for this demonstration. In essence, the cost assumptions are mostly the same as those made in the Public demonstration.

Key Elements of the Cost Framework include the following:

- The treatment technology used to simulate costs to businesses consisted of advanced mechanical treatment combined with reverse osmosis (RO) (**Appendix D**) or Level 5 in **Table 3** (above) with 100% RO. Treatment costs included those associated with nitrification/denitrification and biological phosphorus removal, high rate clarification, and denitrification filtration. Costs were estimated from Falk, et al. (2011).
- Every business must use reverse osmosis (RO) on 100% percent of their effluent in order to get all of the base criteria. The Interim WERF study assumed RO treatment for 50 percent of effluent flow at the most stringent treatment 'level 5'. WERF Level 5 is not as stringent for N as Montana's base numeric criteria. At level 5, half of the effluent flow remained treated by processes equivalent to WERF Level 4 and the other half received an enhanced level of treatment (reverse osmosis or RO). To meet the MT base numeric nutrient criteria, DEQ calculated that the highest level of treatment is needed for 100 percent of the flow. Thus, cost estimates in this demonstration are based on providing RO treatment to 100 percent of flow. These cost estimates are thus marginally higher than WERF level 5 cost estimates (see **Table 2** below). While it may be possible that some facilities' waste streams and effluent levels do not require 100 percent RO treatment (due to dilution potential in the receiving water, and thus less stringent levels of needed treatment), simulating costs at 100 percent RO provides an upper bound estimate of the potential economic impact to businesses. The Interim WERF study data were adapted to estimate the cost of treating all flow by RO by isolating the marginal unit processes used for Level 4 and Level 5 and calculating the cost for a treatment train with 100 percent RO. (Schmidt, 2010) [1]
- The 51 businesses analyzed are mostly in economic activities of commoditized goods and services with inelastic national or global cost curves that dictate their ability to adjust to changing production costs. Therefore, parent companies, where they exist, will likely not pay to meet nutrient criteria, and this analysis looks at 'plant level' data ---that is, the effects of the base criteria on the local business rather than the larger parent company, where it exists. For example, DEQ examines the cost effects on the Billings Exxon Refinery rather than on the Exxon Mobile Corporation as a whole.
- Because most of these industries involve nationally or internationally traded commodities, costs of meeting base numeric criteria will not be primarily shifted to consumers. Rather, the private businesses themselves will have to incur the majority of costs.

[1] Schmidt, 2010 Shows that for TP, TN, and other micro-pollutants, RO was indeed the most effective method for removing TN and TP (better than membrane bioreactor, MBR). Thus, this study assumes the use of RO technology for this demonstration of economic hardship. (It is important to note that this does not mean that Montana WWTPs would be expected to implement RO to meet practical Limits of Technology [LOT] or nutrient criteria in practice.).

- Where available, plant level data is used for current costs, financial information, and effluent flow. For situations where information is limited, representative data is used from the U.S. Census of Manufacturing and other sources to estimate a range of financial information for the particular industry group in Montana within which the affected business belongs.
- Discount (Interest) Rate—In some cases, assuming a five percent interest rate, as in the EPA Guidance, may be an appropriate discount rate to annualize the capital costs of treating nutrients, but may not be appropriate for private sector capital markets which could be higher. Additionally, there exists some uncertainty on the rate depending on the general economic conditions at the time the investment is required and the debt capacity and rating of the borrower. Costs estimates are developed for sensitivity analysis scenarios with both a five percent and seven percent discount (interest) rate.
- Labor Costs—For the sensitivity scenarios developed below, labor costs of 15 and 48 percent of capital costs were included in the total cost estimates. The original draft WERF study cost estimates used for this demonstration did not include labor costs, which can be a significant proportion of a treatment process. This is the additional labor that would be needed to operate the new unit processes that would be installed, so it would be added on to O&M costs (yet is based on capital costs). An analysis of the life-cycle costs for a number of technologies used to control nitrogen and phosphorus in wastewater treatment plants estimated that labor costs are between 15-21 percent of the annualized capital costs for nitrogen and 15-48 percent of annualized capital costs for phosphorus (Kang and Omstead, 2011).[2]
- Costs are under-estimated for small facilities and those with low flows, because the WERF cost data was multiplied by effluent flow providing a linear cost estimate based on flow. Clearly, there will be a minimum cost of treating to base nutrient standards for facilities with small flows such as pouring concrete, hiring labor, etc. that is greater than the linear cost estimates for these low-flow and small facilities. DEQ believes that small facilities could not afford RO or even mechanical treatment in many cases.

Current effluent nutrient levels and estimates of current treatment costs at the 51 businesses were compared to costs that would be needed to meet base numeric nutrient standards based on the WERF study. In this way, annual capital and operations costs needed for meeting base nutrient criteria (above current nutrient treatment costs) were applied to each business. In other words, existing water nutrient treatment costs for private businesses were subtracted from estimated costs to meet the base criteria, if some treatment of nutrients was already being done. If a business already met WERF level 2 nutrient levels, for example, then the level 2 costs for both capital expenditures and operations were subtracted from 100% RO costs (stricter than level 5) to arrive at the additional cost to meet the criteria.

A cost sensitivity analysis is conducted to account for differences in assumptions. As mentioned above, the operations costs of meeting base numeric criteria taken from the WERF study (**Table 3**) do not include labor and maintenance costs, so the costs estimates may be slightly low (conservative). This is addressed below in the cost sensitivity analysis. Discount rates may vary for borrowing money to meet the criteria. Also, WERF level 5 is not quite as stringent as the Montana base nutrient criteria for most stringent TN criterion (0.3 mg TN/L), so the costs to reach nutrient standards estimated for this demonstration are potentially underestimated in that sense as well. This is also addressed below in the cost sensitivity analysis.

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[2] Based on information in: Introduction of Nutrient Removal technologies Manual, EPA, 2008 and WEF/WERF Cooperative Study of Nutrient Removal Plants: Achievable Technology Performance Statistics for Low Effluent Limits)

Multiple estimated expected treatment costs were calculated based on six cost sensitivity scenarios (see **Table 4**). To reach these six scenarios, the discount (interest) rate was varied at 5 or 7 percent and the addition of both high (48 percent) and low (15 percent) labor costs as a percentage of capital costs were considered across each scenario.[3] Then, the 100% RO was added on to the original cost estimates separately to isolate how that assumption alone would affect costs.

Table 4. Scenarios for Sensitivity Analysis

Scenario	Description	Discount Rate	Labor Cost
Original	5% discount rate and 0% labor cost	5%	0%
Scenario A	Change of labor cost to 15% of capital cost	5%	15%
Scenario B	Change of labor cost to 48% of capital cost	5%	48%
Scenario C	Discount rate increase from 5% to 7% and 0% labor cost	7%	0%
Scenario D	Discount rate increase from 5% to 7% AND change of labor cost to 15% of capital cost	7%	15%
Scenario E	Discount rate increase from 5% to 7% AND change of labor cost to 48% of capital cost	7%	48%

COST RESULTS

Table 5 presents the estimated annual costs (annualized capital costs plus annual operation and maintenance costs) resulting from the installation of the additional water treatment controls to meet numeric nutrient criteria for each of the scenarios analyzed. Note that permittees with 'NA' as a cost estimate indicates those facilities without enough information to make a determination (i.e., no flow data available). **Table 5** shows the estimated average annual cost across all six scenarios for each affected business. It also shows which general business sector the individual business was categorized in (in bold italic type).

Table 5. Estimated Average Annual Costs (Capital and O&M Cost) for Affected Montana Businesses (***Sector Codes: M-metal mining, C-Coal Mining, OG-Oil and Gas, E-Electric Generation, R-Refineries, Mfg-General Manufacturing, Oth-Other***)

Company (Sector code)	Original	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
Cenex Refinery (***R***)	$4,644,227	$5,141,402	$6,235,188	$5,228,721	$5,813,570	$7,100,239
Conoco Refinery (***R***)	$5,682,449	$6,290,768	$7,629,070	$6,397,607	$7,113,200	$8,687,504
Burlington-Northern Railroad Whitefish Facility (***Oth***)	$152,129	$168,190	$203,525	$171,011	$189,904	$231,471
John R. Daily meat packing (***Oth***)	$475,402	$525,593	$636,015	$534,409	$593,451	$723,345
Montana Resources Inc. mine (Copper) (***M***)	$7,181,262	$7,969,886	$9,704,860	$8,108,392	$9,036,086	$11,077,012
Montana Sulfur and Chemical (***Mfg***)	$5,209,641	$5,759,662	$6,969,708	$5,856,262	$6,503,276	$7,926,708
Sidney Sugars Inc. (***Mfg***)	$2,777,137	$3,074,436	$3,728,493	$3,126,650	$3,476,376	$4,245,773
Western Sugar Cooperative (***Mfg***)	$19,995,386	$22,135,937	$26,845,150	$22,511,882	$25,029,908	$30,569,564

[3] DEQ first annualizes the capital cost and then multiplies it by the 15 or 48 percent factor.

Table 5. Estimated Average Annual Costs (Capital and O&M Cost) for Affected Montana Businesses
(***Sector Codes: M-metal mining, C-Coal Mining, OG-Oil and Gas, E-Electric Generation, R-Refineries, Mfg-General Manufacturing, Oth-Other***)

Company (Sector code)	Original	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
Montana Dakotas Utility-Lewis and Clark Electric generation plant (***E***)	$67,237,631	$74,336,416	$89,953,743	$75,583,176	$83,933,792	$102,305,148
Montana Rail Link—Livingston Rail Yard (***Oth***)	$225,911	$249,762	$302,234	$253,951	$282,008	$343,734
Corette electrical generation plant-PPL Montana (***E***)	$207,592,027	$229,509,086	$277,726,616	$233,358,378	$259,140,390	$315,860,816
Ash Grove Cement Company (***Mfg***)	$59,787	$66,099	$79,985	$67,207	$74,632	$90,968
Exxon-Mobile Refinery (***R***)	$5,767,900	$6,385,366	$7,743,793	$6,493,812	$7,220,166	$8,818,143
Trident Cement Plant (***Mfg***)	$13,154	$14,506	$17,480	$14,743	$16,334	$19,832
Big Sky Coal Company (***C***)	$4,825,326	$5,334,772	$6,455,554	$5,424,246	$6,023,530	$7,341,956
Decker Coal mine (west mine) (***C***)	$1,595,836	$1,771,086	$2,156,635	$1,801,865	$2,008,019	$2,461,558
Yellowstone Boys and Girls Ranch (***Oth***)	$42,725	$47,299	$57,361	$48,102	$53,483	$65,320
Absaloka Coal Mine (***C***)	$2,281,928	$2,522,848	$3,052,873	$2,565,161	$2,848,566	$3,472,058
MT Behavioral Health Inc WWTP (***Oth***)	$213,626	$236,495	$286,807	$240,512	$267,414	$326,598
Elkhorn Health Care WWTP (***Oth***)	$32,044	$35,474	$43,021	$36,077	$40,112	$48,990
Savage Coal Mine (***C***)	$912,771	$1,009,139	$1,221,149	$1,026,064	$1,139,426	$1,388,823
Boulder Hot Springs WWTP (***Oth***)	$134,697	$148,918	$180,204	$151,416	$168,145	$204,948
Rosebud Coal Mine (***C***)	NA	NA	NA	NA	NA	NA
Decker Coal mine (east mine) (***C***)	$1,268,120	$1,407,381	$1,713,755	$1,431,839	$1,595,658	$1,956,060
Spring Creek Coal Mine (***C***)	$31,693	$35,040	$42,401	$35,627	$39,563	$48,223
Stillwater Mining Company-1 (***M***)	$1,341,870	$1,489,230	$1,813,422	$1,515,111	$1,688,457	$2,069,819
Stillwater Mining Company-2 (***M***)	$1,026,867	$1,135,282	$1,373,793	$1,154,322	$1,281,855	$1,562,426
Beaverhead Talc Mine (***Mfg***)	$221,487	$245,198	$297,362	$249,362	$277,254	$338,617
Exxon Mobile Refinery (***R***)	$12,526,340	$13,867,313	$16,817,454	$14,102,828	$15,680,274	$19,150,656
Montana Tunnels Mining (***M***)	NA	NA	NA	NA	NA	NA
Luzenac-Yellowstone talc mine (***Mfg***)	NA	NA	NA	NA	NA	NA
Bull Mountain Coal Mine (***C***)	$570,482	$630,712	$763,218	$641,290	$712,142	$868,014
Barretts Mineral (***Mfg***)	$2,498,711	$2,762,519	$3,342,896	$2,808,851	$3,119,180	$3,801,903
Montana Aviation Research (***Oth***)	$79,234	$87,599	$106,003	$89,068	$98,909	$120,558
M & W Milling & Refining (***Mfg***)	$46,143	$51,083	$61,950	$51,950	$57,761	$70,545

Table 5. Estimated Average Annual Costs (Capital and O&M Cost) for Affected Montana Businesses
(***Sector Codes: M-metal mining, C-Coal Mining, OG-Oil and Gas, E-Electric Generation, R-Refineries, Mfg-General Manufacturing, Oth-Other***)

Company (Sector code)	Original	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
Asarco/Mike Horse mine water treatment (***M***)	$99,977	$110,680	$134,226	$112,559	$125,150	$152,848
Columbia Falls Aluminum Co (***Mfg***)	$952,237	$1,052,772	$1,273,949	$1,070,429	$1,188,693	$1,448,873
Asarco Inc. (***Mfg***)	$217,091	$240,010	$290,434	$244,036	$270,998	$330,313
YELP electric generation (***E***)	$395,217	$436,943	$528,741	$444,272	$493,356	$601,341
Montanore Mine (***M***)	$11,475	$12,714	$15,441	$12,932	$14,390	$17,597
REC Advanced Silicon (***Mfg***)	$1,825,542	$2,018,278	$2,442,298	$2,052,129	$2,278,853	$2,777,646
M&K Oil Co-waste disposal (***OG***)	$26,622	$29,433	$35,617	$29,927	$33,233	$40,507
Sleeping Buffalo Hot Springs (***Oth***)	$27,558	$30,508	$36,998	$31,026	$34,496	$42,131
Pinnacle Gas Resources (***OG***)	NA	NA	NA	NA	NA	NA
Barretts-Regal Talc Mine (***Mfg***)	$228,193	$252,285	$305,287	$256,516	$284,857	$347,206
Fidelity Oil and Gas (***OG***)	$3,476,643	$3,858,437	$4,698,384	$3,925,492	$4,374,613	$5,362,681
Headwaters Livestock Auction (***Oth***)	CAFO					
Wolf Mountain Coal (***C***)	$10,254	$11,352	$13,767	$11,545	$12,836	$15,677
Cattle Development Center (***Oth***)	CAFO					
James Guercio-OW ranch (***OG***)	$270,722	$300,452	$365,858	$305,674	$340,646	$417,586
IOFINA Natural Gas Water Treatment Facility (***OG***)	NA	NA	NA	NA	NA	NA
Total	**$364,205,474**	**$402,798,363**	**$487,702,718**	**$409,576,430**	**$454,974,963**	**$554,851,734**

SIGNIFICANT IMPACT ANALYSIS[4]

IMPACTS ON BUSINESSES AND MONTANA AS A WHOLE

Because of the technical challenges and high costs of meeting the nutrient standards with today's technologies (especially TN), Montana believes that some firms in the state having to meet the base numeric nutrient standards would have to shut down or cut back their operations, or have to determine if affordable 'non-discharge' options are available. Non-discharge options include, for example, a. land application, b. total/seasonal retention, c. piping water long distances away from state waters, and d. trading. These non-discharge options, including land application, could be expensive or might not be feasible in certain areas (such as places far from open land or with few trade partners) or during the cold

[4] Note: Much of the data in this section came from the U.S. Census Bureau, 2007 Economic Census, 2007 Economic Census of Island Areas, and 2007 Nonemployer Statistics (U.S. Census Bureau, 2007). These files are released on a flow basis from March 2009 through mid-2011. The national data are subject to change; they will be replaced when updated data are added from the Geographic Area Series and Summary Series.

times of the year. Some of the 51 affected businesses might simply not have a non-discharge option available, and some non-discharge options may be as expensive as discharge options.

Montana believes that there will also be an adverse effect from having to meet base nutrient criteria on new businesses starting up in Montana. Small businesses, with generally thinner margins, will especially be affected because they will most likely not be able to afford RO or non-discharge options.

In the Billings areas, companies discharging into the Yellowstone River would likely have to meet, end of pipe, criteria comparable to the 'wadeable streams' standards of Western Montana (i.e., about 0.03 mg TP/L and 0.3 mg TN/L). Treating to criteria at the end of pipe would be extremely costly to businesses in Billings, including refineries. These businesses might have to shut down, or might choose to relocate due to high treatment costs. Montana's refineries provide almost all of Montana's liquid petroleum products as well as about 50% of Spokane's and 30% of North Dakota's (Montana Department of Environmental Quality, 2012). Shutting down two or all three refineries in the Billings areas would be very damaging to Montana in terms of petroleum products supply shortages. In addition, the Stillwater mine is one of the only sources of palladium and platinum in North America, and a shutdown would choke off that supply. Clearly, these four businesses are crucial to the larger overall economy in the state, and their shutting down (or even scaling back) would have significant and widespread effects within and outside of Montana.

One way to look at the 'significant impact' on businesses is to see what the impacts would be on the largest affected businesses in Montana. If the largest businesses are significantly impacted, then it is very likely that smaller businesses will also be impacted significantly due to the 'economies of scale' advantage of larger businesses and their deeper pockets of available financial resources.

Some businesses that would not have to shut down, as a result of having to meet base standards, may have to scale back production, and/or lay off workers. An example of this is Fidelity Exploration & Production Company who is currently engaged in developing and extracting coal bed methane natural gas from subsurface formations in the Powder River Basin. As indicated in a recent DEQ economic analysis, as a result of the new water quality sodium standards under TBELs (Technology-Based Effluent Limits), Fidelity would have to cut back temporarily or permanently some current and future natural gas production resulting in lower revenues/profits, less jobs being created, and fewer tax and royalty payments to the State of Montana. These estimated effects from a cutback to Fidelity include production taxes paid to Montana down by almost $13 million from 2011-2015, federal royalties down by almost $9 million (half of which goes to Montana), state royalties down by just over $1 million, and fee royalties down by almost $16 million to private land owners. Also, up to 735 million cubic feet (MCF) of lost production of natural gas would occur due to shutting in some wells (Montana Department Environmental Quality, 2010). Another estimated $5.4 million in additional annual costs for Fidelity (as estimated in **Table 1**) to meet nutrient criteria on top of water quality sodium standards would result in even further cutbacks.

The main affected business sectors in Montana, as defined on page 4, would be significantly affected. The oil and gas industry as a whole in Montana has revenues of about $1.3 billion annually (U.S. Census Bureau, 2007). The annual cost of meeting nutrient criteria would be about $4 to $6 million annually or about 0.3% to 0.5% of annual revenue. The percentage of actual profits (revenues minus costs) would be higher, as it would be for all major sectors.

Metals mines in Montana as a whole would have to pay an estimated $9.7 to $14.9 million annually in estimated costs to meet base numeric criteria which is 0.9% to 1.3% of total annual revenue in Montana for that sector (about $1.11 billion) (U.S. Census Bureau, 2007). In addition, metal mines prices and thus revenues fluctuate a lot and present a further challenge to bearing additional costs.

Coal mines in Montana as a whole would have to pay $11.5 to $17.5 million annually in estimated costs to meet base numeric criteria which is 2.4% to 3.7% of total annual revenue for that sector (U.S. Census Bureau, 2007). In addition, the mine-mouth price of coal in Montana (in 2007 dollars) has been cut in half since the early 1980s and shows no signs of rising in the coming years (Montana Department of Environmental Quality, 2012).

DEQ estimates that each of the three large refineries in Montana would require annual investments of between $4.6 and $19.2 million per year to comply with the nutrient criteria (see **Table 5** above), and about $28.6-$43.8 million for all three. Based on information from the U.S. Census Bureau, the annual revenue for that sector is $5.45 billion. This indicates that the annual costs to meet nutrient criteria are between 0.5% and 0.8% of total revenue for the refinery sector. In addition, refineries will be hit with substantial new air quality regulations from EPA in 2012 (e.g. mercury standards) that will cost additional money causing a cumulative economic impact.

An additional alternate analysis was performed for refineries in the Billings area. These refineries as a whole had an annual input of 60 million barrels of crude from 2004-2007 (Montana Department of Environmental Quality, 2012b). Based on the financial reports for one of the major oil companies in the US, earnings (which is revenues minus costs) from US-based refining for five fiscal quarters (the fourth quarter of 2009 and all four quarters of 2010) have fluctuated between ($1.80) and $2.68 per barrel (Exxon Mobil Corporation, 2011). This provides estimated earnings for each of the Billings-area refineries between ($36) million and $53.6 million per year (assuming about 20 million barrels of crude input to each annually), making the annual investments of between $4.6 and $19.2 a significant portion of their earnings or an exacerbation of their losses (between 9% and 36% of earnings in the best-case estimated scenario of $53.6 million in earnings per refinery, and a much greater share for any earnings less than that). In some fiscal quarters, refineries appear to be losing money, making such costs harder to bear.

The electric power generation, transmission and distribution sector in Montana in 2007 employed 2,348 total workers (U.S. Census Bureau, 2007). Annual costs for the three affected generating plants would be an estimated $418.7 million (see **Table 1** above). The total electricity retail sales for Montana in 2010 were $921 million (U.S. Energy Information Administration, 2011). Thus, estimated annual costs of complying with nutrient criteria would be 29.8% to 45.5% of total annual revenues for this sector. (Note: For electric generation, DEQ took the overall sector labeled “Electric power generation, transmission and distribution” (NAICS 2211) and subtracted the information for “Electric Power Distribution” (NAICS subsector 221122) to isolate ‘Electric generation” as much as possible. The data in this file come from separate 2007 Economic Census Industry Series, Geographic Area Series, and Summary Series data files, as well as data files from the 2007 Economic Census of Island Areas and the 2007 Non-employer Statistics (U.S. Census Bureau, 2007).

Further, if electricity costs were all passed on to consumers (we actually assumed otherwise for this analysis), this would translate to more than $30 each month in an electric bill increase for every resident in Montana. The Corette generation plant of 153 Megawatts (MW) capacity has an estimated cost of up to $316 million per year to meet criteria (due to a large effluent flow) and is a baseload generation plant for PPL Montana for its electricity customers. It would likely close with such high costs, causing PPL to lose a significant portion of its electricity supply portfolio, and causing electricity customers in Montana to have to get a portion of their electricity supply elsewhere.

The Sugar and confectionery product manufacturing sector in Montana in 2007 had payroll costs of $13.3 million and undetermined revenues (U.S. Census Bureau, 2007). Annual costs to meet base numeric criteria are estimated $34.8 million annually or almost three times the payroll costs. Sidney Sugars in Sidney, one of the two sugar plants affected, is a major employer in Richland County providing full time employment for approximately 150 people and part time employment for 280 more with an annual payroll of approximately $5.7 million. Another $50 million is paid out to local farm families for sugar beets grown on 47,500 acres of irrigated land. The refinery also ships 200,000 tons of sugar, pulp and molasses by rail and 75,000 tons by truck (Sidney Area Chamber of Commerce and Agriculture, 2012). Sugar refineries often break even in their operation and accept a sugar price set by a government-set ‘price floor’, leaving a very slim operating margin for nutrient treatment costs that would be over $30 million annually.

For manufacturing in Montana overall, the nutrient criteria would affect a small number of businesses in Montana out of the total number that are in that sector. The estimated annual costs to those affected businesses would be $35 to $54 million, versus $10.6 billion in Montana for all of manufacturing in Montana.

Smaller businesses such as small manufacturers and family-run businesses would almost certainly not be able to afford advanced biological treatment, much less RO. The larger businesses with small costs due to small effluent flows, such as the coal mines, would almost certainly look for alternate ways of disposing their water rather than accept such large costs. The impact to those businesses would depend upon the costs of something like land application.

It is important to note in considering the numbers above that the total revenues listed cover the entire sector in Montana, whereas only some businesses in these sectors are affected. For example, only a small portion of the entire “General Manufacturing” sector would be affected by the nutrient criteria, but DEQ only found revenue numbers for the entire sector. Thus, affected businesses in that category would be impacted more than the sector numbers indicate.

Adding up the sectors, total cost would be $364 to $555 million annually out of a total revenue of $19.9 billion or about 1.8% to 2.8% of total revenue. While some sectors’ annual costs would be less than 1% of total revenue, all of the costs of these six main sectors together would probably be greater than 2% of total annual revenues each year. Thus, these costs are likely to be significant to Montana’s affect businesses sectors and thus to Montana business overall.

CASE STUDIES FOR INDIVIDUAL BUSINESSES

Stillwater Mining Company[5]

The Stillwater Mining Company (SMC) operates two underground mines and processing facilities in south-central Montana and is one of the largest private employers in Montana (over 1000 employees). SMC is the only primary producer of palladium and platinum in the United States with the majority of

[5] (Gilbert, Bruce, personal communication 2011)

the metal production from the mines utilized in clean air technologies and catalytic converters for the auto industry. SMC's multiple stage water management and water treatment facilities are engineered for treatment of nitrogen species that occur in mine waters due to the use of blasting agents in underground mining operations (**Table 6**). Ammonium nitrate (the same compound used in agricultural fertilizer) is the primary component of the explosives used for mining.

The following is a brief outline of SMC's water treatment/management system components:

a. Primary Treatment: Clarification (removal of suspended solids)
b. Secondary Treatment: Biological denitrification (fixed bed and moving bed bioreactors)
c. Enhanced Secondary Treatment: Biological nitrification + denitrification (moving bed bioreactors)
d. Tertiary water management/treatment: recycle/reuse (mine support) and recycle/reuse for land application (Agronomic uptake - Stillwater Mine Hertzler facility)
e. Backup treatment system: Reverse osmosis (low volume unit to be used in short-term situations where primary and secondary treatment sustain an unplanned upset)

SMC's most recent study on nitrogen treatment technologies was conducted in 2004 in order to identify the Best Available Technology for treatment of ammonia. The study looked at biological treatment, reverse osmosis, ion exchange, breakpoint chlorination, and ammonia stripping. The study concluded that enhanced biological nutrient removal was the best available technology for the water management systems at SMC and that the treatment efficiency was equal to or better than the other technologies. Additionally, the study found that the treatment technology (true nutrient removal, not just separation/filtration of nitrogen compounds) was superior to the other treatment technologies due to the lack of waste stream and lower energy consumption and operating cost.

Biological treatment of nutrients does, however, come with limitations and challenges. Consistency in treatment efficiency is one of the primary challenges. The nitrifying and denitrifying bacteria are sensitive to changes in water temperature and chemistry as slight changes in temp or pH can cause sharp fluctuations in bacteria vitality and treatment efficiency. Likewise a sudden increase in flows or sustained increases over time can cause reduced contact and retention time resulting in decreased efficiencies.

Biological treatment of nitrogen (primary, secondary, and enhanced secondary) at the SMC facilities results in Total Nitrogen (TN) effluent concentrations that average 4-5 mg/L TN during the past 3 years (**Table 6**). During that same time frame, the range of TN effluent concentrations at the SMC treatment systems is 1 mg/L to 15 mg/ l (does not include upset conditions). Consistency of treatment efficiency is easier to maintain during the summer time when water temperatures are warmer and water chemistry is more consistent. (It should be noted that base numeric nutrient standards will only apply in summer in most cases.) During the summer, the SMC nutrient treatment systems are able to consistently achieve 5 mg/L, however, during the winter months (6 months of the year), colder temperatures and higher TDS in the mine waters can trigger periods of variability in treatment efficiency that can result in effluent concentrations of up to 15 mg/L. Because of this variability, it is difficult to numerically quantify the limits of technology (with less than 5 mg/L accuracy) for enhanced biological nutrient treatment such as SMC experiences in the mountainous headwaters areas across Montana.

Table 6. General Summary of Treatment Efficiency for total nitrogen (TN) as well as representative effluent TN concentrations

Nitrogen Removal Performance	Stillwater Mine	East Boulder Mine
Effluent TN* (99% confidence interval**)	10 mg/L	10 mg/L
Effluent TN (5-yr avg)	6 mg/L	10 mg/L
Effluent TN (3-yr avg) (w/EBNR***)	5 mg/L	4 mg/L
Effluent TN (3-yr effluent range)	1 mg/L – 15 mg/L	1 mg/L – 15 mg/L
Effluent TN (3-yr avg)	32 mg/L	39 mg/L
Nitrogen Removal Efficiency (3-yr)	85%	90%

*TN = Total Nitrogen (Nitrate + Nitrite + Ammonia + Organic N)
** 99% confidence interval means that (on average) the effluent is less than or equal to 10 mg/L 99% of the time, or the effluent exceeds 10 mg/L approx. once every 100 days
*** Enhances Biological Nutrient Reduction is a systems upgrade that includes mixed-bed bioreactors for nitrification (ammonia reduction) and denitrification (nitrate reduction)

Table 7 below is a summary of capital expenditures for water treatment systems at each of the mine sites. The capital expenditures represent the time period of 1995 to 2011.

Table 7. Capital expenditures for water treatment systems at each of the mine sites

Water Treatment	Stillwater Mine	East Boulder Mine	Total
Capital Cost (1995-2011)	$7,500,000	$3,800,000	$11,300,000

In addition to capital expenditures, operating and maintenance costs for the SMC water treatment systems can range between $350K and $500K per year per site depending on flow rates, maintenance requirements (including labor), and mechanical replacements. Additionally, it should be noted that treatment capacity is more sensitive to flow than concentration which adds potential to inflate both capital and operating costs dramatically even if overall influent concentrations are relatively low. Mine size, hydraulic setting, changing hydraulic conditions, production rate and commodity pricing (to name a few) can impact significantly on capital requirements to sustain and grow the company and meet changing regulatory mandates. Complicating the picture further is the fact that current operational costs and future cost projections are influenced by more site-specific parameters (flow, temperature, ph, TDS, contact time, bacterial regime etc.) that are ever-changing. In order to meet these operating challenges and maintain operational flexibility, biological treatment design normally requires process redundancies and additional capacity to compensate for upset conditions and assure a reasonable availability in order to meet treatment design criteria. These factors all impact upon the ability of new and existing mines to meet the new, low surface water standards and add an additional complexity to the economic decision-making process inherent to mine development. Likewise, the variability and cyclic nature of commodity prices can significantly impact on a Company's ability to meet new or increased capital budget allocations associated with new regulatory standards.

The proposed removal targets (Montana's base nutrient criteria) would require nitrogen removal rates of over 99% which are at least an order of magnitude lower than can be achieve with the current Best Available Technology, according to SMC.

Annualizing the above costs (existing, current treatment costs) would come to $1.8 million ($1.06 million capital annualized plus $350,000-$500,000 annual operating costs at each site). This is in addition to an estimated $3.6 million annually to get to base nutrient criteria or about $5.4 million per year total in annual costs for nutrient treatment. Is $3.6 million in additional annual cost significant and widespread? Here are Stillwater's (Stillwater Mining Company, 2011) earnings before taxes:

- 2010 $50.4 million
- 2009 -$8.7 million
- 2008 -115.8 million

Palladium and platinum prices reached high levels in 2010 from very low levels in 2008. In the best year, the annual additional cost of nutrient treatment beyond current treatment is 7% of profits. In the worst years, the company does not make a profit. Stillwater is experiencing great uncertainty in commodity prices and would probably not invest a lot of additional money for treatment beyond what it has already done. Palladium and Platinum prices as of December, 2011 are down about 20-30% from 2010 levels (KITCO, 2012).

Smurfit Stone

When they were still in operation in 2009, Smurfit-Stone stated that they could not afford advanced mechanical or biological treatment. They estimated that advanced mechanical treatment would have cost on the order of $53 million in capital costs, and they stated that the mill would have closed faced with this level of treatment costs (Caprara, 2009). The mill closed down anyway in 2010 (due to economic influences unrelated to nutrient water-quality criteria), so this is only a cost example.

Refineries[6]

Using Best Available Demonstrated Technology

TN: The current average release of TN in the water effluent of Billings area refineries is >5 TN mg/L. The maximum is 12-55 TN mg/L. From primary treatment to discharge the steps for best available technology (BAT) are: primary treatment to aeration tank to anoxic denitrification to final aerobic treatment to clarifier to filter to discharge. A supplemental carbon source is added between the aeration tank and the anoxic denitrification. For a 60,000 barrels per day (BPD) refinery already nitrifying, the approximate capital cost of adding the anoxic denitrification, final aerobic treatment, and a filter is $5 million.

TP: The typical average effluent concentration of TP in the Billings area refineries is 0.08 mg/L to 0.14 mg/L; 95th percentile effluent total phosphorus = 0.2 mg/L to 0.7 mg/L. From Biological WWTP to discharge the steps for BAT are: add alum, ferric chloride or lime to chemical precipitation (clarifiers) to discharge. Sludge is removed from the chemical precipitation step for dewatering and disposal. For 60,000 bpd refinery, approximate capital cost is $6 million, and sludge generation is approximately 80 tons/year.

According to Dr. Matt Gerhardt of Brown and Caldwell (engineering consultants), limits of treatment technology for nitrogen at the refineries is:

- ~3 mg/L as N average
- >10 mg/L as N maximum

The limits of technology for phosphorus removal are:

- 0.07 mg/L as P average

[6] (Gerhardt, 2009)

- 0.2 – 0.7 mg/L as P maximum

These costs are lower than the estimated costs to meet Montana's base nutrient criteria because this technology would only meet BAT (~WERF Level 3) and not base numeric criteria.

WIDESPREAD ANALYSIS

The final step in the S&W demonstration for Montana businesses is the widespread test. USEPA guidance (U.S. Environmental Protection Agency, 1995) allows flexibility to go beyond direct ratios or specific tests for a Widespread finding, and suggests a list of "essay style" questions to address. From the EPA guidance:

> "The financial impacts of undertaking pollution controls could potentially cause far-reaching and serious socioeconomic impacts. If the financial tests outlined in Chapter 2 and 3 suggest that a discharger (public or private) or group of dischargers will have difficulty paying for pollution controls, then an additional analysis must be performed to demonstrate that there will be widespread adverse impacts on the community or surrounding area. There are no economic ratios per se that evaluate socioeconomic impacts. Instead, the relative magnitudes of indicators such as increases in unemployment, losses to the local economy, changes in household income, decreases in tax revenues, indirect effects on other businesses, and increases in sewer fees for remaining private entities should be taken into account when deciding whether impacts could be considered widespread. Since EPA does not have standardized tests and benchmarks with which to measure these impacts, the following guidance is provided as an example of the types of information that should be considered when reviewing impacts on the surrounding community." (U.S. Environmental Protection, 2012).

DEQ considered the widespread analysis based on the following basic question: For Montana, what are the economic and social ripple effects of the substantial impacts to businesses on the local area where the business is located and on the state as a whole? Other questions considered included the following: If some small and medium sized businesses shut down, what is the impact? What is the impact of lower tax revenue to Montana in a time of lower revenue due to the Recession?

An important step in these questions was to define the geographic area where project costs pass through to the local economy. For Montana's widespread analysis, DEQ established the entire state as the "geographic area" considered in the widespread demonstration.

Another important aspect of Widespread impacts is to look at the effects of the current Recession on Montana's businesses.

MONTANA MANUFACTURING DEFINED

Manufacturing Defined

Montana's manufacturing, energy and mining sectors would be hit the hardest by having to meet nutrient criteria due to the number of businesses affected in those sectors. Compared with the U.S. state average, Montana has less manufacturing as a percentage of the whole economy according to the University of Montana Bureau of Business and Economic Research (BBER)(Bureau of Business and

Economic Research, 2012). Unlike manufacturing, however, mining clearly has a higher percentage of workers in Montana than the U.S. average. About 2% of employment in Montana is located in the Mining industry, while only about 0.6% of U.S. employment is in Mining (Montana Department of Labor & Industry, 2011). As mentioned earlier, Stillwater is the only primary source of palladium and platinum in North America, with approximately 1,300 employees.

Despite a lower than average share of the U.S. economy, the state's manufacturers as defined by the Montana Bureau of Business and Economic Research (BBER) employed 21,000 workers in 2010, producing more than $1 billion in labor income, and $10 billion in total sales (Morgan, et al., 2011). As of September 2011, the Natural Resource and Mining industry accounted for 8,200 jobs in Montana. Together, this was just over 5% of all 436,000 non-agricultural jobs in Montana (Montana Department of Labor & Industry, 2011). The U.S. Bureau of Economics states that in 2010, out of $36.1 billion in total Montana gross domestic product (GDP), that $1.8 billion was from Mining and Oil and Gas, and that $1.8 billion was from Manufacturing (U.S. Department of Commerce, 2012)(**Table 8**). Taken together, that is about 10% of Montana's GDP in 2010, although it is important to note that only a few of the businesses in those two sectors would have to meet nutrient criteria.

Table 8. Montana GDP--Bureau of Economics Regional Analysis (millions of current dollars) (U.S. Department of Commerce, 2012).

	2009	2010
Mining	1,593	1,838
Oil and gas extraction	304	(NA)
Montana Mining (except oil and gas)	997	(NA)
Montana Support activities for mining	291	(NA)
Nonmetallic mineral product manufacturing		62
Montana industry total	34,999	36,067

Recession concerns

Montana's industries, as in the rest of the U.S., are suffering from the recession. During the current recession, Montana's Construction, Manufacturing, and Trade, Transportation, and Utilities industries were the hardest hit sectors. Job losses in these industries had a larger impact in some regions than others. The Northwest region of Montana was highly concentrated in Manufacturing in 2007, and was impacted by the losses in the industry. Declines in manufacturing since 2001 were largest in Montana's wood and paper products industry with segments of Montana's metals, machinery, and nonmetallic minerals manufacturers also suffering declines (Montana Department of Labor & Industry, 2011).

More than 60 percent of responding firms to BBER's annual manufacturers survey (Jan 2010) indicated the recession has caused their firm to fundamentally change the way they plan to operate in the future. Most of the major changes involved reducing costs and operating more efficiently. Other major changes included diversification into new products and markets, or focusing on key products and projects. The survey results indicate the widespread impacts in 2009, with over 60 percent of responding Montana manufacturers reporting decreased production and sales. Sixty-five percent of surveyed Montana manufacturing firms reported decreased profits, with only 17 percent indicating profits equal to 2008. The proportion of respondents that reported curtailments of production increased to 49 percent, up from 37 percent in 2008. Seventeen percent permanently eliminated production capacity in 2009 versus 9 percent in 2008. The number of workers in 2009 relative to 2008 declined at 50 percent of the respondent facilities while 10 percent showed an employment increase (Bureau of Business and Economic Research, University of Montana, 2010).

During the recession, payroll employment in Montana declined 4.8%, leaving a large number of Montana workers unemployed. Manufacturing lost 4,040 jobs from 2007-2010, and natural resources and mining, 760 jobs. Job growth exiting the recession is expected to be slower than before the recession, with employment growth from 2010 to 2020 expected to average 0.9% annually compared to 1.2% per year from 2000 to 2007. At this pace, it will take at least four to five years to regain the jobs lost in recent years unless economic recovery picks up. Because of slow job growth, combined with the large number of existing unemployed workers plus the younger workers joining the labor force for the first time, the unemployment rate in Montana is expected to remain at higher levels for several years. Manufacturing would gain pre-Recession jobs lost not until after 2020 according to the Montana Department of Labor and Industry (Wagner, 2011).

The Production, Transportation and Material Moving, and Construction and Extraction occupational groups are also not expected to return to the 2007 employment peak before 2020. These occupational groups likely will continue to have excess labor throughout the next decade. In Montana, about 23,000 jobs requiring only on-the-job training or work experience were lost. It will take many years to re-employ these workers, even though about 3,000 new lower-skill jobs are expected to be added each year. In comparison, jobs requiring some type of post-high school education, almost none of which would be affected by having to meet nutrient criteria, did not show overall losses. The roughly 1,000 jobs in this category added annually will need to be filled by newly trained workers (Wagner, 2011).

In sum, Montana’s manufacturing, mining and energy production sectors are the areas most affected by nutrient standards and their associated costs. They are also among the areas that were hit hardest during the recession, and could have special challenges taking on significantly more costs.

WIDESPREAD CONCLUSIONS

- The current U.S. Recession is making the economics of businesses and their workers more challenging that during non-recession periods. The very high costs of meeting base numeric criteria today would deepen these challenges.
- Montana was 41st in the nation in per capita income as of 2009 at $22,881 (U.S. Census Bureau, 2012). Prices in Montana are about average for the U.S. across all goods, with housing slightly cheaper and certain types of goods (e.g. fresh foods) slightly more expensive due to geographical remoteness. Montanans on average do not have as much disposable income as the average American, and may have slightly higher living expenses due to long travel distances and higher heating bills. Losses in income from affect businesses could especially impact Montanans.
- As noted above, some affected businesses are located in or near small towns. Since most small towns do not have diverse economies, even a small decrease in business and in population can have a large effect on them. For example, some small Montana towns have less than 10 businesses total (e.g. Fromberg, MT).
- To the extent that gas and oil wells shut down due to meeting base numeric nutrient criteria, royalty payments to landowners (those who own their mineral rights) would decrease. Reduced royalty money as a result of reduced oil and gas production and mining production would lessen supplemental income to mostly rural landowners (including members of Reservations) who now collect that money. Royalty payment losses could include those to private landowners, the state of Montana and the Tribes.

- To meet the base numeric nutrient criteria will require hiring highly qualified wastewater engineers for each affected business. There could be widespread impacts associated with finding these qualified staff for facilities across the state and then paying them a competitive salary. Such operators may be hard to find for Montana businesses—especially those located in remote areas.
- Some businesses may not choose to locate in Montana if Montana were to require compliance with stringent criteria immediately, while other states do not. Eventually, all U.S. states will have to meet nutrient criteria, so this effect will probably decline over time.
- If electricity costs from meeting nutrient criteria were passed on to consumers, the average electricity bill per person in Montana would go up $30 per month (averaged over all Montana citizens). More likely, the affected generation plants (e.g. Corette) might simply close down.
- The 2010 census data showed that Montana's population is aging, with many on fixed incomes. This trend, coupled with any increased living expenses associated with meeting the base nutrient standards, could have negative impacts on a statewide scale.
- All states including Montana need every dollar of tax money collected during the current Recession and resulting time of state deficits. Any reduction in tax money collected from decreased business activity would hurt state coffers. Montana's budget is only one or two or three states to remain positive (budget surplus) due in part to natural resource taxes.
- DEQ's substantial and widespread analysis is based on the assumption that reverse osmosis or some ion exchange treatment technology would be required. Either technology is both economically and environmentally costly. Reverse osmosis generates brine that must be disposed of properly and results in significantly higher greenhouse gas emissions, electricity and chemical usage. Aggregated at the statewide scale, both the economic and environmental implications of meeting Montana's criteria would have widespread impacts for the State of Montana, including finding faraway places to dispose of the RO brine.
- Closure of two or three of the three major refineries in the Billings area could result in petroleum product shortages for Montana and nearby states.
- Most of the businesses affected pay higher wages than the Montana average. Any loss in these jobs would thus have a greater effect than the loss of lower paying jobs.

CONCLUSION

It is DEQ's best professional judgment that the resulting costs of complying with the base numeric nutrient criteria today would result in substantial costs beyond what individual firms can internalize. This would result in some businesses closing and a scaling down in economic activity in particular economic sectors of Montana. Energy production (electricity and fossil fuel), metals mining and certain manufacturing businesses would be hit the hardest. At this point in time, using reverse osmosis on 100% of effluent flow is simply too expensive for businesses to operate, and comes with a host of technical problems given Montana's winters and the business operations of affected companies (such as highly variable water flows at certain mines and greatly fluctuating annual revenues). The cumulative impact on these individual firms will create a widespread economic negative effect that would exacerbate Montana's current economic situation within the general U.S. recession. Aside from widespread impacts such as potential businesses and jobs lost, electricity prices and supply along with refined petroleum products could be greatly impacted if certain Billings plants had to scale back or close down.

Adding up the affected industry sectors in Montana, total cost to meet base numeric nutrient criteria would be $364 to $555 million annually out of a total revenue in those sectors of $19.9 billion, or about 1.8% to 2.8% of total revenue. While some sectors annual costs to meet nutrient criteria would be less than 1% of total revenue, all of the costs of these six main sectors together would probably be greater than 2% of total annual revenues each year. Thus, these costs are likely significant to Montana's affected businesses sectors and thus to Montana business overall. Impacts on Montana's tax revenues, rural citizens, energy usage in water treatment, royalty payments, and business climate would probably be widespread as a result of existing and new businesses having to meet the stringent base nutrient criteria.

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APPENDIX A - COST WORKSHEETS

Table A-1. Base Criteria Calculations Using 5% Interest Rate

NPDES ID	Facility Name	Flow (mgd)	Current Level of Nutrient Treatment	Required Level of Nutrient Treatment under the Criteria	Capital Cost per MGD ($million /MGD)	Facility Upgrade Capital Costs ($million)	Annualized Capital Costs (Assumed 20 years, 5% rate; $million/year)	Annualized Capital Costs (Assumed 20 years, 5% rate; $/year)	Operations ($1/ MG/day Treated)	Operations Costs ($/ year/ 1 MGD)	Facility Upgrade Operations Costs (annual) based on Facility MGD	Membrane Replacement Cost ($24,000 /yr/1 MGD)*Actual Flow	Total Operations costs including membrane replacement	Total Operations costs including membrane replacement + Labor Low (15%)	Total Operations costs including membrane replacement + Labor Hi (48%)
MT0000264	CENEX HARVEST STATES COOP.	***2.174***	***Level 1***	**100% RO**	19	$41.31	$3.31	$3,314,500	1610	587,650	1,277,551	52,176	1,329,727	1,826,902	2,920,687
MT0000256	CONOCOPHILLIPS - BILLINGS REFINERY	***2.66***	***Level 1***	**100% RO**	19	$50.54	$4.06	$4,055,460	1610	587,650	1,563,149	63,840	1,626,989	2,235,308	3,573,610
MT0000191	MONTANA RESOURCES	***5.04***	***Level 4***	**100% RO**	13	$65.52	$5.26	$5,257,494	980	357,700	1,802,808	120,960	1,923,768	2,712,392	4,447,365
MT0000248	SIDNEY SUGARS INCORPORATED	***1.3***	***Level 1***	**100% RO**	19	$24.70	$1.98	$1,981,992	1610	587,650	763,945	31,200	795,145	1,092,444	1,746,501
MT0000281	WESTERN SUGAR COOPERATIVE	***9.36***	***Level 1***	**100% RO**	19	$177.84	$14.27	$14,270,342	1610	587,650	5,500,404	224,640	5,725,044	7,865,595	12,574,808
MT0000477	EXXONMOBIL REFINING & SUPPLY	***2.7***	***Level 1***	**100% RO**	19	$51.30	$4.12	$4,116,445	1610	587,650	1,586,655	64,800	1,651,455	2,268,922	3,627,348
MT0000485	TRIDENT PLANT	***0.0072***	***Level 2***	**100% RO**	15.6	$0.11	$0.01	$9,013	1510	551,150	3,968	173	4,141	5,493	8,467
MT0000892	DECKER COAL CO (WEST MINE)	***1.12***	***Level 4***	**100% RO**	13	$14.56	$1.17	$1,168,332	980	357,700	400,624	26,880	427,504	602,754	988,303
MT0020460	YELLOWSTONE BOYS & GIRLS RANCH	***0.02***	***Level 1***	**100% RO**	19	$0.38	$0.03	$30,492	1610	587,650	11,753	480	12,233	16,807	26,869
MT0021431	MT BEHAVIORAL HEALTH INC WWTP	***0.1***	***Level 1***	**100% RO**	19	$1.90	$0.15	$152,461	1610	587,650	58,765	2,400	61,165	84,034	134,346
MT0023566	ELKHORN HEALTH CARE WWTP	***0.015***	***Level 1***	**100% RO**	19	$0.29	$0.02	$22,869	1610	587,650	8,815	360	9,175	12,605	20,152
MT0023639	BOULDER HOT SPRINGS WWTP	***0.085***	***Level 3***	**100% RO**	13.9	$1.18	$0.09	$94,807	1220	445,300	37,851	2,040	39,891	54,111	85,398
MT0024210	DECKER COAL CO (EAST MINE)	***0.89***	***Level 4***	**100% RO**	13	$11.57	$0.93	$928,407	980	357,700	318,353	21,360	339,713	478,974	785,348
MT0024716	STILLWATER MINING COMPANY	***0.94176***	***Level 4***	**100% RO**	13	$12.24	$0.98	$982,400	980	357,700	336,868	22,602	359,470	506,830	831,022
MT0026808	STILLWATER MINING COMPANY	***0.648***	***Level 3***	**100% RO**	13.9	$9.01	$0.72	$722,761	1220	445,300	288,554	15,552	304,106	412,521	651,032
MT0027821	BEAVERHEAD TALC MINE	***0.10368***	***Level 1***	**100% RO**	19	$1.97	$0.16	$158,071	1610	587,650	60,928	2,488	63,416	87,127	139,290
MT0028321	EXXON MOBIL BILLINGS REFINERY	***5.86368***	***Level 1***	**100% RO**	19	$111.41	$8.94	$8,939,820	1610	587,650	3,445,792	140,728	3,586,520	4,927,493	7,877,634
MT0028428	MONTANA TUNNELS MINING INC	***0***	***Level 5***	**100% RO**	6.5	$0.00	$0.00	$0	490	178,850	0	0	0	0	0
MT0028584	LUZENAC AMERICA INC - YELLOWSTONE MINE	***0***	***Level 3***	**100% RO**	13.9	$0.00	$0.00	$0	1220	445,300	0	0	0	0	0
MT0029891	BARRETTS MINERALS INC	***1.5768***	***Level 3***	**100% RO**	13.9	$21.92	$1.76	$1,758,719	1220	445,300	702,149	37,843	739,992	1,003,800	1,584,177
MT0030015	M & W MILLING & REFINING INC	***0.0216***	***Level 1***	**100% RO**	19	$0.41	$0.03	$32,932	1610	587,650	12,693	518	13,212	18,151	29,019
MT0030031	ASARCO LLC - MIKE HORSE/ANACONDA MINE WATER TREATMENT SYSTEM	***0.0468***	***Level 1***	**100% RO**	19	$0.89	$0.07	$71,352	1610	587,650	27,502	1,123	28,625	39,328	62,874
MT0030147	ASARCO INC	***0.136994***	***Level 3***	**100% RO**	13.9	$1.90	$0.15	$152,799	1220	445,300	61,003	3,288	64,291	87,211	137,635
MT0030180	YELLOWSTONE ENERGY LIMITED PARTNERSHIP FACILITY	***0.2494***	***Level 3***	**100% RO**	13.9	$3.47	$0.28	$278,174	1220	445,300	111,058	5,986	117,043	158,769	250,567
MT0030279	MONTANORE MINERALS CORP MONTANORE MINE	***0.01584***	***Level 5***	**100% RO**	6.5	$0.10	$0.01	$8,262	490	178,850	2,833	380	3,213	4,452	7,179
MT0030350	REC ADVANCED SILICON MATERIALS LLC	***1.152***	***Level 3***	**100% RO**	13.9	$16.01	$1.28	$1,284,909	1220	445,300	512,986	27,648	540,634	733,370	1,157,390
MT0030643	SLEEPING BUFFALO HOT SPRINGS - LAGOON	***0.0129***	***Level 1***	**100% RO**	19	$0.25	$0.02	$19,667	1610	587,650	7,581	310	7,890	10,840	17,331
MT0030660	PINNACLE GAS RESOURCES - COAL CREEK DEVELOPMENT UNIT	***0***	***Level 1***	**100% RO**	19	$0.00	$0.00	$0	1610	587,650	0	0	0	0	0
MT0030724	FIDELITY - TONGUE RIVER PROJECT WTF	***2.44***	***Level 4***	**100% RO**	13	$31.72	$2.55	$2,545,295	980	357,700	872,788	58,560	931,348	1,313,142	2,153,090
MT0030741	HEADWATERS LIVESTOCK AUCTION	***0***	***CAFO***	**100% RO**	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	0	#N/A	#N/A	#N/A
MT0031411	WOLF MOUNTAIN COAL	***0.0048***	***Level 1***	**100% RO**	19	$0.09	$0.01	$7,318	1610	587,650	2,821	115	2,936	4,034	6,449
MT0031534	CATTLE DEVELOPMENT CENTER	***0***	***CAFO***	**100% RO**	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	0	#N/A	#N/A	#N/A
MT0031593	JAMES GUERCIO - OW RANCH	***0.19***	***Level 4***	**100% RO**	13	$2.47	$0.20	$198,199	980	357,700	67,963	4,560	72,523	102,253	167,659
MT0031623	IOFINA NATURAL GAS WATER TREATMENT FACILITY	***0***	***Level 5***	**100% RO**	6.5	$0.00	$0.00	$0	490	178,850	0	0	0	0	0

Table A-1. Base Criteria Calculations Using 5% Interest Rate

WERF Numbers									
Level	**Description**	**Capital Cost ($/gpd)**	**Operations ($1/ MG/day Treated)**		Annualization	***0.08024***		***20 years, 5% rate***	
Level 1	No N and P removal	9.3	250		Factor:				
Level 2	1 mg/l TP; 8 mg/l TN	12.7	350			20 years, 5% rate		0.08024	
Level 3	0.1-0.3 mg/l TP; 4-8 mg/l TN	14.4	640			20 years, 7% rate		0.09439	
Level 4	<0.1 mg/l TP; 3 mg/l TN	15.3	880						
Level 5	<0.01 mg/l TP; 1 mg/l TN	21.8	1370						
100% RO	<0.01 mg/l TP; 1 mg/l TN	**28.3**	**1860**						

Table A-2. Base Criteria Calculations Using 7% Interest Rate

NPDES ID	Facility Name	Flow (mgd)	Current Level of Nutrient Treatment	Required Level of Nutrient Treatment under the Criteria	Capital Cost per MGD ($million / MGD)	Facility Upgrade Capital Costs ($million)	Annualized Capital Costs (Assumed 20 years, 7% rate; $million/year)	Annualized Capital Costs (Assumed 20 years, 7% rate; $/year)	Operations ($1/ MG/day Treated)	Operations Costs ($/ year/ 1 MGD)	Facility Upgrade Operations Costs (annual) based on Facility MGD	Membrane Replacement Cost ($24,000 /yr/1 MGD)*Actual Flow	Total Operations costs including membrane replacement	Total Operations costs including membrane replacement + Labor Low (15%)	Total Operations costs including membrane replacement + Labor Hi (48%)
MT0000264	CENEX HARVEST STATES COOP.	2.174	Level 1	*100% RO*	19	$41.31	$3.90	$3,898,994	1610	587,650	1,277,551	52,176	1,329,727	1,914,576	3,201,244
MT0000256	CONOCOPHILLIPS - BILLINGS REFINERY	2.66	Level 1	*100% RO*	19	$50.54	$4.77	$4,770,618	1610	587,650	1,563,149	63,840	1,626,989	2,342,582	3,916,886
MT0000019	BN WHITEFISH FACILITY	0.096	Level 3	*100% RO*	13.9	$1.33	$0.13	$125,958	1220	445,300	42,749	2,304	45,053	63,946	105,513
MT0000094	JOHN R DAILY INC	0.3	Level 3	*100% RO*	13.9	$4.17	$0.39	$393,619	1220	445,300	133,590	7,200	140,790	199,833	329,727
MT0000191	MONTANA RESOURCES	5.04	Level 4	*100% RO*	13	$65.52	$6.18	$6,184,624	980	357,700	1,802,808	120,960	1,923,768	2,851,462	4,892,388
MT0000230	MONTANA SULPHUR & CHEMICAL CO	3.28752	Level 3	*100% RO*	13.9	$45.70	$4.31	$4,313,429	1220	445,300	1,463,933	78,900	1,542,833	2,189,847	3,613,279
MT0000248	SIDNEY SUGARS INCORPORATED	1.3	level 1	*100% RO*	19	$24.70	$2.33	$2,331,505	1610	587,650	763,945	31,200	795,145	1,144,871	1,914,268
MT0000281	WESTERN SUGAR COOPERATIVE	9.36	level 1	*100% RO*	19	$177.84	$16.79	$16,786,838	1610	587,650	5,500,404	224,640	5,725,044	8,243,070	13,782,726
MT0000302	MDU - LEWIS & CLARK PLANT	42.43	Level 3	*100% RO*	13.9	$589.78	$55.67	$55,670,777	1220	445,300	18,894,079	1,018,320	19,912,399	28,263,015	46,634,372
MT0000388	MONTANA RAIL LINK -LIVINGSTON RAIL YARD	0.14256	Level 3	*100% RO*	13.9	$1.98	$0.19	$187,048	1220	445,300	63,482	3,421	66,903	94,961	156,686
MT0000396	CORETTE THERMAL PLANT	131	Level 3	*100% RO*	13.9	$1,820.90	$171.88	$171,880,078	1220	445,300	58,334,300	3,144,000	61,478,300	87,260,312	143,980,738
MT0000451	ASH GROVE CEMENT COMPANY	0.037728	Level 3	*100% RO*	13.9	$0.52	$0.05	$49,501	1220	445,300	16,800	905	17,706	25,131	41,466
MT0000477	EXXONMOBIL REFINING & SUPPLY	2.7	Level 1	*100% RO*	19	$51.30	$4.84	$4,842,357	1610	587,650	1,586,655	64,800	1,651,455	2,377,809	3,975,786
MT0000485	TRIDENT PLANT	0.0072	Level 2	*100% RO*	15.6	$0.11	$0.01	$10,602	1510	551,150	3,968	173	4,141	5,731	9,230
MT0000884	BIG SKY COAL COMPANY - BIG SKY MINE	3.045	Level 3	*100% RO*	13.9	$42.33	$4.00	$3,995,228	1220	445,300	1,355,939	73,080	1,429,019	2,028,303	3,346,728
MT0000892	DECKER COAL CO (WEST MINE)	1.12	Level 4	*100% RO*	13	$14.56	$1.37	$1,374,361	980	357,700	400,624	26,880	427,504	633,658	1,087,197
MT0020460	YELLOWSTONE BOYS & GIRLS RANCH	0.02	Level 1	*100% RO*	19	$0.38	$0.04	$35,869	1610	587,650	11,753	480	12,233	17,613	29,450
MT0021229	WESTMORELAND RESOURCES INC - ABSALOKA MINE	1.44	Level 3	*100% RO*	13.9	$20.02	$1.89	$1,889,369	1220	445,300	641,232	34,560	675,792	959,197	1,582,689
MT0021431	MT BEHAVIORAL HEALTH INC WWTP	0.1	level 1	*100% RO*	19	$1.90	$0.18	$179,347	1610	587,650	58,765	2,400	61,165	88,067	147,251
MT0023566	ELKHORN HEALTH CARE WWTP	0.015	level 1	*100% RO*	19	$0.29	$0.03	$26,902	1610	587,650	8,815	360	9,175	13,210	22,088
MT0023604	WESTMORELAND SAVAGE CORP - SAVAGE MINE	0.576	Level 3	*100% RO*	13.9	$8.01	$0.76	$755,748	1220	445,300	256,493	13,824	270,317	383,679	633,076
MT0023639	BOULDER HOT SPRINGS WWTP	0.085	Level 3	*100% RO*	13.9	$1.18	$0.11	$111,525	1220	445,300	37,851	2,040	39,891	56,619	93,423
MT0023965	WESTERN ENERGY CO - ROSEBUD MINE	0	Level 3	*100% RO*	13.9	$0.00	$0.00	$0	1220	445,300	0	0	0	0	0
MT0024210	DECKER COAL CO (EAST MINE)	0.89	Level 4	*100% RO*	13	$11.57	$1.09	$1,092,126	980	357,700	318,353	21,360	339,713	503,532	863,934
MT0024619	SPRING CREEK MINE	0.02	Level 3	*100% RO*	13.9	$0.28	$0.03	$26,241	1220	445,300	8,906	480	9,386	13,322	21,982
MT0024716	STILLWATER MINING COMPANY	0.94176	Level 4	*100% RO*	13	$12.24	$1.16	$1,155,641	980	357,700	336,868	22,602	359,470	532,816	914,178
MT0026808	STILLWATER MINING COMPANY	0.648	level 3	*100% RO*	13.9	$9.01	$0.85	$850,216	1220	445,300	288,554	15,552	304,106	431,639	712,210
MT0027821	BEAVERHEAD TALC MINE	0.10368	Level 1	*100% RO*	19	$1.97	$0.19	$185,947	1610	587,650	60,928	2,488	63,416	91,308	152,670
MT0028321	EXXON MOBIL BILLINGS REFINERY	5.86368	Level 1	*100% RO*	19	$111.41	$10.52	$10,516,308	1610	587,650	3,445,792	140,728	3,586,520	5,163,966	8,634,348
MT0028428	MONTANA TUNNELS MINING INC	0	level 5	*100% RO*	6.5	$0.00	$0.00	$0	490	178,850	0	0	0	0	0
MT0028584	LUZENAC AMERICA INC - YELLOWSTONE MINE	0	Level 3	*100% RO*	13.9	$0.00	$0.00	$0	1220	445,300	0	0	0	0	0

Table A-2. Base Criteria Calculations Using 7% Interest Rate

NPDES ID	Facility Name	Flow (mgd)	Current Level of Nutrient Treatment	Required Level of Nutrient Treatment under the Criteria	Capital Cost per MGD ($million / MGD)	Facility Upgrade Capital Costs ($million)	Annualized Capital Costs (Assumed 20 years, 7% rate; $million/year)	Annualized Capital Costs (Assumed 20 years, 7% rate; $/year)	Operations ($1/ MG/day Treated)	Operations Costs ($/ year/ 1 MGD)	Facility Upgrade Operations Costs (annual) based on Facility MGD	Membrane Replacement Cost ($24,000 /yr/1 MGD)*Actual Flow	Total Operations costs including membrane replacement	Total Operations costs including membrane replacement + Labor Low (15%)	Total Operations costs including membrane replacement + Labor Hi (48%)
MT0028983	BULL MOUNTAIN MINE #1	0.36	Level 3	***100% RO***	13.9	$5.00	$0.47	$472,342	1220	445,300	160,308	8,640	168,948	239,799	395,672
MT0029891	BARRETTS MINERALS INC	1.5768	Level 3	***100% RO***	13.9	$21.92	$2.07	$2,068,859	1220	445,300	702,149	37,843	739,992	1,050,321	1,733,044
MT0029980	MONTANA AVIATION RESEARCH CO	0.05	Level 3	***100% RO***	13.9	$0.70	$0.07	$65,603	1220	445,300	22,265	1,200	23,465	33,305	54,954
MT0030015	M & W MILLING & REFINING INC	0.0216	Level 1	***100% RO***	19	$0.41	$0.04	$38,739	1610	587,650	12,693	518	13,212	19,022	31,806
MT0030031	ASARCO LLC - MIKE HORSE/ANACONDA MINE WATER TREATMENT SYSTEM	0.0468	Level 1	***100% RO***	19	$0.89	$0.08	$83,934	1610	587,650	27,502	1,123	28,625	41,215	68,914
MT0030066	COLUMBIA FALLS ALUMINUM CO	0.600905	Level 3	***100% RO***	13.9	$8.35	$0.79	$788,424	1220	445,300	267,583	14,422	282,005	400,268	660,448
MT0030147	ASARCO INC	0.136994	Level 3	***100% RO***	13.9	$1.90	$0.18	$179,745	1220	445,300	61,003	3,288	64,291	91,253	150,569
MT0030180	YELLOWSTONE ENERGY LIMITED PARTNERSHIP FACILITY	0.2494	Level 3	***100% RO***	13.9	$3.47	$0.33	$327,228	1220	445,300	111,058	5,986	117,043	166,128	274,113
MT0030279	MONTANORE MINERALS CORP MONTANORE MINE	0.01584	Level 5	***100% RO***	6.5	$0.10	$0.01	$9,719	490	178,850	2,833	380	3,213	4,671	7,878
MT0030350	REC ADVANCED SILICON MATERIALS LLC	1.152	Level 3	***100% RO***	13.9	$16.01	$1.51	$1,511,495	1220	445,300	512,986	27,648	540,634	767,358	1,266,151
MT0030392	M&K OIL COMPANY - WRIGHT CREEK WATER DISPOSAL FACILITY	0.0168	Level 3	***100% RO***	13.9	$0.23	$0.02	$22,043	1220	445,300	7,481	403	7,884	11,191	18,465
MT0030643	SLEEPING BUFFALO HOT SPRINGS - LAGOON	0.0129	Level 1	***100% RO***	19	$0.25	$0.02	$23,136	1610	587,650	7,581	310	7,890	11,361	18,995
MT0030660	PINNACLE GAS RESOURCES - COAL CREEK DEVELOPMENT UNIT	0	Level 1	***100% RO***	19	$0.00	$0.00	$0	1610	587,650	0	0	0	0	0
MT0030678	BARRETTS MINERALS - REGAL MINE	0.144	Level 3	***100% RO***	13.9	$2.00	$0.19	$188,937	1220	445,300	64,123	3,456	67,579	95,920	158,269
MT0030724	FIDELITY - TONGUE RIVER PROJECT WTF	2.44	Level 4	***100% RO***	13	$31.72	$2.99	$2,994,144	980	357,700	872,788	58,560	931,348	1,380,470	2,368,537
MT0030741	HEADWATERS LIVESTOCK AUCTION	0	CAFO	***100% RO***	**#N/A**	**#N/A**	**#N/A**	**#N/A**	**#N/A**	**#N/A**	**#N/A**	0	**#N/A**	**#N/A**	**#N/A**
MT0031411	WOLF MOUNTAIN COAL	0.0048	Level 1	***100% RO***	19	$0.09	$0.01	$8,609	1610	587,650	2,821	115	2,936	4,227	7,068
MT0031534	CATTLE DEVELOPMENT CENTER	0	CAFO	***100% RO***	**#N/A**	**#N/A**	**#N/A**	**#N/A**	**#N/A**	**#N/A**	**#N/A**	0	**#N/A**	**#N/A**	**#N/A**
MT0031593	JAMES GUERCIO - OW RANCH	0.19	Level 4	***100% RO***	13	$2.47	$0.23	$233,151	980	357,700	67,963	4,560	72,523	107,496	184,435
MT0031623	IOFINA NATURAL GAS WATER TREATMENT FACILITY	0	Level 5	***100% RO***	6.5	$0.00	$0.00	$0	490	178,850	0	0	0	0	0

WERF Numbers

Level	Description	Capital Cost ($/gpd)	Operations ($1/ MG/day Treated)	Annualization Factor:		
					0.09439	***20 years, 7% rate***
Level 1	No N and P removal	9.3	250			
Level 2	1 mg/l TP; 8 mg/l TN	12.7	350		**20 years, 5% rate**	**0.08024**
Level 3	0.1-0.3 mg/l TP; 4-8 mg/l TN	14.4	640		**20 years, 7% rate**	**0.09439**
Level 4	<0.1 mg/l TP; 3 mg/l TN	15.3	880			
Level 5	<0.01 mg/l TP; 1 mg/l TN	21.8	1370			
100% RO	<0.01 mg/l TP; 1 mg/l TN	***28.3***	***1860***			

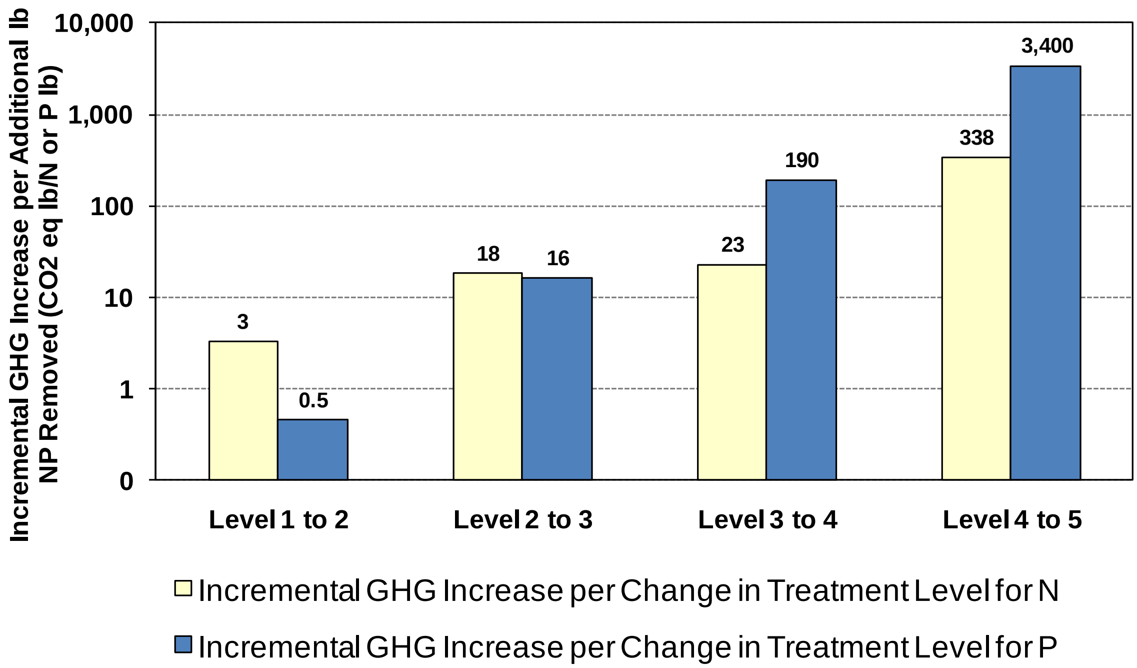
APPENDIX B - ASSUMPTIONS IN THE COST ANALYSIS

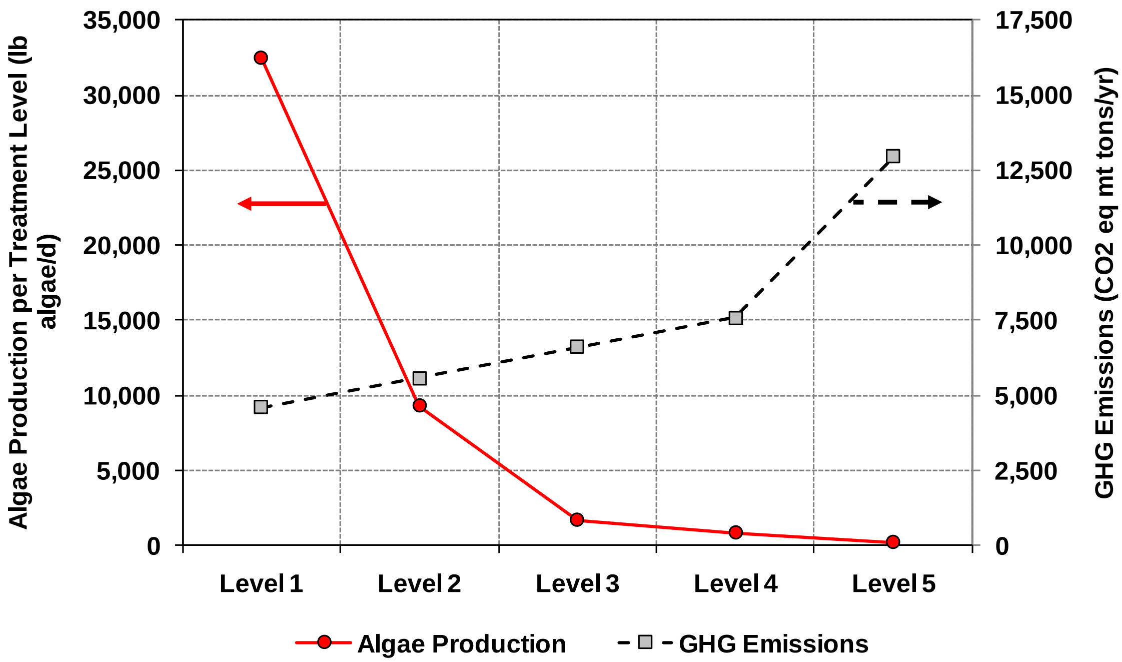
DESCRIPTION OF THE KEY CALCULATIONS/ DETAILS IN THE SPREADSHEET

- The spreadsheet numbers are intended to provide ROUGH ESTIMATES for discussion purposes and do not reflect the site-specific conditions at each plant.
- The cost estimates for upgrading WWTPs are obtained from the Interim WERF study: "Finding the Balance Between Wastewater Treatment Nutrient Removal and Sustainability, Considering Capital and Operating Costs, Energy, Air and Water Quality and More" (Falk, et al., 2011). This report is in Draft form and the capital costs are anticipated to increase in the final report based on feedback from the technical reviewers. Based on actual costs observed in EPA Region 1, Region 1 considered the capital costs to be higher than experienced in the final facility plan.
- Reverse osmosis is believed to be the technology that would allow WWTPs to have the best chance at meeting Montana's base numeric criteria at this time. It is ultimately assumed that 100% of wastewater would need to go through the reverse osmosis process to reach Montana's TN standards. Thus, the WERF cost estimate numbers for WERF Level 5 are increased using assumptions from adding 100% RO.
- The design flows of new or upgraded wastewater plants at businesses would be the same as current flows, unless otherwise noted. This is a conservative assumption.
- Capital costs were assumed to cover a 20-year bond with 5% interest. An alternate assumption used a 7% interest rate.
- For the Montana businesses in this analysis with advanced treatment, the cost associated with the WERF level they are currently treating to is subtracted from WERF level 5 costs (plus 100% RO) in the study. That means that all businesses in our sample already treating at WERF level 2 will have the same estimated unit capital and O&M costs per MGD flow to meet base numeric criteria. Estimate total costs will differ based on facility flow.
- Operation costs in the WERF study, and therefore in this analysis, include energy and chemical costs only and do not include labor and maintenance cost. As such, the O&M cost numbers in this analysis are on the low side. An alternate assumption in the sensitivity analysis addresses this issue by adding labor costs.
- The costs in this demonstration do not include existing treatment plant abandonment, so they may underestimate total costs.
- Capital and O&M costs for businesses to get up to WERF 5 are based on building from scratch, assuming that no infrastructure exists.
- To get to RO, a membrane Replacement Cost is added which is estimated at $24,000/yr/1 MGD. Brine disposal costs are already included within the WERF numbers, but may be higher in Montana due to lack of suitable disposal sites in-state.
- Design flow of a given business treatment plant was used to determine the capital costs and actual flow was used for the Operations costs. Flows for businesses were taken from wastewater permits.

APPENDIX C - NON MONETARY COSTS DISCUSSION

Source: DRAFT Interim WERF study "*Finding the Balance Between Wastewater Treatment Nutrient Removal and Sustainability, Considering Capital and Operating Costs, Energy, Air and Water Quality and More*" (Falk, et al., 2011).





Nearly 95 percent of the potential algae production is eliminated when changing from Levels 1 to Level 3 with a 44 percent increase in Green House Gas (GHG) emissions. An additional 4 percent of potential algae production (with respect to Level 1) is eliminated in Levels 4 and 5, while nearly doubling the GHG emissions (6,590 to 12,950 mt CO_2 equivalents/year).

GHG emissions associated with chemical usage (production and distribution) at WWTPs are often overlooked. It is critical that the amount of GHG emissions associated with each individual chemical during production is incorporated into the evaluation. Several chemicals are mined in a few locations globally and the mining and transportation of the chemicals contribute to global GHG emissions. For example, the closest ferric mine to the United States is in Jamaica. For treatment plants located on the west coast, Jamaica is several thousand miles away and requires hauling. Additionally, the distance travelled, fuel type and truck fuel efficiency all play a role in quantifying their respective GHG emissions.

Recalcitrant dissolved organic nitrogen, commonly referred to as refractory dissolved organic nitrogen (rDON), impairs a WWTPs ability to reliably achieve low TN objectives. Effluent limits that require nitrogen values of 2 mg N/L or less might require the use of expensive and energy intensive strategies, such as reverse osmosis, that result in elevated GHG emissions.

Using reverse osmosis to achieve extremely low levels of nitrogen and phosphorus increases costs and GHG emissions. Brine reject management remains a challenge for reverse osmosis applications, especially for inland applications.

APPENDIX D - RO LITERATURE

Source: Tetratech, Alejandro Escobar

REVERSE OSMOSIS EFFICACY FOR TN REMOVAL

Bench, pilot, and full-scale studies describing Reverse Osmosis (RO) treatment to very low Total Nitrogen (TN) levels were obtained and summarized. The Montana Draft Nutrient Criteria are proposed to be between 0.3-1.2 mg/L TN depending on Level III Ecoregion.

The nitrogen removal capabilities described in these studies are described in **Table D-1** and more detailed descriptions of the studies reviewed are provided following **Table D-1**. Species of nitrogen removal rates are also included as percentages in **Table D-1**. Merlo et al. (2011) noted that organic nitrogen may not be reliably removed by RO treatment. The data provided in **Table D-1** are generally average (mg/L) values and the percentages listed are removal rates. Studies do not always report influent conc., effluent conc., and percent removal for all nitrogen species. Associated cost data was most often not available, but when available it was included in **Table D-1**.

A number of studies have been done on facilities with low influent concentrations of TN, especially after pretreatment by various means including biological nutrient removal, microfiltration, ultrafiltration, and other standard wastewater treatment processes.

Table D-1 also includes information obtained from RO technology manufacturers. Five manufacturers were contacted via e-mail and responses were received from two of the manufacturers: CSM Filters and Pure Aqua, Inc. The nitrogen rejection by Pure Aqua, Inc. filters was said to be highly variable and the manufacturer did not provide specific concentrations other than ranges of nitrogen removal. The CSM Filter data is summarized in **Table D-1** below.

A number of factors may influence the reliability of RO in meeting criteria. Studies generally listed average values without any reliability data. Removal of dissolved nitrogen species is dependent on the initial concentrations in the influent and characteristics of the wastewater. Very high and very low TDS, temperature, pH and may impact some membrane's nitrogen removal properties. Also some variability in RO efficacy in nitrogen species removal is seen from site to site and as a result of differing pretreatment processes prior to RO. Membrane rejection values can only be guaranteed after the RO process has been defined.

A 2-pass RO system for wastewater treatment is quite expensive to operate according to a CSM Filter representative, who was not aware of many applications of 2-pass systems in practice for TN removal. Some studies from the literature indicated arrays of membranes including a mixture of nanofiltration and RO (Drewes, et al., 2005; Ushikoshi, et al., 2002) or Forward Osmosis and RO (Cath, et al., 20006).

International studies are included in **Table D-1** from countries such as Norway, Poland, Australia, Finland, China, Czech Republic, South Africa, Japan, and France. Both industrial and municipal wastewater streams were also included.

Table D-1. Summary of studies describing Reverse Osmosis treatment and Nitrogen species removal results[7]

Source	Type	Location	Effluent Type	Description	TN	NH_3-N	NO_3	NO_2	TKN	N_{org}	Cost
Bilstad 1995	Pilot	Norway	Mixture of municipal and industrial wastewater (wool scouring)	Tubular RO membrane	Inf: 24.1-33.5 Eff: 0.8-2.2 94-97%	Inf: 15.3-29.5 Eff:0.5-1.3	Inf: 0-10 Eff: 0-0.5		-	-	2x$ as BNR
Bohdziewicz 2005	Bench-scale	Swine Processing Plant Uni-Lang – Wrzosowa, Poland	Industrial - Meat processing plant	High pressure membrane (SEPA CF-HP)	Inf: 13 Eff: 1.3 90.0%	-	-	-	-	-	NA
Cath 2006	Full-scale	Coffin Butte Landfill – Corvallis, OR	Industrial – Landfill leachate	One FO membrane and four RO membranes	-	Inf: 1110 Eff: 1.6 99.9%	-	-	Inf: 780 Eff: ND 100%	-	NA
Drewes 2005	Pilot	West Basin WRP – Segundo, CA	Not nitrified micofiltered secondary wastewater effluent	Low pressure membranes Toray TMG-10 Dow NF-90	-	Inf:31.5 (est) Eff: 1.2 Inf:37.4 (est) Eff:2.3	Inf: <1 mg/L Eff: ND Eff: ND	-	-	-	NA
Ghayeni 1998	Pilot	Sydney, Australia	Secondary municipal wastewater w/ biological nutrient removal	Two membranes Film Tec - NF45 Fluid Systems - TFCL	-	No measureable amount of ammonia in Inf or Eff	Inf: 0.37 (BOTH) Eff:0.3 Eff:0.2		-	-	NA
Häyrynen 2008	Bench-scale	Gold Mine - Finland	Industrial – Gold mine effluent	Four RO membranes Filmtec SW30HR HydranauticsESPA2 KOCH TFC ULP Sepro – RO1	-	Inf: 9.53 Eff: 1.64 82.8% Eff: 0.54 94.3% Eff:0.81 91.5% Eff:0.80 91.6%	Inf: 15.6 Eff: 0.94 93.9% Eff: 0.40 97.4% Eff:0.36 97.7% Eff:0.61 96.0%	-	-	-	0.31-0.34 Euro/m^3 for plant capacities of 250,000 m^3 and 1,000,000 m^3
Häyrynen 2008	Bench-scale	Chromite Mine - Finland	Industrial – Chromite mine effluent	Four RO membranes Filmtec SW30HR HydranauticsESPA2 KOCH TFC ULP Sepro – RO1	-	Inf: 5.50 Eff: 0.86 84.4% Eff: 0.33 94.0% Eff:0.60 89.1% Eff:0.60 89.1%	Inf: 20.8 Eff: 1.66 92.0% Eff: 1.33 93.6% Eff:1.94 90.7% Eff:0.89 95.7%	-	-	-	0.31-0.34 Euro/m^3 for plant capacities of 250,000 m^3 and 1,000,000 m^3
Häyrynen 2008	Bench-scale	Phosphate Mine - Finland	Industrial – Phosphate mine effluent	Four RO membranes Filmtec SW30HR HydranauticsESPA2 KOCH TFC ULP Sepro – RO1	-	Inf: 11.8 Eff: 1.07 90.9% Eff: 0.65 94.5% Eff:0.82 93.0% Eff:1.64 86.1%	Inf: 44.0 Eff: 1.17 97.3% Eff: 1.76 96.0% Eff:2.68 93.9% Eff:3.05 93.1%	-	-	-	0.31-0.34 Euro/m^3 for plant capacities of 250,000 m^3 and 1,000,000 m^3

[7] Nitrogen species values are average mg/L values unless otherwise noted. The percentages provided are removal percentages. When multiple RO processes or filters were tested in a study, they are all listed in **Table D-1** in separate rows.

Table D-1. Summary of studies describing Reverse Osmosis treatment and Nitrogen species removal results[7]

Source	Type	Location	Effluent Type	Description	TN	NH_3-N	NO_3	NO_2	TKN	N_{org}	Cost
Huang 2011	Bench-scale	Iron and Steel Manufacturer - China	Industrial – Iron and Steel Manufacturer mixed effluent	Two treatment processes UF/RO process CW/UF/RO process	-	Inf: 1.82 Eff: 0.11 94.0% Inf:0.40 Eff:ND 100%	-	-	-	-	NA
Merlow 2011	Pilot	HAARF treatment plant - Escondido, CA	Municipal wastewater	Toray TML-10 membrane	Inf: 19.4 Eff: 1.88 91.0%	Inf: 8.78 Eff: 0.61 91.8%	Inf: 6.57 Eff: 0.45 94.5%	Inf: 2.17 Eff: 0.19 89.4%	Inf: 10.8 Eff: 1.22	Inf: 1.97 Eff: 0.61 68.2%	NA
Merlow 2011	Full-scale	GWRS – Fountain Valley, CA	Secondary Municipal wastewater	Hydranautics ESPA2 membrane	-	Inf: 21.0 Eff: 1.24 93.3%	-	-	Inf: 23.0 Eff: 1.32	Inf: 2.03 Eff: 0.10 94.4%	NA
Merlow 2011	Full-scale	Vander Lans AWTF – Southern CA	Tertiary Municipal wastewater	Hydranautics ESPA2 membrane	Inf: 8.91 Eff: 1.47 89.5%	Inf: 1.07 Eff: 0.17 81.2%	Inf: 5.85 Eff: 1.22 89.8%	Inf: 0.08 Eff: 0.06 26.3%	-	Inf: 1.91 Eff: <0.2 93.5%	NA
Merlow 2011	Full-scale	Scottsdale Water Campus AWTF – Scottsdale, AZ	Municipal wastewater	RO	-	Inf: 0.88 Eff: 0.16 78.6%	-	-	-	Inf: 1.05 Eff: 0.14 80.7%	NA
Merlow 2011	Pilot	Miami-Dade County, FL	Municipal wastewater	Six RO pilot processes CSM Koch Toray Dow Hydranautics Hydranautics	No Influent Effluent 1.49 1.51 2.01 2.30 1.81 2.67	-	-	-	-	No Influent Effluent 0.34 0.32 0.38 0.54 0.39 0.66	NA
Merlow 2011	Pilot	Luggage Point AWTF – Queensland, Australia	Municipal wastewater	Toray TML-4040 membrane	88%	83%	87%	94%	88%	92%	NA
Merlow 2011	Pilot	Bureau of Reclamation	Municipal wastewater	Dow RO (BW30-4040) membrane	-	Inf: 0.79 Eff: 0.20	Inf: 8.46 Eff: 0.47	-	Inf: 1.55 Eff: 0.30	Inf: 0.76 Eff: 0.10 86.6%	NA
Merlow 2011	Pilot	City of San Diego, CA	Municipal wastewater	Dow RO (BW30-4040) membrane	-	-	-	-	-	No Influent Eff: <0.18-0.25	NA
Šir 2011	Pilot	Landfill in northern Bohemia, Czech Republic	Industrial – Hazardous Landfill Leachate	Filmtec SW30-4040 membrane	-	Inf: 142 Eff: 8.54 94.0%	Inf: 0.83 Eff: 0.04 95.2%	-	-	-	NA
Schoeman 2003	Full-scale	South Africa	Groundwater treatment for drinking water	Delta 4040-LHA-CPA2 membrane	-	-	Inf: 42.46 Eff: 0.85 98.0%	-	-	-	Capital: $29,900 Op: $0.50/m^3 for a 50 m^3/d output
Ushikoshi 2002	Full-scale	Landfill in Japan	Industrial – Landfill Leachate	High Pressure 2-stage RO/NF	Inf: 12.9-164 Eff: <1-2 92.2-98.9%	Inf: 3.9-53 Eff: 0.29-1.53 90.2-98.4%	-	-	-	-	-
Vourch 2008	Bench-scale	France	Industrial – Wastewater from three Dairy Farms	KOCH TFC HR SW 2540 membrane	-	-	Eff: <2	-	96.1%	-	NA
Qin 2005	Pilot	Coconut Island, Hawaii	Industrial – Aquaculture Wastewater	Flimtec XLE-4040 membrane	94.7%	Eff:<0.05	Eff:<0.03	Eff:<0.003	-	-	$4.00/m^3 permeate

Table D-1. Summary of studies describing Reverse Osmosis treatment and Nitrogen species removal results[7]

Source	Type	Location	Effluent Type	Description	TN	NH_3-N	NO_3	NO_2	TKN	N_{org}	Cost
CSM Filter	Full-scale	Orange County, CA	Municipal wastewater	RO	Inf: 12.5 Eff: 0.7 96.3%	Inf: 1.0 Eff: 0.3 78.6%	Inf: 11.4 Eff: 0.35 97.9%	Inf: 0.09 Eff: <0.002 99.1%	Inf: 1.0 Eff: 0.3 81.3%	-	Cap: $481M. (70 mgd plant). $600 acre/foot (total process)
CSM Filter	Full-scale	Los Angeles, CA	Municipal wastewater	RO	-	Inf: 36.0 Eff: 2.4 93.3%	Inf: 2.36 Eff: 0.41 82.6%	Inf: 6.97 Eff: 0.69 90.1%	-	-	NA
CSM Filter	Full-scale	Richmond, CA	Municipal wastewater	RO	-	Inf: 0.98 Eff: 0.3 89.6%	Inf: 24.0 Eff: 1.5 98.4%	Inf: 0.013 Eff: 0.0025 97.1%	Inf: 1.4 Eff: 1.0 71.4%	-	NA
CSM Filter	Bench-scale	Anaheim, CA	NA	FE – low fouling BE – brackish BLR – low pressure HUE – High TOC	97.6% 98.1% 97.5% 98.3%	98.2% 98.5% 98.0% 98.7%	88.0% 93.1% 89.5% 94.7%	88.8% 93.1% 89.5% 94.3%	-	-	NA

STUDY SUMMARIES

Bilstad, T. 1995. Nitrogen separation from domestic wastewater by reverse osmosis.

Norwegian pilot-scale study of nitrogen removal using spiral-wound membranes and tubular membranes for RO. Results were only included for the Tubular membrane output. Three separate runs of the treatment setup were completed with the results summarized in **Table D-1**. Treatment costs using membrane separation by RO were noted to be twice as expensive as biological nutrient removal (BNR). Tubular membranes are considered unrealistic for high-volume influent such as that in domestic wastewater for nitrogen removal due to high costs.

Bohdziewicz, J. and E. Sroka. 2005. Integrated System of activated sludge-reverse osmosis in the treatment of the wastewater from the meat industry.

This study used samples from a swine processing facility in southern Poland to test the efficacy of a hybrid system of combining the biological methods of activated sludge in an SBR and reverse osmosis. Initial effluent was TN 198 mg/L, 13 mg/L after pretreatment with SBR, and 1.3 mg/L after RO process.

Cath, T.Y. et al. 2006. Forward osmosis: Principles, applications, and recent developments.

This paper reviewed a number of applications of osmosis or forward osmosis, many of which did not have applicable results for nitrogen. However, the Coffin Butte Landfill in Corvallis, OR did have results for nitrogen. RO treatment at the landfill started as a pilot project to treat landfill leachate and as a result of the success of the treatment became full-scale in 1998. A combination of traditionally used RO through four filters (Osmonics-CE, Osmonics-CD, Hydranautics-LFC1, and Hydranautics-LFC3) and forward osmosis (FO) through a CTA-Osmotek filter were used to treat most contaminants to greater than 99% rejection.

Drewes, J.E. et al. 2005. Can Nanofiltration and Ultra-low Pressure Reverse Osmosis Membranes Replace RO for the Removal of Organic Micropollutants, Nutrients, and Bulk Organic Carbon? – A Pilot-scale Investigation.

This pilot-scale study was a low pressure nanofiltration (NF) and ultra-low pressure reverse osmosis (ULPRO) pilot study run at the West Basin Water Recycling Plant in Segundo, California, a water reuse facility. Two lower pressure membranes were tested on the pilot-scale skid: a Toray TMG-10 (ULPRO) and a Dow NF-90 (NF). Results indicated that nitrogen species, along with other contaminants, could be removed to levels comparable to traditional RO processes using these low pressure filters.

Ghayeni, S.B.S. et al. 1998. Water reclamation from municipal wastewater using combined microfiltration-reverse osmosis (ME-RO): Preliminary performance data and microbiological aspects of system operation.

Two membranes, a traditional RO membrane and a nanofiltration membrane, were studied on a pilot-scale analysis at a wastewater treatment plant in Sydney, Australia. The pretreatment process included an activated sludge process with biological nitrogen and phosphorus removal and microfiltration prior to the RO treatment, so nutrient levels were already very low prior to RO. Prior to RO Ammonia levels were ND and NOx levels were already very low at 0.37. Phosphate levels were at 3.5 mg/L, and both filters reduced the levels to ND (reported as 0). The RO filters used in the pilot study were a Fluid Systems TFCL (cross-linked aromatic polyamide, thin-film composite) and a Film Tec NF45 membrane (nanofiltration, polypiperazine amide, thin-film composite).

Häyrynen, K. et al. 2008. Separation of nutrients from mine water by reverse osmosis for subsequent biological treatment.

This study examined the treatment capabilities for nutrient removal on three different mines in Finland. First a bench-scale analysis was done testing the capabilities of four different membranes (Filmtec SW30HR, Hydranautics ESPA2, KOCH, TFC ULP, Sepro RO1). In addition a pilot-scale analysis was performed using the Sepro membrane and effluent from the Gold and Chromite mines, but nutrient removal results were not reported. Cost data were included including a breakdown for two plant capacities (250,000 m3 and 1,000,000 m3) by energy, chemicals, membranes, labor, and capital costs (calculations in Euro).

Huang, X. et al. 2011. Advanced treatment of wastewater from an iron and steel enterprise by a constructed wetland/ultrafiltration/reverse osmosis process.

This study evaluates the effectiveness of two different treatment processes at filtering contaminants from the Baosteel iron and steel manufacturing plant. The two treatment processes were an UF/RO: ultrafiltration system followed by a reverse osmosis process and CW/UF/RO: a constructed wetland, followed by the ultrafiltration system and the reverse osmosis process. The RO membrane tested in both cases was a Filmtec BW30FR-based polyamide composite membrane.

Merlow, R. et al. 2011. Analysis of Organic Nitrogen Removal in Municipal Wastewater by Reverse Osmosis.

A synthesis of pilot-scale and full-scale case studies on various treatment processes and capabilities of nitrogen species removal using RO. Facility process descriptions and methods are summarized. Primary sources were not obtained at this time.

Šir, M. et al. 2011. The effect of humic acids on the reverse osmosis treatment of hazardous landfill leachate.

The potential for RO treatment at an abandoned brown coal pit in northern Bohemia, Czech Republic was evaluated. A pilot-scale study treated the landfill leachate with very minimal pretreatment by running the leachate through a Filmtec SW30-4040 membrane. The first stage concentrate was additional run through another RO membrane to further concentrate the contaminants. It was noted that ammonia nitrogen was the only indicator that still exceeded limits in the permeate and subsequent study of ammonia removal methods is needed.

Schoeman, J.J. and A. Steyn. 2003. Nitrate Removal with reverse osmosis in a rural area in South Africa.

Due to high nitrate-nitrogen and salinity levels in boreholes in South Africa, a RO plant was built to produce safe drinking water. This study examines results from this facility. A Delta 4040-LHA-CPA2 membrane was used with sand filters as a pretreatment step. Cost estimates were provided for this system including capital costs of $29,900 for a 50 m^3/day output RO plant with operational costs for denitrification of $0.50/$m^3$.

Ushikoshi, K. et al. 2002. Leachate treatment by the reverse osmosis system.

Results from a full-scale DT-Mudule system for landfill leachate treatment installed at the Clean Park KINU landfill in Yachiyo Town, Japan are presented. The treatment process includes a settling basin, sand filters, micron filters, and a two-stage RO system with a high pressure RO membrane followed by a nanofiltration membrane. Sampling is from 1999-2001.

Vourch, M. et al. 2008. Treatment of dairy industry wastewater by reverse osmosis for water reuse.

A bench-scale analysis of wastewater treatment from three dairy farms in France was summarized. A RO spiral-wound membrane (KOCH TFC HR SW 2540) to treat samples from the farms.

Qin, G. et al. 2005. Aquaculture wastewater treatment and reuse by wind-driven reverse osmosis membrane technology: a pilot study on Coconut Island, Hawaii.

This study summarizes the results of a pilot-study to treat aquaculture wastewater with a wind-driven RO system on Coconut Island, HI. The process included a cartridge filter as pretreatment before the spiral wound Filmtec XLE-4040 membrane. Detailed cost estimates are provided that indicate a relatively high $4.00/ 1$m^3$ permeate cost for the pilot study. However if scaled up to between 9000-13200 m^3/year (currently between 1500-2200 m^3/year), it is anticipated unit costs would drop to between $1.11-$1.62/m^3 permeate.

MANUFACTURERS CONTACTED

1. http://www.csmfilter.com/ - contacted 9/26/11 via e-mail 'csmusa@wjcsm.com'
2. http://www.water.siemens.com/en/products/membrane_filtration_separation/reverse_osmosis_systems_ro/Pages/Reverse_Osmosis_Pretreatment_System.aspx - contacted 9/26/11 via e-mail 'information.water@siemens.com'/'iwsinquiry.water@siemens.com'
3. http://www.reskem.com/pages/reverse-osmosis.php - contacted 9/26/11 via e-mail ‘sales@reskem.com’
4. http://www.appliedmembranes.com/Reverse_Osmosis_Systems.htm - contacted 9/26/11 via e-mail ‘sales@appliedmembranes.com’
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Identification and Assessment of Montana Reference Streams: A Follow-up and Expansion of the 1992 Benchmark Biology Study

Prepared by

Michael Suplee[1], Rosie Sada de Suplee[2], David Feldman[1] and Tina Laidlaw[3]

[1]Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau, Water Quality Standards Section, 1520 East 6th Ave, Helena, MT 59620.

[2]Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau, Water Quality Monitoring Section, 1520 East 6th Ave, Helena, MT 59620.

[3] U.S. Environmental Protection Agency, USEPA Montana Office, 10 West 15th Street, Suite 3200, Helena, MT 59626.

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EXECUTIVE SUMMARY

The following report describes the Montana Department of Environmental Quality's endeavor to identify and assess reference sites for flowing waters. Identifying reference sites is an outgrowth of the reference condition concept. The reference condition concept asserts that there exist for any group of waterbodies relatively undisturbed examples that can represent the natural biological, physical and chemical integrity of a region; therefore, reference stream sites are those that represent the reference condition. The MT Department of Environmental Quality (DEQ) is interested in reference sites because they help the Department interpret narrative water-quality standards. A number of Montana's narrative standards require that water quality be compared to "naturally occurring", and the DEQ uses reference sites to help interpret what naturally occurring is.

The work detailed in this report was undertaken from 2000 to 2005, and is a continuation and expansion of DEQ work described by Bahls et al. (1992). In 2000, the DEQ re-initiated a Reference Stream Project and began to collect data at existing reference sites (per Bahls et al. 1992) as well as at new sites that were identified around the state. In addition to conducting field sampling, in 2004 the DEQ began to assemble a comprehensive list of potential reference stream sites (and their associated data) available in the Water Quality Planning Bureau. This list included not only the sites from the DEQ Reference Stream Project, but also sites from a variety of other statewide water-quality sampling projects (e.g., the USGS Hydrologic Benchmark Network).

An evaluation process was developed and used to assess each candidate reference site in a consistent way. (Some established reference sites that had already been thoroughly reviewed using similar techniques did not go through this process, and were automatically classified as final reference sites.) The process consisted of performing quantitative watershed and water-quality analyses for each site, as well as qualitative assessments of stream health and condition using a set of criteria and best professional judgment (BPJ). Each quantitative analysis or BPJ criterion evaluated some aspect of stream or watershed condition that could potentially impact water quality and aquatic life. Sixteen BPJ criteria (e.g., bank erosion, sediment deposition, grazing impacts) were tailored for cold-water streams (mountainous regions), and were slightly different from thirteen BPJ criteria tailored for warm-water streams (prairie regions). A series of seven tests, or "screens", was then used to create the final list of reference sites. The screens were constructed from the qualitative BPJ assessments and also from numeric values identified as impact thresholds in the quantitative analyses, and addressed factors operating at the watershed-scale, site-specific scale and, in many cases, both. The seven screening tests were: cumulative impacts from multiple causes; site-specific data sufficiency; impacts from land-use based on the proportion of agriculture; numeric water-quality standards exceedences for heavy metals; impacts from mines; road density; and timber-harvest intensity (the later two applicable to cold-water streams only). To make the final list, a site had to pass each applicable screen. Sites that passed all applicable screens can be considered general-purpose reference sites, since they were found to be in an unimpacted condition for all categories.

Using the process described above, a group of Montana reference stream sites has been identified. However, there remains the need to assure that the reference sites are sufficiently similar to the stream sites against which they are compared. In general, Omernik level-III ecoregions have shown themselves to be an excellent tool for the initial partitioning of Montana reference streams. However, in certain cases more specific geospatial characteristics than level III ecoregions alone may need to be determined for the reference site and the comparison site. What those geospatial characteristics will be varies according to the parameter of interest. For example, elevation is important when considering aquatic insect (macroinvertebrate) populations, watershed area is important when considering prairie stream fish populations, and nutrient concentrations are best explained by level IV (fine-scale) ecoregions. It is likely that some water quality parameters and biological assessment metrics can be “referenced” at a fairly coarse scale (e.g. level III ecoregions), while others cannot. The reader should refer to specific reports (many cited in this report) and their associated stream assessment “tools” to decide how to best apply the reference sites provided here. And there are limitations to the use of the reference stream data. Most of the sites are located in lower Strahler stream orders — mainly 1st through 4th but including a few 5th order sites — and the data are most applicable to streams of that size range (the so-called “wadeable” streams). Therefore, the extension of these data to sites from much larger waterbodies (e.g., Yellowstone River, 6th order) should be undertaken with caution.

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SECTION 1.0
INTRODUCTION AND RATIONALE FOR STUDYING REFERENCE SITES

1.1 Introduction

This document describes work undertaken by the Montana Department of Environmental Quality (DEQ) to identify and assess reference sites for flowing waters of the state. The need to identify reference stream sites is an outgrowth of the reference condition concept. The reference condition concept asserts that there exist for any group of waterbodies relatively undisturbed examples that can represent the natural biological, physical and chemical integrity of a region (Hughes et al. 1986; Barbour et al. 1996; Gibson et al. 1996); therefore, reference stream sites are those that can represent the reference condition. The work detailed in this report was carried out from 2000 to 2005, and is a continuation and expansion of the DEQ's earlier reference stream work described in the 1992 report, "Benchmark Biology of Montana Reference Streams" by L. Bahls and others. Unlike the Bahls et al. (1992) report, this report does not detail the physical, chemical and biological characteristics of MT reference sites. The main purpose of this report is to propose a process for consistently identifying reference stream sites, including specific techniques that can be used to assess the quality of each reference site. We also describe the fieldwork undertaken as part of the DEQ Reference Stream Project, the effort to collate reference data from other agencies working in the state and the final development of the reference-site list.

1.2 Rationale for Studying Reference Stream Sites, and Definitions

The DEQ needs to identify reference sites because they help the Department interpret water quality standards. Water quality standards are expressed in either numeric or narrative forms. Numeric standards are specific values not to be exceeded, for example the MT human health standard for copper which is 1.3 mg/L (DEQ 2004a). Narrative standards, on the other hand, describe in a concise way a water quality condition that must be maintained and do not have specific numbers associated with them. These types of standards are often referred to as the "free from" standards, since many of them are worded to include that phrase (e.g., ARM[1] 17.30.637, "State surface waters must be free from substances attributable to municipal, industrial, agricultural practices…that will…create floating debris, scum, a visible oil film…").

A number of Montana's narrative standards specifically require that water quality be compared to "naturally occurring" conditions. The state of Montana has defined naturally occurring as "conditions or materials present from runoff or percolation over which man has no control or from developed land where all reasonable land, soil and water conservations practices have been applied" (ARM 17.30.602[19]). The Administrative Rules of Montana (ARM) then define reasonable land, soil and water

[1] Administrative Rules of Montana.

conservation practices as activities that, in essence, completely protect all beneficial water uses (see ARM 17.30.602[24]). The core idea here is that man's activities in a watershed are an integral component of the landscape, as long as those activities do not negatively impact the various uses of the water (drinking, recreation, fisheries, etc.). Reference sites, therefore, are used to characterize naturally occurring conditions and reflect a group of waterbodys' greatest potential for water quality. (See also the reference condition definition in Appendix A of DEQ [2004b].)

The DEQ has taken this concept a step further by drafting definitions for two levels (tiers) that meet the state's naturally occurring definition, but which more specifically describe the gradient of conditions that may be expected. The development of these definitions is part of a larger effort underway nationally to better define criteria to protect aquatic-life uses (U.S. EPA 2005a). The definitions for each tier are as follows:

> *Tier 1 — Natural Condition***:** The characteristics of a waterbody that is unaltered from its natural state, or there are no detectable human-caused changes in the completeness of the structure and function of the biotic community and the associated physical, chemical, and habitat conditions. All numeric water quality standards must be met and all beneficial uses must be fully supported unless impacts are clearly linked to a natural source. The natural condition is the highest attainable biological, chemical, physical, and riparian condition for waterbodies.

> *Tier 2 — Minimally Impacted Condition*: The characteristics of a waterbody in which the activities of man have made small changes that do not affect the completeness of the biotic community structure and function and the associated physical, chemical, and habitat conditions, and all numeric water quality standards are met and all beneficial uses are fully supported unless measured impacts are clearly linked to a natural source. Minimally impacted conditions can be used to describe attainable biological, chemical, physical, and riparian habitat conditions for waterbodies with similar watershed characteristics within similar geographic regions and represent the water body's best potential condition.

Waterbodies that meet the conditions described in either of the two definitions above may be used as reference sites, since both definitions fall under the broader definition of naturally occurring found in the ARM.

Provided below is the definition for the next tier in the series, tier 3. Tier 3 describes waterbodies that have a degree of impairment sufficient to generally warrant listing on the state's 303(d) list of impaired waterbodies (i.e., DEQ 2004b). The tier-3 definition will become important later in this report in relation to the assessment of candidate reference sites. There are two further definitions for waterbodies that have increasingly severe levels of impairment (tiers 4 and 5), however waterbodies of this nature are outside the context of this report and therefore their definitions have not been provided.

Tier 3 — Moderately Impaired Condition: The characteristics of a waterbody in which the activities of man have made obvious changes to the completeness of the biotic community structure and function and the associated physical, chemical, and riparian habitat conditions, but the impacts have not caused a major displacement of sensitive taxa and acute numeric water quality standards are not exceeded.

SECTION 2.0
IDENTIFICATION & SAMPLING OF REFERENCE SITES

2.1 The Montana DEQ Reference Stream Project

After the initial work by Bahls et al. (1992) there was some follow-up work in the mid-1990's at the original sites, but no effort was made to locate new regional stream reference sites. With additional funding made available in 2000, the DEQ reinitiated a wadeable stream reference-site project. The objective of the project is to locate and characterize new reference-stream sites around the state, and to perform periodic follow-up visits to sites originally examined by Bahls et al. (1992). In 2000 and 2001 the field-work was undertaken statewide in coordination with the U.S. EPA's Environmental Monitoring and Assessment Program (EMAP; Lazorchak et al. 1998; U.S. EPA 1998), since the same field crews performed the work. Beginning in 2003, the project was undertaken in coordination with the University of Montana by field crews exclusively focused on reference-site monitoring. Sampling sites were located statewide in 2003. In 2004, there was an emphasis on foothill and valley streams of southwest Montana. In 2005, sampling focused on foothill and valley streams of both southwest and southeast Montana. Table 1.0 below shows the candidate reference stream sites that were sampled in each year of the project.

DEQ has successfully relied upon intensive field reconnaissance and best professional judgment (BPJ) to locate reference watersheds and sites. Crews examined overall watershed conditions during driving reconnaissance tours, and when a watershed was deemed to have minimal human impacts specific stream sampling sites were selected. Preliminary Geographic Information System (GIS) work examining watershed logging intensity, intensity of agricultural use and presence of abandoned mines has in recent years been used to pre-screen potential sites and watersheds. Reference sites were normally visited three times during each field season (roughly June through September). Stream sites were assessed in short reaches using the Western Pilot EMAP physical habitat characterization protocols (Lazorchak et al. 1998). Reach lengths were established as 40x the wetted width measured at the initial visit, or a minimum of 150 m. Each reach was divided into 10 equally spaced subreaches, which provide a total of 11 transects perpendicular to stream flow along the entire reach. In addition to EMAP habitat characterization, all sites were assessed using standardized DEQ or NRCS stream reach assessment forms (e.g., Pick et al. 2004; DEQ 2005a). These forms document human impacts to the streams, overall riparian condition and geomorphic stability. Geomorphic classification following Rosgen (1996) was also determined. Water quality samples such as sediment and water-column metals concentrations, nutrients, and common ions were collected from each site. Biological sampling has varied somewhat from year to year, but typically involved sampling for diatom and macroinvertebrate populations (DEQ 2004c, DEQ 2005b), as well as biomass (measured as chlorophyll *a* and ash free dry weight) of both benthic and water-column algae. A subset of sites has been sampled over multiple years to better understand the year-to-year variation of the sites.

Table 1.0. Stream sites identified as candidate reference sites and sampled for the DEQ Reference Stream Project. Sites are organized by year sampled.

Year Sampled‡											
2000			2001			2003			2004		
Site Name	Lat (DMS)	Long (DMS)	Site Name	Lat (DMS)	Long (DMS)	Site Name	Lat (DMS)	Long (DMS)	Name	Lat (DMS)	Long (DMS)
W FK Poplar R†	483354	1053607	Wisconsin Cr*	453548	1132025	Blacktail Deer Cr	450019	1122642	Rock Cr 2†	483509	1065953
Larb Cr†	480936	1071707	Waldron Cr*	475512	1125002	Wisconsin Cr*	453548	1132025	Rock Cr (BLM land)	483925	1070220
Rock Cr 1*	485233	1065348	N FK Teton River*	475801	1124840	Clear Cr	481822	1092926	Bitter Cr	483856	1065409
O'Fallon Cr 1†	460608	1044518	Blackleaf Cr	480047	1124135	E Rosebud Cr	451336	1093621	Porcupine Cr	481229	1062253
Spring Creek	460811	1044001	Cow Cr	475140	1085748	Seeley Cr	450553	1091757	Wolf Cr at Wolf Pt.	480512	1054037
Muddy Cr	481220	1094534	Battle Cr†	485310	1092326	O'Fallon Cr 1†	460608	1044518	WF Poplar River†	484150	1054955
			Stony Cr*	461833	1134009	O'Fallon Cr 2†	462816	1044611	Pole Cr	452119	1131050
			Seymour Cr*	455945	1131114	W FK Poplar River†	484149	1054955	Willow Cr (I)	452653	1124940
			E FK Bull River*	480730	1154339	Stony Cr*	461833	1134009	Willow Cr (II)	452617	1124432
			Calf Cr*	465042	1105736	Calf Cr*	465042	1105736	Blacktail Deer Cr	450019	1122642
			Spring Park Creek	465551	1105214	Tenderfoot Cr†	465525	1105348	Elk Springs Cr	443840	1113949
			Tenderfoot Cr†	465525	1105348	Roaring Lion Cr *	461134	1141436	Cottonwood Cr	445633	1122546
			Blacktail Deer Cr	450019	1122643	Wolf Cr at Wolf Point	480516	1054041			
			Seeley Cr	450553	1091758	E FK Bull River*	480730	1154339			
			Wyoming Cr	450313	1092425	N FK Teton River*	475801	1124827			
			Elk Cr†	453544	1112320	Rock Cr 1*	485233	1065348			
			Elk Springs Cr	443840	1113949	Rock Cr 2†	483525	1070004			
			Frenchman Cr	485452	1071832						
			Redwater River†	480127	1051452						
			O'Fallon Cr 2†	462818	1044611						
			Little Powder R	452015	1041840						
			Little Powder R	451908	1051904						
			Fish Cr	461502	1094608						
			M FK Beaver Cr	465724	1093257						

*Site originally sampled by Bahls et al. (1992).

† Same stream sampled by Bahls et al. (1992) but in a different location.

‡ No reference sites were sampled in 2002.

Table 1.0, Cont. Candidate sites sampled in 2005 for the DEQ Reference Stream Project.

Year Sampled		
2005		
Name	Lat (DMS)	Long (DMS)
Willow Cr (I)	45 26 53	112 49 40
Cherry Creek	45 35 27	112 45 59
Willow Cr (II)	45 26 17	112 44 32
Cottonwood Cr	44 56 33	112 25 46
EF Blacktail Deer Creek	44 51 57	112 13 07
Sarpy Creek (#2)	46 05 54	107 04 09
Sunday Creek	46 27 20	105 52 29
Pumpkin Creek	46 11 18	105 37 18
Custer Creek	46 42 34	105 33 36
O'Fallon Creek (Site 2)[†]	46 28 16	104 46 11
Cedar Creek	46 47 29	104 33 27
Little Missouri River #1	44 59 43	104 25 25
Little Missouri River # 3.5	45 14 11	104 14 28
Box Elder Creek	45 50 42	104 08 37

[†] Same stream sampled by Bahls et al. (1992) but in a different location.

2.2 Collation of Other Reference Sites and Associated Data

Over the years, DEQ Water Quality Planning Bureau staff has been using data from a variety of least disturbed sites around Montana to interpret the state's narrative water quality standards. In March 2004 we began collating all data associated with stream sites that were considered "reference" by the Water Quality Planning Bureau, but which were external to the DEQ Reference Stream Project. Bureau staff was asked to provide sites and associated data that they believed met the following definition: "relatively undisturbed stream segments that can serve as examples of the natural biological, physical and chemical integrity of a region". Although in use by the DEQ, these data were not necessarily collected by it. The data were collected over a number of years by a variety of agencies, including the United State Geological Survey (USGS), the MT Department of Natural Resources and Conservation, The Bureau of Land Management (BLM), the U.S. Environmental Protection Agency (U.S. EPA), the University of Montana and others.

In addition to the DEQ Reference Stream Project sites and the sites collated from within the Bureau, a number of potential reference sites were identified in other projects and programs from around the state. These were:

I. *USGS Hydrologic Benchmark Network (HBN) stations for MT (1963-1995*). Three sites in Montana were identified by USGS as meeting the objectives of the HBN program. The HBN program sought to collect water quality data from basins minimally affected by human activities and which would serve as controls for separating natural from artificial changes in other streams (Alexander et al. 1996). The sites are: Swiftcurrent Creek at Many Glaciers, MT; Rock Creek below Horse Creek near International Boundary; and Beauvais Creek near St. Xavier, MT.

II. *Tri-State Water Quality Council (1998-2002).* Two sites were included (one on Rock Cr near Clinton and the other on the lower Blackfoot River) from this ongoing monitoring study in the Clark Fork River basin. The project is focused on nutrient sampling and the data are reported in a series of reports, one of the more recent being Land and Water Consulting Inc. (2003). McGuire (2001) also identified these two sites as being of the highest quality, and having excellent biological integrity, in a long-term study (1986 to present) of aquatic macroinvertebrates in the Clark Fork River basin.

III. *Western EMAP (2000-2004).* EMAP sites were sampled throughout the Western United States (including Montana) in a stratified random design developed by the U.S. EPA (U.S. EPA 1998). The DEQ and University of Montana provided the field sampling crews and logistics used to conduct this work in Montana. Twenty-five stream sites were identified during the course of sampling as potential reference sites, and these were added to the list of candidate sites.

IV. *Regional EMAP (1999-2001).* The regional EMAP project (REMAP) was a cooperative effort between the U.S. EPA and Montana State University. The project's objective was to develop indices of biological integrity (IBIs) for fish, macroinvertebrates and diatoms for eastern Montana prairie streams. In order to identify reference sites necessary to develop the IBI's, a series of evaluation criteria were developed and a total of eight reference reaches were identified (Bramblett et al. 2003). These eight sites were added to the list of candidate sites.

V. *Utah State University Science to Achieve Results (STAR) reference stream work, & data from the U.S. Forest Service PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program (PIBO; 2000-2004).* Dr. C. Hawkins of Utah State University (USU) is developing regional models (RIVPACS; Hawkins et al. 2000) for the western United States that are designed to predict stream condition based on aquatic insect populations. He collected nutrient, macroinvertebrate, periphyton and other data from more than 400 candidate reference sites around the western U.S., a number of which were located in Montana and some of which were part of the PIBO network of reference sites (Kershner et al. 2004; Hawkins 2005). He then applied a screening process analogous to ours that rated the sites as pristine, minimally disturbed or least disturbed. The forty-four sites that rated as pristine and minimally disturbed were considered as established reference sites in the present work. (A number of these sites overlapped with sites the DEQ had sampled for its Reference Stream Project.)

VI. *The Montana Natural Heritage Program (2001 to present).* The Natural Heritage Program is the state's source of information on the status and distribution of native animals and plants, emphasizing species of concern and high quality habitats. Fifty-three sites that the Natural Heritage Program identified as high quality were acquired and added to the candidate reference sites.

VII. *DEQ Fixed Station Monitoring Project (1999-present).* Three stream sites in this ongoing trends-analysis project were selected as candidate reference sites (lower Blackfoot River, Rock Cr nr Clinton, and the Middle Fork Flathead River nr West Glacier). Although sampled near some of the Tri-State Water Quality sites, the sampling locations were spatially separated and/or undertaken at different times.

VIII. *DEQ Nutrient Pilot Project (2001-2002).* DEQ and the University of Montana carried out a two-year study of prairie streams in Northeastern Montana with the objective of developing regional nutrient criteria (Suplee 2004). The study identified four stream sites meeting the general reference site definition given above. The study included monthly sampling of nutrients and algal biomass during the growing season (May to September), less frequent samplings for macroinvertebrates and periphyton, and detailed evaluations of stream habitat conditions.

SECTION 3.0
EVALUATION OF THE CANDIDATE REFERENCE SITES

3.1 General Considerations and Assumptions

After having assembled a list of candidate reference stream sites, the next step was to evaluate each potential reference site using a set of criteria. (Sites from the STAR project, the Suplee (2004) study, and sites from Bahls et al. (1992) that were rated as 'fully supporting all uses', did not go through the evaluation process described here. These sites had been extensively reviewed and were already considered to be established reference sites.) We assigned each site a unique number and used level III ecoregions (Omernik 1987; Woods et al. 2002) to categorize them as either warm water or cold water. Ecoregions are designed to be multi-purpose ecological zones in which the aggregate of all aquatic and terrestrial ecosystem characteristics of one zone differs from that of the other zones (Omernik and Bailey 1997). Stream sites located in the level III ecoregions Canadian Rockies (41), Northern Rockies (15), Idaho Batholith (16) and Middle Rockies (17) were labeled as cold water, and those in the Northwestern Glaciated Plains (42), Northwestern Great Plains (43) and the Wyoming Basin (18) were labeled as warm water (Woods et al. 2002). Cold-water streams are generally located in the western mountainous region of the state, and are expected to support salmonids — fish preferring temperatures lower than 65 ^{o} C. Warm-water streams are generally located east of the Rocky Mountain Front, and comprise prairie streams and rivers that support walleye, bullhead, bass and a variety of other fish that prefer temperatures 65 ^{o} C or greater (Holton and Johnson 1996). Overall, the geographic location of warm- and cold-water sites based on ecoregions closely parallels the state's beneficial use classifications for warm- and cold-water fisheries (see ARM 17.30.607).

There existed a number of stream sites that were of reference quality for some attributes (e.g., riparian condition, geomorphology) but failed in another important category, for example having excessive abandoned mine sites & elevated metals concentrations. It was our intent that sites of this description would **not** be included on the final list of reference sites, and that only those that passed all key criteria would be included. That is, to be considered a reference site using our approach a site needed to satisfy all evaluation categories reasonably well, and not possess any "fatal" flaws. (This general concept has elsewhere been referred to as the Anna Karenina principal [Diamond 1997].) It has been shown that both local-scale and watershed-scale human impacts play a role in affecting stream ecology (Snelder and Biggs 2002; King et al. 2005). How factors operating at these two scales interact is complex, and not fully understood. Therefore, one of our key assumptions was that local, on-stream impacts were equal in importance to upstream, watershed-scale impacts, an approach similar to that used by the Oregon Department of Environmental Quality (Drake 2003). Assessment of the local and watershed scale factors was undertaken using two approaches, one based on best professional judgment (BPJ) and the other based on quantitative watershed analyses. Each of these approaches is described below.

3.2 Best Professional Judgment

A series of evaluation criteria were selected based on DEQ and EPA staff expertise, and other state's examples (Table 2.0, 3.0). Each criterion assessed some aspect of stream or watershed condition that could potentially impact water quality and aquatic life. Slightly different criteria were used for cold-water streams (those in level III ecoregions 41, 15, 16 and 17) than for warm-water streams (those in level III ecoregions 42, 43 and 18).

Using available data, each candidate site was evaluated using the applicable criteria by DEQ or EPA staff. (This process is a very simplified version of the DEQ's sufficient credible data/ beneficial use-support assessment process (DEQ 2004b) that is used to develop the biennial 303(d) list of impaired state waters.) Criteria that addressed watershed level factors were evaluated by reviewing aerial photographs delineated at the 5^{th} code HUC level (Seaber et al. 1987). If the site had a high Strahler order (e.g., 5; Strahler 1964) then a larger basin was examined. A larger basin could be a 4^{th} code HUC, or an aggregation of 5^{th} code HUCs that — together— best defined a stream's watershed. For each site, each criterion was assigned a score of 1 (reference condition), 0 (stressed condition), or ND (insufficient data to assess). For example, if examination of aerial photos showed few or no roads in the upstream watershed, that criterion would receive a 1. Reviewers also recorded notes as to why a criterion was given a particular score. Finally, based on the totality of information reviewed, an overall condition rating for the site was made by indicating if the stream site was tier 1 (natural condition), tier 2 (minimally impacted) or tier 3 (non-reference; Tables 2.0, 3.0). A stream site could have been rated as tier 4 or 5, however only sites of fairly high quality made it to the candidate list to begin with, and we did not identify any sites that rated 4 or 5. Reviewers had the discretion to decide which data were most important in determining a site's tier level. It was important that reviewers recorded their assessment notes with sufficient detail that a second reviewer could understand why a particular tier rating was made.

It should be noted that locating stream sites that rigidly fit the tier 1 definition on page 2 that there be "no detectable human-caused changes in the completeness of the structure and function of the biotic community…" may be very difficult to achieve, even in some wilderness areas, given the degree to which non-native salmonids were actively stocked in the 19^{th} and 20^{th} century in mountainous areas of Montana and the West (Hanzel 1959; Brown 1971; Moyle et al. 1976; Liknes and Gould 1987). Streams that received a tier 1 rating in this report, therefore, should be viewed as having the absolute minimal human influences observable, but could still contain some non-native species.

Table 2.0. Criteria used to evaluated reference and stressed conditions for cold water streams. Example evaluation conclusions for each criterion are shown in the 'Reference Condition' column. In this example, the large number of zeros resulted in a tier 3 (non-reference) rating.

Parameter Evaluated	Reference Criteria (1)	Stressed Criteria (0)	Reference Condition Score
Physical Habitat Category			1=Y; 0=N; ND= No Data
Road densities (secondary data)	Minimal number of roads and roads are not close to streams.	> 4 miles/ sq. mile	0
New and old timber harvests (secondary data)	Minimal harvest activities, outside of the riparian areas; timber management activities attempt to mimic a natural fire regime	Extensive harvesting within watershed (>25%); harvests occurring within riparian area.	0
Percent surface fines	Low fines, representative of the geologic conditions.	>30% fines less than 2mm.	ND
Sediment Deposition	Between 0-25% of the substrate surrounded by fine sediment (RBP language).	Greater than 75% of the substrate surrounded by fine sediment (RBP language).	0
Bank Erosion	No erosion or limited to “natural” occurrences. Stable banks.	Extensive bank erosion caused by anthropogenic activities.	0
Bank Vegetation	Over 90% of the streambank covered by stabilizing vegetation; vegetated zone width > 100 feet.	Less than 50% of the streambank covered by stabilizing vegetation; vegetated zone width < 10 feet.	0
Permitted point sources	Few to no point source discharges in the watershed. Site located greater than 5miles downstream or above the permitted discharge.	Many point sources discharges present. Site located less than 2 miles downstream of a point source discharge.	1
Land under agricultural use	Minimal to no agricultural use occurring.	Extensive agricultural activities present and may occur within the riparian area.	1
Grazing Use	Light grazing occurs; impacts are minimal.	Heavy grazing causing moderate impacts.	ND
Mining sites	Site not located in DEQ priority abandoned hardrock mining subbasin; or, basin mine density low.	Site located in DEQ priority abandoned hardrock mining subbasin; or, basin mine density high.	1
Professional Judgment Category			
Anecdotal evidence from non-standard sources	No anecdotal evidence of significant disturbance encountered.	Evidence of significant recent or persistent physical or chemical disturbance is credible and verifiable.	ND
Field observations not listed as criteria	No source of stress or evidence of existing stress exists and is not considered in other criteria.	A source of stress or evidence of existing stress exists and is not considered in other criteria.	0
Aesthetics	Site has exceptional aesthetic quality without apparent disturbances in watershed, riparian areas, or channel.	Aesthetically unappealing due to elements that probably affect water resource quality (must be described).	0
Other Determinations	Other agency/entity has determined that the site is of reference quality using acceptable documented procedures or non-biological criteria.	Other agency/entity has determined that the site is stressed using documented procedures or non-biological criteria	ND
Previous Investigations and Regulatory Involvement Secondary data http://nris.state.mt.us/interactive.html			ND
On 2004 303(d) list (or re-assessment list)?			***NO***
Probable Tier Level (1, 2, 3, 4, 5)			**3**

Table 3.0. Criteria used to evaluated reference and stressed conditions for warm water streams. Example evaluation conclusions for each criterion are shown in the 'Reference Condition' column. In this example, the large number of ones resulted in a tier 2 (reference) rating.

Parameter Evaluated	Reference Criteria (1)	Stressed Criteria (0)	Reference Condition Score
Physical Habitat Category			1=Y; 0=N; ND= No Data
Road densities (secondary data)	Minimal number of roads and roads are not close to streams.	> 4 miles/ sq. mile	1
New and old timber harvests (secondary data)	Minimal harvest activities, outside of the riparian areas; timber management activities attempt to mimic a natural fire regime	Extensive harvesting within watershed (>25%); harvests occurring within riparian area.	1
Permitted point sources	Few to no point source discharges in the watershed. Site located greater than 5miles downstream or above the permitted discharge.	Many point sources discharges present. Site located less than 2 miles downstream of a point source discharge.	1
Land under agricultural use	Minimal to no agricultural use occurring.	Extensive agricultural activities present and may occur within the riparian area.	1
Grazing Use	Light grazing occurs; impacts are minimal.	Heavy grazing causing moderate impacts.	ND
Mining sites	Site not located in DEQ priority abandoned hardrock mining subbasin; or, basin mine density low.	Site located in DEQ priority abandoned hardrock mining subbasin; or, basin mine density high.	1
Oil and Gas Wells	Absence of oil and gas development in the watershed.	Presence of oil and gas development above the site.	1
Professional Judgment Category			
Anecdotal evidence from non-standard sources	No anecdotal evidence of significant disturbance encountered.	Evidence of significant recent or persistent physical or chemical disturbance is credible and verifiable.	ND
Field observations not listed as criteria	No source of stress or evidence of existing stress exists and is not considered in other criteria.	A source of stress or evidence of existing stress exists and is not considered in other criteria.	ND
Aesthetics	Site has exceptional aesthetic quality without apparent disturbances in watershed, riparian areas, or channel.	Aesthetically unappealing due to elements that probably affect water resource quality (must be described).	1
Other Determinations	Other agency/entity has determined that the site is of reference quality using acceptable documented procedures or non-biological criteria.	Other agency/entity has determined that the site is stressed using documented procedures or non-biological criteria	ND
Previous Investigations and Regulatory Involvement Secondary data http://nris.state.mt.us/interactive.html			1
On 2004 303(d) list (or re-assessment list)?			***NO***
Probable Tier Level (1, 2, 3, 4, 5)			**2**

3.3 Quantitative Watershed-level and Site-specific Analyses

Two quantitative watershed-level analyses and one quantitative site-specific analysis were undertaken for each candidate reference site. How these data were used in the assessment of candidate reference sites will be further detailed in the next section.

The first watershed analysis determined the proportional area of different land-cover types using the MT Gap Analysis Program (GAP) GIS layer (Fig. 1.0; Fisher et al. 1998). We were most interested in the proportion of agricultural land use in each basin. The area delineated was the area within the 5th code HUC upstream of the reference site.

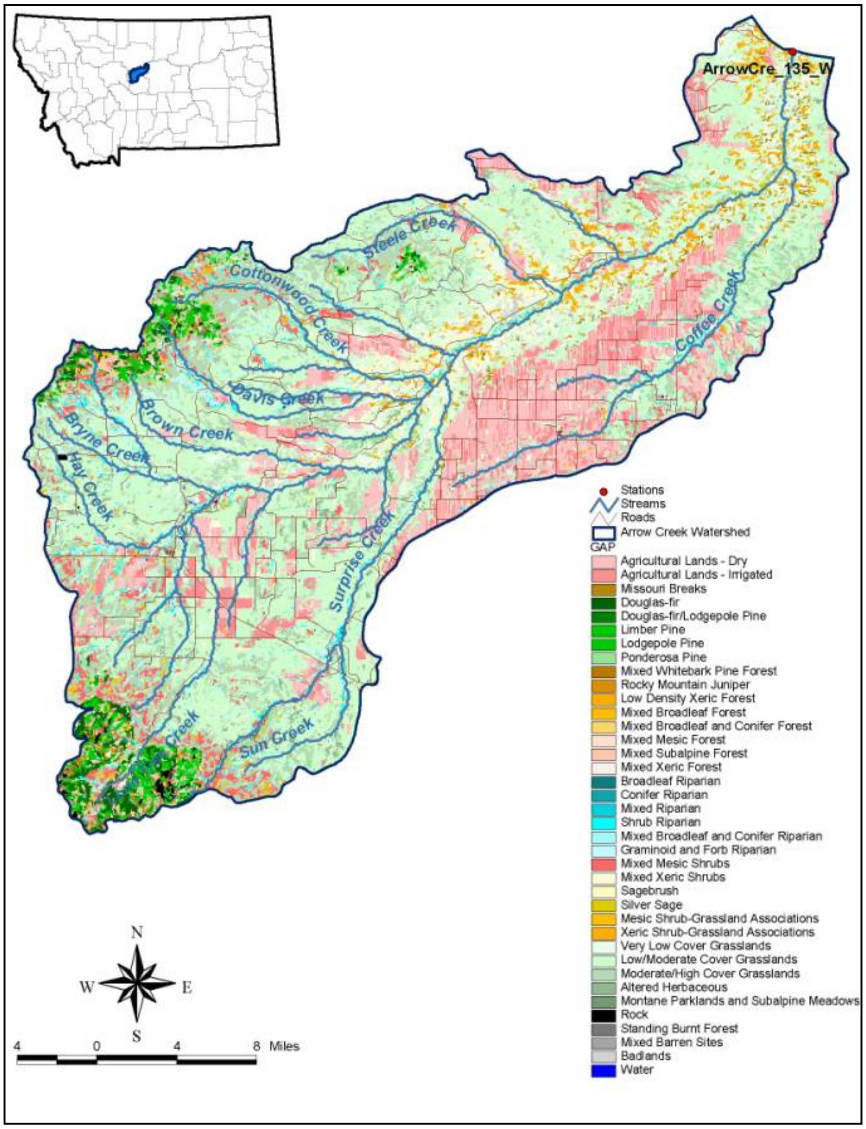

Fig. 1.0. Example quantitative watershed land-type analysis. The basin is delineated at the 5th code HUC level. The area of each GAP land-type was calculated for the watershed upstream of the reference site. Map provided courtesy of Tetra Tech Inc.

The second watershed analysis determined the density of roads (miles/mile^2) in the 5th code HUC upstream of each site. Road density of reference site watersheds was based on the U.S. Census Bureau's 2000 Topologically Integrated Geographic Encoding and Referencing system (TIGER) road-density atlas (1:100,000 scale; U.S. Census Bureau 2005). Road density was delineated for individual road types (e.g., secondary and connecting roads, vehicular trails), as well as total roads.

Where available, metals and hardness water-quality data were located, primarily from EPA's modernized and Legacy STORET databases. The chronic and acute aquatic life standards (DEQ 2004a) for individual sites were then calculated using these data. These site-specific analyses examined whether or not heavy metals (cadmium, copper, lead, zinc, mercury and dissolved aluminum) exceeded Montana's numeric standards.

3.4 Final Screening Process Applied to Candidate Reference Sites

Final selection of the reference sites was achieved by passing each site through a series of tests, or "screens" (Fig. 2.0). The screens addressed impacts at the watershed-scale, the site-specific scale and, in many cases, both. Screens were constructed from both the BPJ assessment criteria and results from the quantitative analyses outlined in Section 3.3. For example, the timber harvest screen (screen 4) is based on the BPJ criteria "New and Old Timber Harvests" (Table 2.0). During the BPJ evaluation the assessor considered not only overall intensity of timber harvest in the watershed, but also field notes regarding localized timber harvest activities that may have locally impacted the reference site.

Sites that passed all of the screens were included on the final reference site list. Each screen is discussed in detail below.

Screen 1: Probable Tier Level: Any site that was determined to be tier 3 (or worse) in the BPJ assessments was removed at this step. A tier 3 rating generally reflected cumulative impacts from multiple causes (i.e., too many zero ratings in the criteria; see example in Table 2.0). The corollary to this is that a zero rating in a single criterion (e.g. "Bank Vegetation") may not necessarily have warranted a tier 3 rating. Zero ratings could occur due to problems at the watershed level, site-specific level, or a combination of both. As discussed in Section 3.2, each reviewer had discretion to make the final tier-level determinations.

Screen 2: Local Level Screen for Sufficiency: The purpose of this screen was to remove sites that had insufficient site-specific data. Although watershed-scale data were generated by GIS methods for all sites, on-site data collected in the field were considered critical. Therefore, candidate reference sites lacking on-site data were removed at the step. (Note: these sites should be targeted for future data collection.)

> *For Warm Water Streams:* If sufficient data existed to assess $\geq$ 3 of 7 BPJ categories (i.e., the categories received a 0 or 1), the waterbody was passed to the next screen. The seven BPJ categories were: Grazing Use; Aesthetics; Field Observations not Listed as Criteria; Other Determinations; Previous

Investigations and Regulatory Involvement Secondary Data; Anecdotal Evidence from Non-standard Sources; and Land under Agricultural Use (Table 3.0).

<u>*For Cold Water Streams:*</u> If sufficient data existed to assess $\geq$ 4 of 11 BPJ categories (i.e., the categories received a 0 or 1), the waterbody passed to the next screen. The eleven BPJ categories were: Grazing Use; Aesthetics; Field Observations not Listed as Criteria; Other Determinations; Previous Investigations and Regulatory Involvement Secondary Data; Anecdotal Evidence from Non-standard Sources; Percent Surface Fines; Sediment Deposition; Bank Erosion; Bank Vegetation; and Land Under Agricultural Use (Table 2.0).

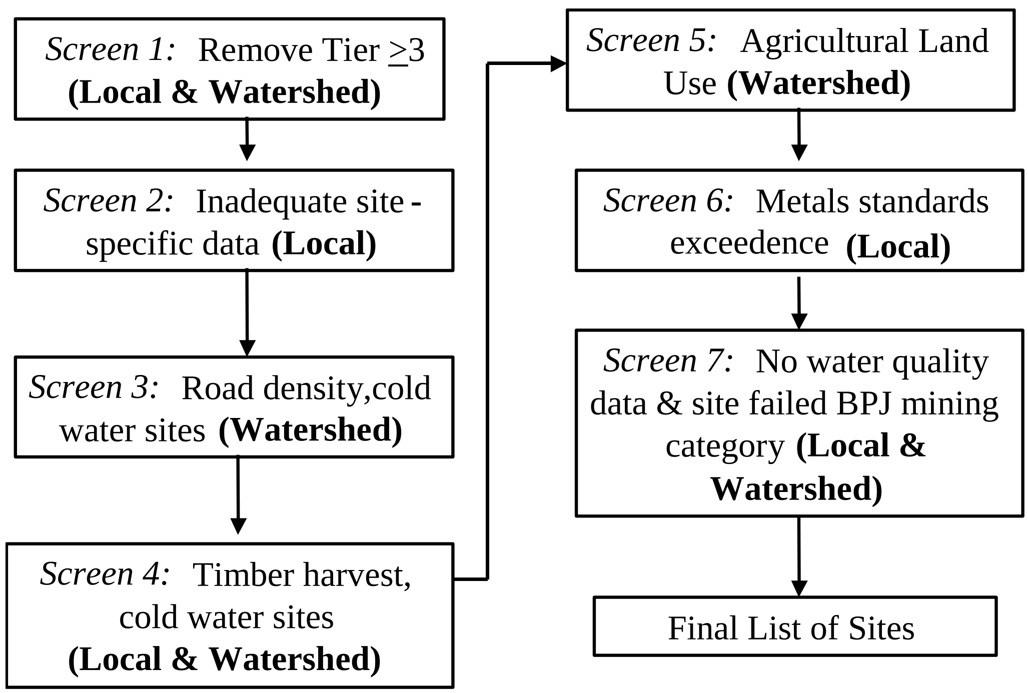

Figure 2.0. Diagram showing the screening steps reference sites were passed through. Each box indicates whether the screen operated at a watershed scale, site-specific scale, or both. A site that failed any one of the applicable screening steps was not included among the final reference sites.

Screen 3: Road Density Screen (*applicable to cold water streams only*): Cold-water streams (as defined in the present work) are found in MT level III ecoregions 41, 15, 16 and 17, ecoregions that are predominantly forested (Woods et al. 2002). Studies show that logging roads produce by far the largest proportion of sediment among forest management practices (see literature review by Waters [1995]), and for this reason this screen was applied only to the cold water streams. The TIGER road density atlas does not map small, unimproved roads (i.e., logging roads) to the degree we expected (Boschen, personal communication). Due to the incompleteness of the TIGER road coverage, it was unclear if we could use previously published road-density recommendations for forested areas (e.g., USDA 1995; USFWS 1998). We opted to build a road-density index for this screen, based on the BPJ assessments of the candidate cold-water reference sites. The TIGER road densities for cold water streams that received a 0 on the BPJ assessment criterion "Road Densities" (Table 1.0) were placed in one group and those that received a 1 were placed in another. The mean road density in the group receiving 0 was significantly higher than the mean for the other group ($p = 0.03$; T-test for means, unequal variance). We selected as a threshold the 90th percentile of total road density for the stream group that received 1's, equal to 1.19 mi/mile^2. This value, whose metric equivalent is 0.74 km/km^2, is fairly close to the road density threshold (< 0.5 km/km^2) used by the U.S. Forest Service for defining reference watersheds in forested lands of the West (Kershner et al. 2004). Cold-water sites having a road density greater than 1.19 mi/mile^2 (0.74 km/km^2) were removed at this step.

Screen 4: Timber Harvest Screen *(applicable to cold water streams only)***:** We applied this screen only to cold water streams, since this land-use activity is applicable almost exclusively to the level III ecoregions 41, 15, 16 and 17 (Woods et al. 2002). We reviewed the assessment notes for each candidate reference site that received a 0 in the BPJ assessment criterion "New and Old Timber Harvests". Depending upon the intensity of harvest in the watershed (based on the BPJ of the assessor and team consensus after reviewing the orthoquads), the site was either removed at this step or allowed to move to the next.

Screen 5: Agricultural Land Use Screen: Studies show that the percent of agricultural land use in a basin has an effect on water quality, for example increasing nutrient concentrations (Miller et al. 2005; King et al. 2005). Negative impacts to aquatic life may occur when approximately 30 to 60% of a basin's land area is in agricultural use (Sheeder and Evans 2004 [2]; Zheng et al. 2005). This screen was designed to reflect this range. Percent agricultural land use in each basin was determined from the 5th code HUC GAP analyses described above. The percent of lands in the GAP system classified as "Agricultural Land-Dry" (No. 2010) and "Agricultural Land-Irrigated" (No. 2020) was summed to determine the total agricultural land use in the basin. (These two land-use types do not include lands used exclusively for cattle grazing. Cattle grazing could only be assessed at the site-specific level via the BPJ criterion "Grazing Use", and would have been addressed in Screen 1.) The screen was then run as follows:

a) Sites in basins having $\geq$ 51% agricultural land-use were removed.

b) For the remaining sites, those streams that received a 0 in the BPJ criterion "Land under agricultural use" were removed.

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[2] Sheeder and Evans (2004) report that unimpaired watersheds have 18.1% agricultural land cover, while impaired watersheds have 46.6%. However, the authors do not indicate at what proportion of agricultural use the transition from unimpaired to impaired occurs. Therefore, in this report we approximated the impact threshold as the percent agricultural land use (32%) midpoint between the unimpaired and impaired designations reported by Sheeder and Evans (2004).

c) The remaining waterbodies with agricultural land use between 30-50% were flagged. These were then hand-reviewed by staff and, based on the notes recorded by the assessor and a review of the orthoquads by the team, sites were removed at this step or allowed to move to the next. In making these decisions, we considered the proximity of agricultural land to the stream site and to the stream channel upstream of the site.

Screen 6: Screen for Numeric Water Quality Standards Exceedences: Sites were screened for exceedences of the MT acute and chronic aquatic life water-quality standards for cadmium, copper, lead, zinc, mercury and dissolved aluminum (DEQ 2004a). These metal were selected as they are among the most common heavy metals found in streams contaminated by hardrock mining, and have been extensively sampled throughout the mined regions of the state. Hardness and metals data were matched by date of collection, and for each site the number of cases was recorded where a metal concentration did (or did not) exceed the standard. Sites that showed water quality exceedences were then hand screened. In those cases where an isolated exceedence could have been the result of poor data quality (e.g., metals data from before 1980 often had very high detection limits) or some other one-time, short term reason, the site was flagged but allowed to move to the last step. Sites that showed a clear tendency towards water quality exceedences were removed at this step.

Screen 7: Screen for Abandoned Mine Sites: This screen applied to sites flagged in Screen 6 and also to sites lacking metals water-quality data. If a site received a 0 in the BPJ criterion “Mining Sites”, the site was hand-reviewed by staff and, based on the notes recorded by the assessor and the team consensus, was either removed at this step or allowed to move to the end. In general, a ‘guilty until proven innocent’ approach was taken. For example, sites were removed that had substantial (based on BPJ) mine density in their watersheds but did not have water quality or sediment data that could be used to assess actual standards exceedences.

The sites that passed through all seven screens were considered reference sites, and are mapped in Fig. 3.0 and listed in Appendix A.

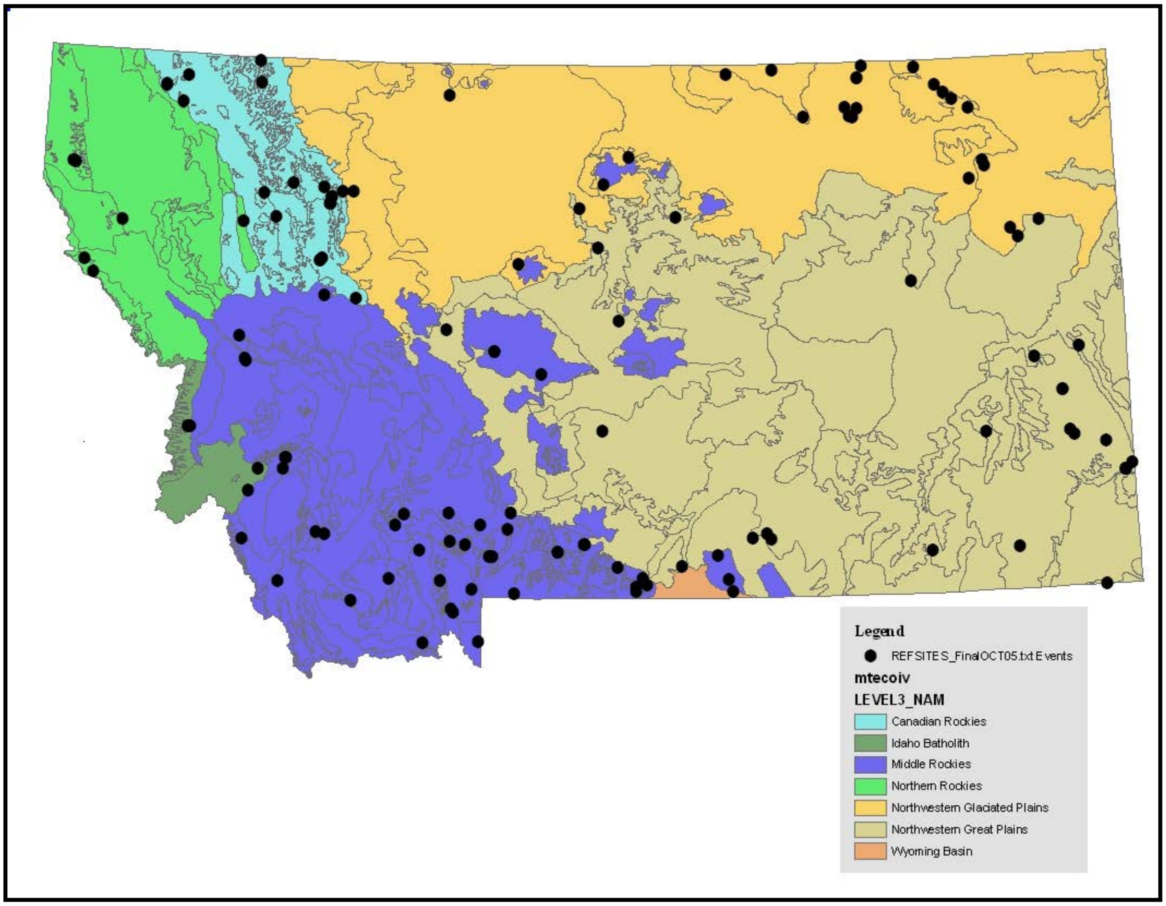

Fig. 3.0. Screened DEQ reference sites identified in this project. Colored background areas show the Omernik Level III ecoregions.

SECTION 4.0
DISCUSSION AND CONCLUSIONS

4.1 Application of the Reference Sites

The reference sites shown in Fig. 3.0 and listed in Appendix A may be considered "general purpose" reference stream sites that can be used in regional analyses and site-specific applications. This list of reference sites has been evaluated using a comprehensive list of assessment categories (riparian condition, mining impacts, geomorphology, point sources, etc.), and because the sites were found to be in an unimpacted condition for all important categories, data collected from the sites may be considered of reference quality. However, there remains the need to assure that the reference sites are sufficiently similar to the sites against which they are being compared.

The process of selecting appropriate reference sites for the purpose of making comparisons of water quality and other parameters has already been outlined as DEQ's "primary" methodology, found in Appendix A of DEQ 2004b. For example, the DEQ's periphyton biocriteria procedural guidelines (Bahls 1993) describe a two-part, sub-regionalization process for making comparisons to reference sites. Protocol I is a coarse-scale geographic approach, in which stream impairment assessments are made by comparing a particular site's diatom assemblage to diatom data from reference streams aggregated into two groups of level III ecoregions ('mountain' ecoregions and 'prairie' ecoregions). Protocol II uses a nearby reference site with similar Strahler stream order for more sensitive comparisons (Bahls 1993), and is supposed to be used when greater confidence in the analysis is required.

In general, Omernik ecoregions (Woods et al. 2002) have shown themselves to be excellent tools for (at least) the initial grouping of Montana reference streams. For example, recent work shows that level III ecoregions explain the broadest patterns of fish and aquatic macroinvertebrate community diversity found across Montana (Stagliano 2005). Similarly, statistical analyses by Varghese and Cleland (2005) indicate that ecoregions —both level III and IV— are superior to two other geospatial classifications (MT lithology and Strahler stream order) for the purpose of segregating Montana stream nutrient concentrations.

In certain cases, more specific geospatial characteristics than just ecoregions alone may need to be determined for the reference site and the stream site against which it is compared. Example geospatial characteristics include: elevation (important to the distribution of aquatic macroinvertebrates and fish; Hughes et al. 1987; Bollman 1998); soil-type and stream order (Snelder and Biggs 2002); and stream morphology, including the stream classification system of Rosgen (1996) and alternative classification approaches such as that of Montgomery and Buffington (1993). Snelder and Biggs (2002) propose a river/stream classification system for New Zealand that places stream-controlling factors into the following hierarchical order of importance (listed from most to least important): climate; source of flow; geology; land cover (i.e., vegetative cover in the watershed); stream order; and valley landform (slope, gradient, etc). Since ecoregions essentially integrate the first four of the six factors listed (Omernik 1987), stream order and valley landform can be considered supplemental geospatial characteristics that might be used to further enhance any comparison to reference. Closer to home, Suplee (2004) uses stream entrenchment ratio and characteristics of the riparian area to create two prairie-stream groups that best explain variation in northeastern Montana's aquatic plant communities. Sub-stratification was needed in spite of the fact that the prairie streams are wholly contained within a single level III ecoregion (the Northwestern Glaciated Plains). Reference sites sharing geospatial characteristics with prairie streams in a group could then be used to represent the expected condition of the group. Similarly, Bramblett et al. (2005) report that watershed area was a key factor in the development of an index of biological integrity (IBI) for Montana prairie fishes, since fish species richness increases with watershed area. Their fish IBI is applicable to prairie streams found in two Montana level III ecoregions, the Northwestern Glaciated Plains and Northwestern Great Plains.

It is likely that some water quality parameters and biological assessment metrics can be "referenced" at a fairly coarse scale, while others cannot. Ongoing work in Montana on nutrient & sediment criteria development, biocriteria development and other related efforts, some of which have been cited here, is providing indications as to which geospatial factors will achieve the most useful reference comparisons. Table 4.0 below highlights some of these important geospatial factors, and the parameters to which they apply. The reader should refer to specific reports and their associated stream assessment "tools" to decide how to best apply the reference sites listed in Appendix A.

Table 4.0. Montana studies that use geospatial factors to best apply reference stream data to various parameters.

Parameter	Geospatial Factors	Citation
Periphyton	Grouped Omernik level III ecoregions: mountains; prairies.	Bahls (1993); Teply and Bahls (*Draft;* 2005)
Nutrient concentrations	Omernik ecoregions: level III (coarse-scale); level IV (fine-scale).	Varghese and Cleland (2005)
Macroinvertebrates	Columbia River Basin; Canadian Rockies (level III ecoregion); elevation; watershed area	Jessup et al. (*Draft*; 2005)
Prairie fish IBI	Northwestern Glaciated Plains; Northwestern Great Plains; watershed area.	Bramblett et al. (2005)
Filamentous algae & phytoplankton density of prairie streams	Northwestern Glaciated Plains; stream entrenchment ratio; riparian canopy density; woody riparian density.	Suplee (2004)

4.2 Confidence in the Applicability of Reference Sites

The degree of confidence one has in the accuracy of a comparison between reference and a given site can vary, and is influenced by how the reference sites were chosen (i.e., the methods outlined in this report), the types of geospatial stratification factors used to apply the reference sites, and the quantity and quality of data supporting the stratification factors. In general, increased confidence incurs increased cost because of the need for: higher levels of data resolution for the geospatial characteristics; expansion of reference stream sampling; improved understanding of cause/effect relationship between pollutants and impacts to water uses; and, more time to conduct these undertakings.

Deciding if greater confidence (i.e., incurring higher cost) is needed when making a reference-to-stream site comparison can be influenced by a number of considerations. Some key considerations that the DEQ has identified are: public interest in the resource; watershed sensitivity; and resource value. Public interest considers the interest level of local or other groups, the degree of public awareness of the waterbody, etc. Watershed sensitivity considers issues such as the erosiveness of basin soils and the consequential need for bank-stabilizing riparian vegetation, time required for regeneration of riparian areas, etc. Resource value should consider the potential value of the water resource (e.g., contains a blue ribbon fishery, is a core bull trout area) and should also consider the role of the waterbody in the adjacent region (e.g., it is adjacent to a wilderness area, surrounds a national park such as Yellowstone, or is an important agricultural water source). If, for example, the degree of public interest, watershed sensitivity and resource value were all found to be “high”, then a high degree of confidence in the reference-to-stream site comparison is needed.

4.3 Precautionary Considerations When Using the Reference Sites

There are limitations to the use of the reference stream data. Most of the sites are located in lower Strahler stream orders — mainly 1st through 4th but including a few 5th order sites — and the data are most applicable to streams of that size range (the so-called “wadeable” streams). Therefore, the extension of these data to sites from much larger waterbodies (e.g., Yellowstone River, 6th order) should be undertaken with caution.

Some areas of the state are better represented than others. Density of reference sites in Western Montana is generally good, and the Middle Rockies ecoregion is arguably the best represented; it has the largest number of reference sites among all Montana ecoregions and contains sites from high mountain streams, foothill and valley streams, and a 5th order river (the Blackfoot near its mouth). On the other hand, the number of sites in the Northwestern Great Plains and Northwestern Glaciated Plains are less dense and there are probably a number of stream types that remain to be sampled. For example, we know that that the “Woody Draw” (Hansen et al. 1995) is a common riparian stream type in the Northwestern Glaciated Plains, but there are currently no reference examples of the Woody Draw in the dataset.

4.4 Recommendations For Future Directions, and Data Availability

4.4.1 Recommendations For Future Directions

To date, work on reference sites has focused on the sampling of existing sites, identifying new sites (including locating reference sites in areas where they have been difficult to locate, such as the Northwestern Great Plains), and the collation of existing data from a variety of sources inside and outside of the DEQ. Our experience has shown that annual sampling of a few existing sites while concurrently locating a number of new ones is better than exclusively focusing on locating new sites. This is because the year-to-year variability of conditions at existing sites (driven by climate, fires, etc.) can be characterized, and new field crews can be trained at familiar sites. And at the same time, the total number of reference sites in the database continues to increase, thus providing greater statistical confidence in the analyses made with the data. We have also found that focusing on a region or two each field season is more effective than attempting statewide sampling, as staff and field crews can become better acquainted with a smaller geographic area and the likelihood of locating high-quality reference sites increases.

Currently, the Reference Stream Project is designed to sample each year specific regions of the state (northeast Montana, southwest Montana, etc.) that roughly correspond to Omernik level-III ecoregions. As discussed in Section 4.1, ecoregions (level III and IV) have shown themselves to be useful for the initial stratification of both chemical and biological stream data in Montana. Therefore, we suggest that ecoregions be used to select and monitor reference sites. A single level-III (coarse) ecoregion could be selected each year, with specific level IV ecoregions targeted within it, as needed. The specific level IV ecoregions should be selected based on identified spatial gaps in the landscape coverage of the existing reference site list. Individual sites within each level IV ecoregion can be targeted for sampling based on the preliminary GIS approaches already discussed in Section 2.1. Nested within this approach should be a degree of flexibility, so that as other key stream-influencing variables (e.g., elevation, riparian type) “rise to the top” in terms of importance, they may be incorporated into the sampling plan.

How often should reference sites be re-sampled? Sanders et al. (1983) describe in detail techniques that can be used to determine sampling frequency, most of which involve statistical analysis of time-series data. We have not carried out these analyses to date. There are currently several sites in the Reference Stream Project having up to three continuous years of data, plus additional non-consecutive sampling events. It is intended that these data be analyzed so that a more reasonable return interval can be developed. However at this point, a BPJ recommendation for a return interval would be on the order of every 5 years. We also recommend continuation of the current sampling approach, wherein three sampling events per field season (roughly once each month during the summer) are completed at each site. This is because reference sites are targeted and few in number, and therefore water quality and biological parameters that vary over the sampling season cannot be characterized — as can be done for stream sites in a large, random-sampling design — by undertaking sampling at many spatially-separated sites during the same time period.

4.4.2 Data Availability

The type and quantity of data inventoried during the 2004 collation process varied greatly from site to site, and are available, along with data from the Reference Stream Project, in several different formats (electronic, hardcopy, etc.). A general inventory of existing data from all sites is available in the DEQ Water Quality Planning Bureau, although the data itself may be housed in a number of different locations (STORET database, hardcopy files, electronic spreadsheets, etc.; see Appendix B for details on database resources). The BPJ criteria assessment records (Tables 2.0, 3.0) are also housed in the DEQ Water Quality Planning Bureau. We acknowledge that a more detailed plan is needed for future tracking and maintenance of data associated with reference sites, and we are coordinating with the DEQ Data Management Section to achieve this goal.

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Appendix A. List of Reference Sites and their Locations

Table A. List of reference sites and associated location information for the Columbia and Hudson Bay major basins.

Major Basin	4th field HUC	LAT(DD)	LONG(DD)	Stream Site Name	REF. SITE No.	Review Notes*
St. Mary (Hudson Bay drainage)	10010001	48.96806	-113.68222	Belly River at 3-mile campsite (Glacier NP)	BellyRiv_408_C	Rvd
St. Mary (Hudson Bay drainage)	10010002	48.7992	-113.6558	Swiftcurrent Creek at Many Glacier MT	Swiftcur_132_C	Estb
Columbia	17010202	46.7225	-113.6822	Rock Creek near Clinton	RockCree_071_C	Rvd
Columbia	17010202	46.70889	-113.6725	Rock Creek	RockCree_070_C	Rvd
Columbia	17010203	46.89944	-113.7561	Blackfoot River	Blackfoo_006_C	Rvd
Columbia	17010203	47.2281	-112.8472	Trib of N. Fork Blackfoot	TribofNF_091_C	Rvd
Columbia	17010204	47.4025	-115.5164	Silver Creek (Site 2)	SilverCr_078_C	Rvd
Columbia	17010204	47.31055	-115.4047	Deer Creek	DeerCree_023_C	Rvd
Columbia	17010205	46.1928	-114.2572	Roaring Lion Creek	RoaringL_068_C	Rvd
Columbia	17010205	46.1928	-114.257	Roaring Lion Ck	RoaringL_241_C	P
Columbia	17010205	46.1939	-114.2406	ROARING LION CREEK NEAR TRAIL HEAD	ROARINGL_069_C	Rvd
Columbia	17010206	48.8292	-114.4908	Moose Creek	MooseCre_056_C	Rvd
Columbia	17010207	48.0639	-113.2453	Schafer Creek	SchaferC_074_C	Rvd
Columbia	17010209	47.9789	-113.5608	SOUTH FK FLATHEAD R. ABV HUNGRY HORSE RES	SOUTHFKF_115_C	Estb
Columbia	17010209	47.8055	-113.414	S. Fk. Flathead R.	SFkFlath_244_C	P
Columbia	17010210	48.74	-114.7331	Chepat Creek	ChepatCr_108_C	Estb
Columbia	17010210	48.7513	-114.727	Chepat Ck	ChepatCk_194_C	P
Columbia	17010210	48.62556	-114.5275	Chicken Creek	ChickenC_019_C	Rvd
Columbia	17010211	47.75834	-113.7811	Goat Creek (Site 2)	GoatCree_043_C	Rvd
Columbia	17010213	48.125	-115.7275	E. Fk. Bull River (Down)	EFkBullR_025_C	Rvd
Columbia	17010213	48.1219	-115.6983	EAST FORK BULL RIVER ABV. N. FK OF E. FK.	EASTFORK_031_C	Rvd
Columbia	17010213	48.1212	-115.698	E. Fk. Bull R.	EfkBullR_205_C	P
Columbia	17010213	47.7129	-115.128	Deerhorn Ck	Deerhorn_202_C	M

* **Rvd** means reviewed using the screening process in this report; **Estb** means an established reference site, from Bahls et al. (1992) or Suplee (2004); **P** means site was rated as pristine by USU; **M** means site was rated as minimally disturbed by USU.

Table A, Cont. List of reference sites and associated location information for the Upper Missouri major basin.

Major Basin	4th field HUC	LAT(DD)	LONG(DD)	Stream Site Name	REF. SITE No.	Review Notes*
Upper Missouri	10020001	45.0653	-113.2153	Browns Creek	BrownsCr_015_C	Rvd
Upper Missouri	10020001	44.64444	-111.6636	Elk Springs Creek	ElkSprin_037_C	Rvd
Upper Missouri	10020002	44.9425	-112.4294	Cottonwood Cr	Cottonwo_021_C	Rvd
Upper Missouri	10020003	45.12194	-112.0392	N. FK. Greenhorn Creek	NFKGreen_058_C	Rvd
Upper Missouri	10020003	45.5217	-111.9853	Mill (Up)	MillUp99_053_C	Rvd
Upper Missouri	10020004	45.37305	-113.6197	Little Lake Creek	LittleLa_049_C	Rvd
Upper Missouri	10020004	45.7335	-113.57283	Mussigbrod Cr.	Mussigbr_154_C	Rvd
Upper Missouri	10020004	45.9072	-113.4811	Pintler Creek	PintlerC_066_C	Rvd
Upper Missouri	10020004	45.91083	-113.2172	LaMarche Creek	LaMarche_048_C	Rvd
Upper Missouri	10020004	45.9985	-113.19	Seymore Ck.	SeymoreC_249_C	P
Upper Missouri	10020004	45.99583	-113.1872	Seymour Creek	SeymourC_076_C	Rvd
Upper Missouri	10020004	45.4481	-112.8278	Willow Cr (I)	WillowCr_103_C	Rvd
Upper Missouri	10020004	45.4381	-112.7422	Willow Cr (II)	WillowCr_104_C	Rvd
Upper Missouri	10020005	45.6047	-111.897	S. Fk. Willow Ck.	SFkWillo_245_C	M
Upper Missouri	10020007	45.3408	-111.718	O'Dell Ck	ODellCk9_236_C	M
Upper Missouri	10020007	45.115	-111.4981	No Man Creek	NoManCre_059_C	Rvd
Upper Missouri	10020007	45.6267	-111.414	Elk Ck	ElkCk999_209_C	M
Upper Missouri	10020007	44.905	-111.3725	W. Fk. Beaver Creek	WFkBeave_095_C	Rvd
Upper Missouri	10020007	44.8764	-111.34	Cabin Creek	CabinCre_016_C	Rvd
Upper Missouri	10020007	44.65778	-111.06972	Madison R., nr West Yellowstone in Yellowstone NP	MadisonR_406_C	Rvd
Upper Missouri	10020008	45.41028	-111.39861	SF Spanish Cr, Spanish Peaks Wilderness	SFSpanis_407_C	Rvd
Upper Missouri	10020008	45.3904	-111.24	Cascade Ck	CascadeC_193_C	P
Upper Missouri	10020008	45.05444	-111.1564	Gallatin River	Gallatin_040_C	Rvd
Upper Missouri	10020008	45.53444	-111.0806	S. Cottonwood Creek	SCottonw_073_C	Rvd
Upper Missouri	10030102	47.2131	-112.495	Lone Pine Creek	LonePine_051_C	Rvd
Upper Missouri	10030102	47.4994	-110.7164	Highwood Creek	Highwood_044_W	Rvd
Upper Missouri	10030103	46.9947	-111.4858	Whitetail Creek	Whitetai_102_W	Rvd
Upper Missouri	10030103	46.845	-110.96	Calf Creek	CalfCree_017_C	Rvd
Upper Missouri	10030103	46.8464	-110.958	calfck	calfck99_192_C	M
Upper Missouri	10030104	47.4916	-112.909	sfksun	sfksun99_250_C	P
Upper Missouri	10030104	47.4917	-112.9086	South Fk. Sun River	SouthFkS_080_C	Rvd
Upper Missouri	10030104	47.5064	-112.8903	SUN RIVER S. FORK BELOW STRAIGHT CREEK	SUNRIVER_116_C	Estb
Upper Missouri	10040201	46.2506	-109.7689	Fish Creek	FishCree_038_W	Rvd

* **Rvd** means reviewed using the screening process in this report; **Estb** means an established reference site, from Bahls et al. (1992) or Suplee (2004); **P** means site was rated as pristine by USU; **M** means site was rated as minimally disturbed by USU.

Table A, Cont. List of reference sites and associated location information for the Lower Missouri major basin.

Major Basin	4th field HUC	LAT(DD)	LONG(DD)	Stream Site Name	REF. SITE No.	Review Notes*
Lower Missouri	10030201	48.04056	-112.9000	Crazy Cr blw Mount Patrick Gass, Bob Marshall Wilderness	CrazyCre_409_C	Rvd
Lower Missouri	10030204	48.7567	-111.5216	Willow Creek	WillowCr_172_W	Rvd
Lower Missouri	10030205	47.92	-112.8339	Waldron Creek	WaldronC_117_C	Estb
Lower Missouri	10030205	47.9193	-112.817	waldrn	waldrn99_270_C	P
Lower Missouri	10030205	47.9711	-112.811	nfktet	nfktet99_234_C	P
Lower Missouri	10030205	47.96695	-112.8075	N. Fk. Teton River	NFkTeton_114_C	Estb
Lower Missouri	10030205	48.01305	-112.6931	Blackleaf Creek (Site 1)	Blacklea_007_C	Rvd
Lower Missouri	10030205	48.01278	-112.5633	Blackleaf Creek (Site 2)	Blacklea_008_W	Rvd
Lower Missouri	10040101	47.92111	-110.035	Eagle Creek (Site 3)	EagleCre_030_W	Rvd
Lower Missouri	10040101	48.10083	-109.7689	Eagle Creek (Site 1)	EagleCre_028_C	Rvd
Lower Missouri	10040102	47.62564	-109.83562	Arrow Creek	ArrowCre_135_W	Rvd
Lower Missouri	10040103	47.07944	-109.5989	Beaver Creek	BeaverCr_002_W	Rvd
Lower Missouri	10040104	47.86111	-108.9633	Cow Creek	CowCreek_022_W	Rvd
Lower Missouri	10040106	47.3413	-106.363	Little Dry Cr.	LittleDr_151_W	Rvd
Lower Missouri	10040201	46.6756	-110.4389	Basin Creek	BasinCre_001_C	Rvd
Lower Missouri	10050004	48.30611	-109.4906	Clear Creek (Nut pilot)	ClearCre_121_W	Estb
Lower Missouri	10050010	48.92265	-108.37948	Woody Island Coulee	WoodyIsl_174_W	Rvd
Lower Missouri	10050011	48.95661	-107.85937	Whitewater Creek	Whitewat_170_W	Rvd
Lower Missouri	10050011	48.600061	-107.519465	Whitewater Creek	Whitewat_169_W	Rvd
Lower Missouri	10050015	48.6569	-107.0389	Rock Cr (BLM land)	RockCrBL_122_W	Estb
Lower Missouri	10050015	48.59028	-107.0011	Rock Creek (Site 2)	RockCree_124_W	Estb
Lower Missouri	10050015	48.58472	-106.9625	Willow Creek	WillowCr_171_W	Rvd
Lower Missouri	10050015	48.6489	-106.9025	Bitter Cr	BitterCr_120_W	Rvd
Lower Missouri	10050015	48.8789	-106.8992	ROCK CREEK NORTHEAST OF HINSDALE	ROCKCREE_125_W	Estb
Lower Missouri	10050015	48.87583	-106.8967	Rock Creek (Site 1)	RockCree_123_W	Estb
Lower Missouri	10050015	48.9694	-106.8389	ROCK CREEK BELOW HORSE CREEK NEAR INT BOUNDARY	ROCKCREE_133_W	Estb
Lower Missouri	10060001	48.08778	-105.6781	Wolf Creek @ Wolf Pt.	WolfCree_130_W	Estb
Lower Missouri	10060001	48.2236	-105.5175	Tule Creek	TuleCree_092_W	Rvd
Lower Missouri	10060001	48.18355	-105.49147	Tule Creek	TuleCree_164_W	Rvd
Lower Missouri	10060002	47.70639	-105.2456	Pasture Creek (Site 1)	PastureC_064_W	Rvd
Lower Missouri	10060002	47.63972	-105.1617	Pasture Creek (Site 2)	PastureC_065_W	Rvd
Lower Missouri	10060002	47.75806	-104.9228	E. Redwater Creek	E.Redwat_027_W	Rvd
Lower Missouri	10060004	48.9442	-106.2503	WEST FORK POPLAR RIVER AT BRIDGE ON COUNT	WESTFORK_127_W	Estb
Lower Missouri	10060004	48.8081	-106.0206	WEST FORK POPLAR RIVER NEAR RICHLAND MONT	WESTFORK_128_W	Estb
Lower Missouri	10060004	48.7478	-105.9286	WEST FORK POPLAR RIVER S OF PEERLESS	WESTFORK_129_W	Estb
Lower Missouri	10060004	48.69695	-105.8319	W. Fk. Poplar River	WFkPopla_126_W	Estb
Lower Missouri	10060004	48.6225	-105.6525	WEST FORK POPLAR RIVER NEAR FOUR BUTTES	WESTFORK_099_W	Rvd

* **Rvd** means reviewed using the screening process in this report; **Estb** means an established reference site, from Bahls et al. (1992) or Suplee (2004); **P** means site was rated as pristine by USU; **M** means site was rated as minimally disturbed by USU.

Table A, Cont. List of reference sites and associated location information for the Yellowstone major basin.

Major Basin	4th field HUC	LAT(DD)	LONG(DD)	Stream Site Name	REF. SITE No.	Review Notes*
Yellowstone	10070001	45.02944	-110.69944	Gardner River at mouth, Yellowstone NP	GardnerR_404_C	Rvd
Yellowstone	10070002	45.3053	-110.9819	BIG CREEK ABOVE BIG CREEK STATION	BIGCREEK_110_C	Estb
Yellowstone	10070002	45.3034	-110.94	Big Ck	BigCk999_180_C	M
Yellowstone	10070002	45.5063	-110.789	Pine Ck	PineCk99_238_C	M
Yellowstone	10070002	45.6347	-110.7511	ARMSTRONG SPRING CREEK AT O'HAIR RANCH	ARMSTRON_109_W	Estb
Yellowstone	10070002	45.3408	-110.2464	Fourmile Creek	Fourmile_112_C	Estb
Yellowstone	10070002	45.3407	-110.246	Four Mile Ck	FourMile_212_C	P
Yellowstone	10070005	45.3981	-109.9683	WEST FORK STILLWATER CUS001 ABOVE ADIT	WESTFORK_118_C	Estb
Yellowstone	10070005	45.3988	-109.961	wfkstl	wfkstl99_274_C	P
Yellowstone	10070005	45.22667	-109.6058	East Rosebud Creek	EastRose_033_C	Rvd
Yellowstone	10070006	45.0794	-109.4081	LAKE FORK OF ROCK CREEK	LAKEFORK_113_C	Estb
Yellowstone	10070006	45.05361	-109.4069	Wyoming Creek	WyomingC_107_C	Rvd
Yellowstone	10070006	45.15056	-109.33944	West Fork Rock Cr, abv Silver Run	WFRockCr_405_C	Rvd
Yellowstone	10070006	45.09806	-109.2992	Seeley Creek	SeeleyCr_075_C	Rvd
Yellowstone	10070006	45.2356	-108.925	lfkrok	lfkrok99_222_W	P
Yellowstone	10070008	45.31665	-108.5406	Pryor Creek	PryorCre_159_C	Rvd
Yellowstone	10080010	45.1334	-108.428	crookd	crookd99_200_C	P
Yellowstone	10080010	45.0433	-108.385	CROOKED CREEK ABOVE TILLETT RANCH	CROOKEDC_111_C	Estb
Yellowstone	10080015	45.44316	-108.16282	Beauvais Creek	Beauvois_136_W	Rvd
Yellowstone	10080015	45.47694	-108.0081	Beauvais Creek near ST. Xavier MT	Beauvais_131_W	Estb
Yellowstone	10080015	45.4328	-107.9619	Muddy Creek	MuddyCre_057_W	Rvd
Yellowstone	10090102	45.3092	-106.2497	Cow Creek	CowCreek_141_W	Rvd
Yellowstone	10090102	46.189011	-105.621715	Pumpkin Creek	PumpkinC_161_W	Rvd
Yellowstone	10090208	45.3189	-105.3178	Little Powder River	LittlePo_050_W	Rvd
Yellowstone	10100004	46.7917	-104.5583	Cedar Cr.	CedarCr9_140_W	Rvd
Yellowstone	10100005	46.73498	-105.057378	O Fallon	OFallon9_157_W	Rvd
Yellowstone	10100005	46.47068	-104.76994	O Fallon	OFallon9_156_W	Rvd
Yellowstone	10100005	46.47111	-104.7697	O'Fallon Creek (Site 2)	OFallonC_062_W	Rvd
Yellowstone	10100005	46.16694	-104.71528	Milk Creek near mouth	MilkCree_416_W	Rvd
Yellowstone	10100005	46.1364	-104.6669	Spring Creek	SpringCr_081_W	Rvd
Yellowstone	10110201	44.9952	-104.42346	Little Missouri River	LittleMi_152_W	Rvd
Yellowstone	10110201	46.06778	-104.33583	Little Beaver Creek	LittleBe_410_W	Rvd
Yellowstone	10110202	45.8444	-104.1439	Box Elder Creek	BoxElder_013_W	Rvd
Yellowstone	10110202	45.84472	-104.14361	Box Elder Creek	BoxElder_137_W	Rvd
Yellowstone	10110202	45.8448	-104.14289	Box Elder Creek	BoxElder_138_W	Rvd
Yellowstone	10110202	45.89405	-104.07163	Box Elder Creek	BoxElder_382_W	Rvd

* **Rvd** means reviewed using the screening process in this report; **Estb** means an established reference site, from Bahls et al. (1992) or Suplee (2004); **P** means site was rated as pristine by USU; **M** means site was rated as minimally disturbed by USU.

Appendix B. Database Stations Associated with Reference Sites

To aid in the cataloging of data associated with each reference site, applicable station ID's from large water-quality databases (Legacy STORET, modernized STORET, NWIS) were associated with each reference site (Table B, below). Some database stations are not located precisely at the reference site coordinates, and database data could pre-date (by some years) a site's recognition as a reference site. To be associated with a reference site, a database station had to be within 0.5 miles up- or downstream of the reference site coordinates. If the possibility existed that a database station downstream of a reference site was located below a mine adit or other known point source, the database station was not associated with the reference site.

Table B. Database station ID's associated with reference sites. In the 'Database' column, MONT-DEQ and MT-DEQ refer to organizational codes found in modernized STORET. MONT-DEQ is associated with DEQ data collected prior to 1998, MT-DEQ with DEQ data collected since 2000. NWIS is the USGS's National Water Quality Information System database. EMAP and USU designate data collected for particular EPA projects, which will ultimately be housed in modernized STORET.

REF SITE No.	Stream Site Name	LAT (dd)	LONG (dd)	Database Station ID	Database
ARMSTRON_109_W	ARMSTRONG SPRING CREEK AT O'HAIR RANCH	45.6347	-110.7511	2544AR01	Legacy STORET, or Modern STORET under MONT-DEQ
ArrowCre_135_W	Arrow Creek	47.62564	-109.83562	M21ARRWC02	Modern STORET, under MT-DEQ
BasinCre_001_C	Basin Creek	46.6756	-110.4389	WMTP99-0716	EMAP
Beauvais_131_W	Beauvais Creek near ST. Xavier MT	45.47694	-108.0081	06288200	NWIS
BeaverCr_002_W	Beaver Creek	47.07944	-109.5989	M22BEVRC04	Modern STORET, under MT-DEQ
BellyRiv_408_C	Belly River at 3-mile campsite (Glacier NP)	48.96806	-113.68222	GLAC_NURE_732	Legacy STORET
BigCk999_180_C	Big Ck	45.3034	-110.94	EPA01-436	USU
BIGCREEK_110_C	BIG CREEK ABOVE BIG CREEK STATION	45.3053	-110.9819	2241BI01	Legacy STORET, or Modern STORET under MONT-DEQ
BitterCr_120_W	Bitter Cr	48.6489	-106.9025	M43BITRC01	Modern STORET, under MT-DEQ
Blackfoo_006_C	Blackfoot River	46.89944	-113.7561	CFR14	Modern STORET, under MT-DEQ
Blackfoo_006_C	Blackfoot River	46.89944	-113.7561	4118BL01	Legacy STORET, or Modern STORET under MONT-DEQ
Blackfoo_006_C	Blackfoot River	46.8997	-113.7556	12340000	NWIS
Blacklea_007_C	Blackleaf Creek (Site 1)	48.01305	-112.6931	WMTP99-R031	EMAP
Blacklea_008_W	Blackleaf Creek (Site 2)	48.01278	-112.5633	MT14BLKLC01	Modern STORET, under MT-DEQ
BoxElder_013_W	Box Elder Creek	45.8444	-104.1439	WMTP99-0623	EMAP
BoxElder_382_W	Box Elder Creek	45.89405	-104.07163	3097BO01	Legacy STORET, or Modern STORET under MONT-DEQ
BrownsCr_015_C	Browns Creek	45.0653	-113.2153	WMTP99-0745	EMAP
CabinCre_016_C	Cabin Creek	44.8764	-111.34	06038550	NWIS
CabinCre_016_C	Cabin Creek	44.8764	-111.34	1738CA01	Legacy STORET, or Modern STORET under MONT-DEQ
calfck99_192_C	calfck	46.8464	-110.958	EPA01-425	USU
calfck99_192_C	calfck	46.8464	-110.958	4141CA01	Legacy STORET, or Modern STORET under MONT-DEQ
CalfCree_017_C	Calf Creek	46.845	-110.96	REFCAC	Modern STORET, under MT-DEQ
CalfCree_017_C	Calf Creek	46.845	-110.96	WMTP99-R027	EMAP
CascadeC_193_C	Cascade Ck	45.3904	-111.241	EPA01-438	USU
CedarCr9_140_W	Cedar Cr.	46.7917	-104.5583	BKK038	Legacy STORET, or Modern STORET under MONT-DEQ
ChepatCk_194_C	Chepat Ck	48.7513	-114.727	EPA01-452	USU
ChepatCr_108_C	Chepat Creek	48.74	-114.7331	STSF08	DNRC
ChickenC_019_C	Chicken Creek	48.62556	-114.5275	STSF03	DNRC
ClearCre_121_W	Clear Creek (Nut pilot)	48.30611	-109.4906	REFCC	Modern STORET, under MT-DEQ
Cottonwo_021_C	Cottonwood Cr	44.9425	-112.4294	M02CTWD01	Modern STORET, under MT-DEQ
CowCreek_022_W	Cow Creek	47.86111	-108.9633	WMTP99-R032	EMAP
CowCreek_141_W	Cow Creek	45.3092	-106.2497	2280CO01	Legacy STORET, or Modern STORET under MONT-DEQ

Table B, Cont. Database station ID's associated with reference sites.

REF SITE No.	Stream Site Name	LAT (dd)	LONG (dd)	Database Station ID	Database
crookd99_200_C	crookd	45.1334	-108.428	EPA01-429	USU
CROOKEDC_111_C	CROOKED CREEK ABOVE TILLETT RANCH	45.0433	-108.385	1962CR02	Legacy STORET, or Modern STORET under MONT-DEQ
DeerCree_023_C	Deer Creek	47.31055	-115.4047	C04DEERC01	Modern STORET, under MT-DEQ
Deerhorn_202_C	Deerhorn Ck	47.7129	-115.128	EPA01-449	USU
E.Redwat_027_W	E. Redwater Creek	47.75806	-104.9228	M48RDWEC04	Modern STORET, under MT-DEQ
EagleCre_028_C	Eagle Creek (Site 1)	48.10083	-109.7689	M23EAGLC01	Modern STORET, under MT-DEQ
EagleCre_030_W	Eagle Creek (Site 3)	47.92111	-110.035	M23EAGLC06	Modern STORET, under MT-DEQ
EASTFORK_031_C	EAST FORK BULL RIVER ABV. N. FK OF E. FK.	48.1219	-115.6983	EPA01-450	EMAP
EASTFORK_031_C	EAST FORK BULL RIVER ABV N. FK. OF E. FK	48.1219	-115.6983	5503EA01	Legacy STORET, or Modern STORET under MONT-DEQ
EastRose_033_C	East Rosebud Creek	45.22667	-109.6058	06202915	NWIS
EastRose_033_C	East Rosebud Creek	45.22667	-109.6058	REFERC	Modern STORET, under MT-DEQ
EFkBullR_025_C	E. Fk. Bull River (Down)	48.125	-115.7275	WMTP99-R019	EMAP
ElkCk999_209_C	Elk Ck	45.6267	-111.414	EPA01-440	USU
ElkSprin_037_C	Elk Springs Creek	44.64444	-111.6636	M01ELKC01	Modern STORET, under MT-DEQ
ElkSprin_037_C	Elk Springs Creek	44.64444	-111.6636	WMTP99-R038	EMAP
FishCree_038_W	Fish Creek	46.2506	-109.7689	WMTP99-0628	EMAP
Fourmile_112_C	Fourmile Creek	45.3408	-110.2464	2247FO01	Legacy STORET, or Modern STORET under MONT-DEQ
FourMile_212_C	Four Mile Ck	45.3407	-110.246	EPA01-432	USU
Gallatin_040_C	Gallatin River	45.05444	-111.1564	M05GLTNR01	Modern STORET, under MT-DEQ
GardnerR_404_C	Gardner River at mouth, Yellowstone NP	45.02944	-110.69944	Y01GARDR01	Modern STORET, under MT-DEQ
GoatCree_043_C	Goat Creek (Site 2)	47.75834	-113.7811	C10GOATC04	Modern STORET, under MT-DEQ
Highwood_044_W	Highwood Creek	47.4994	-110.7164	WMTP99-0729	EMAP
LAKEFORK_113_C	LAKE FORK OF ROCK CREEK	45.0794	-109.4081	1954LA01	Legacy STORET, or Modern STORET under MONT-DEQ
LaMarche_048_C	LaMarche Creek	45.91083	-113.2172	M03LMCHC01	Modern STORET, under MT-DEQ
lfkrok99_222_W	lfkrok	45.2356	-108.925	EPA01-428	USU
lfkrok99_222_W	lfkrok	45.2356	-108.925	451409108552701	NWIS
LittleDr_151_W	Little Dry Cr.	47.3413	-106.363	M29LDRYC01	Modern STORET, under MT-DEQ
LittleDr_151_W	Little Dry Cr.	47.3413	-106.363	4677LI01	Legacy STORET, or Modern STORET under MONT-DEQ
LittleDr_151_W	Little Dry Cr.	47.3413	-106.363	BKK070	Modern STORET, under MT-DEQ
LittleDr_151_W	Little Dry Cr.	47.3413	-106.363	06130950	NWIS
LittleDr_151_W	Little Dry Cr.	47.3413	-106.363	BKK069	Modern STORET, under MT-DEQ
LittleLa_049_C	Little Lake Creek	45.37305	-113.6197	M03LTLKC01	Modern STORET, under MT-DEQ
LittlePo_050_W	Little Powder River	45.3189	-105.3178	WMTP99-0648	EMAP
LonePine_051_C	Lone Pine Creek	47.2131	-112.495	WMTP99-0722	EMAP
MadisonR_406_C	Madison R., nr West Yellowstone in Yellowstone NP	44.65778	-111.06972	06037500	NWIS

Table B, Cont. Database station ID's associated with reference sites.

REF SITE No.	Stream Site Name	LAT (dd)	LONG (dd)	Database Station ID	Database
MooseCre_056_C	Moose Creek	48.8292	-114.4908	WMTP99-0515	EMAP
MuddyCre_057_W	Muddy Creek	45.4328	-107.9619	MT634738	EMAP
Mussigbr_154_C	Mussigbrod Cr.	45.7335	-113.57283	BKK091	Modern STORET, under MT-DEQ
NFKGreen_058_C	N. FK. Greenhorn Creek	45.12194	-112.0392	M04GHCNF01	Modern STORET, under MT-DEQ
nfktet99_234_C	nfktet	47.9711	-112.811	EPA01-423	USU
nfktet99_234_C	nfktet	47.9711	-112.811	5426TE01	Legacy STORET, or Modern STORET under MONT-DEQ
NFkTeton_114_C	N. Fk. Teton River	47.96695	-112.8075	REFNFTR	Modern STORET, under MT-DEQ
NFkTeton_114_C	N. Fk. Teton River	47.96695	-112.8075	WMTP99-R002	EMAP
NoManCre_059_C	No Man Creek	45.115	-111.4981	MAD-004	EMAP
ODellCk9_236_C	O'Dell Ck	45.3408	-111.718	452032111425201	NWIS
ODellCk9_236_C	O'Dell Ck	45.3408	-111.718	EPA01-441	USU
OFallon9_156_W	O Fallon	46.47068	-104.76994	Y22OFALC07	Modern STORET, under MT-DEQ
OFallonC_062_W	O'Fallon Creek (Site 2)	46.47111	-104.7697	REFOFC2	Modern STORET, under MT-DEQ
PastureC_064_W	Pasture Creek (Site 1)	47.70639	-105.2456	5185PA01	Modern STORET, under MT-DEQ
PastureC_064_W	Pasture Creek (Site 1)	47.70639	-105.2456	M48PSTRC01	Modern STORET, under MT-DEQ
PastureC_065_W	Pasture Creek (Site 2)	47.63972	-105.1617	M48PSTRC02	Modern STORET, under MT-DEQ
PineCk99_238_C	Pine Ck	45.5063	-110.789	EPA01-435	USU
PintlerC_066_C	Pintler Creek	45.9072	-113.4811	WMTP99-0517	EMAP
RoaringL_068_C	Roaring Lion Creek	46.1928	-114.2572	EPA01-448	EMAP
ROARINGL_069_C	ROARING LION CREEK NEAR TRAIL HEAD	46.1939	-114.2406	3314RO01	Legacy STORET, or Modern STORET under MONT-DEQ
RockCrBL_122_W	Rock Cr (BLM land)	48.6569	-107.0389	M43ROCKC01	Modern STORET, under MT-DEQ
RockCree_070_C	Rock Creek nr Clinton	46.7225	-113.6822	464321113405601	NWIS
RockCree_070_C	Rock Creek	46.70889	-113.6725	3918RO01	Legacy STORET, or Modern STORET under MONT-DEQ
RockCree_071_C	Rock Creek near Clinton	46.7225	-113.6822	12334510	NWIS
RockCree_123_W	Rock Creek (Site 1)	48.87583	-106.8967	REFRC1	Modern STORET, under MT-DEQ
RockCree_123_W	Rock Creek (Site 1)	48.87583	-106.8967	WMTP99-R005	EMAP
RockCree_124_W	Rock Creek (Site 2)	48.59028	-107.0011	REFRC2	Modern STORET, under MT-DEQ
ROCKCREE_125_W	ROCK CREEK NORTHEAST OF HINSDALE	48.8789	-106.8992	6472RO01	Legacy STORET, or Modern STORET under MONT-DEQ
ROCKCREE_133_W	ROCK CREEK BELOW HORSE CREEK NEAR INT BOUNDARY	48.9694	-106.8389	06169500	NWIS
SchaferC_074_C	Schafer Creek	48.0639	-113.2453	5522SC01	Legacy STORET, or Modern STORET under MONT-DEQ
SCottonw_073_C	S. Cottonwood Creek	45.53444	-111.0806	M05SCTNC01-A	Modern STORET, under MT-DEQ
SCottonw_073_C	S. Cottonwood Creek	45.53444	-111.0806	M05SCTNC01-B	Modern STORET, under MT-DEQ
SCottonw_073_C	S. Cottonwood Creek	45.53444	-111.0806	M05SCTNC01-C	Modern STORET, under MT-DEQ
SCottonw_073_C	S. Cottonwood Creek	45.53444	-111.0806	M05SCTNC01-D	Modern STORET, under MT-DEQ
SeeleyCr_075_C	Seeley Creek	45.09806	-109.2992	WMTP99-R035	EMAP
SeeleyCr_075_C	Seeley Creek	45.09806	-109.2992	REFSEC	Modern STORET, under MT-DEQ

Table B, Cont. Database station ID's associated with reference sites.

REF SITE No.	Stream Site Name	LAT (dd)	LONG (dd)	Database Station ID	Database
SeymoreC_249_C	Seymore Ck.	45.9985	-113.19	EPA01-446	USU
SeymourC_076_C	Seymour Creek	45.99583	-113.1872	3122SE01	Legacy STORET, or Modern STORET under MONT-DEQ
SeymourC_076_C	Seymour Creek	45.99583	-113.1872	BKK126	Modern STORET, under MT-DEQ
SeymourC_076_C	Seymour Creek	45.99583	-113.1872	WMTP99-R015	EMAP
SFkFlath_244_C	S. Fk. Flathead R.	47.8055	-113.414	EPA01-453	USU
SFkWillo_245_C	S. Fk. Willow Ck.	45.6047	-111.897	EPA01-454	USU
SilverCr_078_C	Silver Creek (Site 2)	47.4025	-115.5164	C04SLVRC04	Modern STORET, under MT-DEQ
SilverCr_078_C	Silver Creek (Site 2)	47.4025	-115.5164	C04SLVRC02	Modern STORET, under MT-DEQ
SOUTHFKF_115_C	SOUTH FK FLATHEAD R. ABV HUNGRY HORSE RES	47.9789	-113.5608	C08FRSFK01	Modern STORET, under MT-DEQ
SOUTHFKF_115_C	SOUTH FK FLATHEAD R. ABV HUNGRY HORSE RES	47.9789	-113.5608	12359800	NWIS
SOUTHFKF_115_C	SOUTH FK FLATHEAD R. ABV HUNGRY HORSE RES	47.9789	-113.5608	5419SO01	Legacy STORET, or Modern STORET under MONT-DEQ
SouthFkS_080_C	South Fk. Sun River	47.4917	-112.9086	EPA01-424	EMAP
SpringCr_081_W	Spring Creek	46.1364	-104.6669	WMTP99-0549	EMAP
SUNRIVER_116_C	SUN RIVER S. FORK BELOW STRAIGHT CREEK	47.5064	-112.8903	4825SU01	Legacy STORET, or Modern STORET under MONT-DEQ
Swiftcur_132_C	Swiftcurrent Creek at Many Glacier MT	48.7992	-113.6558	05014500	NWIS
TribofNF_091_C	Trib of N. Fork Blackfoot	47.2281	-112.8472	WMTP99-0837	EMAP
TuleCree_092_W	Tule Creek	48.2236	-105.5175	WMTP99-0804	EMAP
TuleCree_164_W	Tule Creek	48.18355	-105.49147	5683TU01	Legacy STORET, or Modern STORET under MONT-DEQ
waldrn99_270_C	waldrn	47.9193	-112.817	EPA01-421	USU
WaldronC_117_C	Waldron Creek	47.92	-112.8339	5326WA01	Legacy STORET, or Modern STORET under MONT-DEQ
WaldronC_117_C	Waldron Creek	47.92	-112.8339	WMTP99-R020	EMAP
WESTFORK_099_W	WEST FORK POPLAR RIVER NEAR FOUR BUTTES	48.6225	-105.6525	6181WE01	Legacy STORET, or Modern STORET under MONT-DEQ
WESTFORK_118_C	WEST FORK STILLWATER CUS001 ABOVE ADIT	45.3981	-109.9683	BKK160	Modern STORET, under MT-DEQ
WESTFORK_118_C	WEST FORK STILLWATER CUS001 ABOVE ADIT	45.3981	-109.9683	2349WE03	Legacy STORET, or Modern STORET under MONT-DEQ
WESTFORK_127_W	WEST FORK POPLAR RIVER AT BRIDGE ON COUNT	48.9442	-106.2503	6577WE01	Legacy STORET, or Modern STORET under MONT-DEQ
WESTFORK_128_W	WEST FORK POPLAR RIVER NEAR RICHLAND MONT	48.8081	-106.0206	6379WE01	Legacy STORET, or Modern STORET under MONT-DEQ
WESTFORK_129_W	WEST FORK POPLAR RIVER S OF PEERLESS	48.7478	-105.9286	6379WE02	Legacy STORET, or Modern STORET under MONT-DEQ
WFkBeave_095_C	W. Fk. Beaver Creek	44.905	-111.3725	1738WE01	Legacy STORET, or Modern STORET under MONT-DEQ
WFkPopla_126_W	W. Fk. Poplar River	48.69695	-105.8319	REFWFPR	Modern STORET, under MT-DEQ
wfkstl99_274_C	wfkstl	45.3988	-109.961	EPA01-430	USU
Whitetai_102_W	Whitetail Creek	46.9947	-111.4858	MT222052	EMAP
Whitewat_170_W	Whitewater Creek	48.95661	-107.85937	06156000	NWIS
WillowCr_103_C	Willow Cr (I)	45.4481	-112.8278	06025800	NWIS
WillowCr_103_C	Willow Cr (I)	45.4481	-112.8278	M03WILOC01	Modern STORET, under MT-DEQ
WillowCr_104_C	Willow Cr (II)	45.4381	-112.7422	M03WILOC02	Modern STORET, under MT-DEQ
WolfCree_130_W	Wolf Creek @ Wolf Pt.	48.08778	-105.6781	REFWC	Modern STORET, under MT-DEQ
WoodyIsl_174_W	Woody Island Coulee	48.92265	-108.37948	6560WO01	Legacy STORET, or Modern STORET under MONT-DEQ
WoodyIsl_174_W	Woody Island Coulee	48.92265	-108.37948	BKK165	Modern STORET, under MT-DEQ
WyomingC_107_C	Wyoming Creek	45.05361	-109.4069	WMTP99-R036	EMAP
WyomingC_107_C	Wyoming Creek	45.05361	-109.4069	Y05WYOMC01	Modern STORET, under MT-DEQ

Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers: Update 1

May 2013

Prepared by:
Water Quality Planning Bureau, Standards Section
Montana Department of Environmental Quality
Water Quality Planning Bureau
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

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[1] Environmental Science Specialist, Montana Department of Environmental Quality, 1520 East 6th Ave, Helena, MT, 59620, msuplee@mt.gov;
[2] Professor, Environmental Studies Program, University of Montana, Missoula, MT, 59812.

EXECUTIVE SUMMARY

This document is the first update by the Department of Environmental Quality (Department) to the numeric nutrient criteria recommendations it made in 2008. The science of eutrophication in general and numeric nutrient criteria in particular has continued to advance in the interim years. In addition, the Department has modified somewhat the process by which numeric criteria are derived. In 2008, the Department used ecoregions, stressor-response studies (nutrient as stressor, impact to stream beneficial use as response), and data from reference streams to develop criteria. Various cases studies had established a linkage between nutrient concentrations observed in reference streams and harm to beneficial uses; on average, harm-to-use occurred at the equivalent of the 86th percentile of the nutrient reference distribution. In the 2008 document the Department relied heavily on two percentiles from the ecoregional reference distributions, namely the 75th and 90th, to derive criteria for each ecoregion.

The approach taken in 2008 had its shortcomings, however. In some ecoregions the method resulted in criterion concentrations which other data and studies have shown were unnecessarily stringent, while in other ecoregions the method resulted in criteria at concentrations that were too high (not protective). The method— albeit simple, consistent, and transparent—limited the Department's ability to derive best-fit criteria for each ecoregion. Fundamentally, the Department considers the combined use of ecoregions, stressor-response studies, and reference data to be a sound approach (as did the external peer reviewers of the draft of this document). But compared to 2008, more stressor-response studies are now available and these can better inform the criteria derivation process. As a result, in this update there has been less reliance on specific reference-distribution percentiles and much more reliance on regional as well as non-regional stressor-response studies.

The other major change, relative to the 2008 document, is that the Department will not be recommending nitrate (or nitrate + nitrite) criteria for adoption for the control of eutrophication at this time. Only total phosphorus (TP) and total nitrogen (TN) criteria are provided here. Rapid uptake of soluble nitrogen compounds by aquatic organisms (mainly algae and plants) makes these compounds' concentrations quite variable, and difficult to use as ambient surface water criteria. Total nitrogen and TP provide better overall correlation to eutrophication response than soluble nutrients, and are generally more practical than soluble forms for ambient river monitoring and assessment, total maximum daily loads, etc.

The Department is recommending both TN and TP criteria for stream protection. Phosphorus (P) control is sometimes promoted as the only approach needed to limit eutrophication, this being based largely on the more economical removal of P from wastewater and the assumption that P can be made to become limiting in the waterbody. But data pertaining to streams and rivers indicate that it would be unwise to adopt only P criteria. Mixed assemblages of benthic algae are very often limited by nitrogen or nitrogen and phosphorus (co-limitation) in the region's flowing waters. A P-only approach, in order to work, would require that P standards be set to the very low background levels observed in our western region's reference sites (e.g., 10 µg TP/L). If the P standard were not set to natural background, and no controls on N were undertaken, then the commonly occurring N limitation or N and P co-limitation would lead to algal growth stimulation nonetheless. Worse yet, in the long term, a P-only strategy would result in highly skewed (elevated) N:P ratios accompanying the low P levels. These management-induced conditions might control green algae biomass but may lead to nuisance blooms of the diatom algae *Didymosphenia geminata,* which has in recent years formed nuisance blooms in rivers and streams in Montana and word-wide.

A balanced and prudent policy would be to reduce both N and P and maintain, as nutrient concentration reductions occur, a roughly balanced (i.e., Redfield) ratio between the two. This is the strategy that has been applied on the Clark Fork River (where nutrients standards were adopted in 2002) and it appears to be working there. In addition, other researchers in the field are recommending that both N and P need to be controlled to effectively manage eutrophication. Thus, both N and P criteria for wadeable streams and rivers are proposed in this document.

The document has been organized so that readers can quickly locate key information pertaining to an ecoregion of interest. Data and discussion specific to each ecoregion are then presented on three to four pages. A map of Montana showing the ecoregion in which the criteria apply is shown first, followed by the criteria recommendations, and then tables of descriptive statistics for the ecoregion's reference streams. Then readers will find TN and TP histograms for the reference data, a discussion of the scientific studies (regional and beyond) that were used to help derive the criteria, other considerations pertaining to the derivation of the criteria, and a conclusion with final thoughts about the criteria.

The Department recognizes that within each ecoregional zone there are likely to be some streams with unique characteristics that could render the ecoregional criteria inappropriate. These characteristics include, for example, the presence of a large dam-regulated lake or reservoir upstream, or the upstream influence of a level-IV ecoregion known to have naturally elevated TP concentrations. A few cases have already been identified, and reach-specific criteria for them are presented and discussed in the document.

Below are summarized the criteria concentrations that have been recommended (**Table ES-1**). As was the case in the 2008 document, the criteria should generally apply seasonally.

Table ES-1. Recommended Numeric Nutrient Criteria for Different Montana Ecoregions and Stream Reaches.
Related assessment information is also shown.

		Parameter		
Ecoregion (level III or IV) and number, or Reach Description	**Period When Criteria Apply**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**	**Related Assessment Information***
Northern Rockies (15)	July 1 to September 30	25	275	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Canadian Rockies (41)	July 1 to September 30	25	325	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Idaho Batholith (16)	July 1 to September 30	25	275	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Middle Rockies (17)	July 1 to September 30	30	300	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Absaroka-Gallatin Volcanic Mountains (17i)	July 1 to September 30	105	250	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Northwestern Glaciated Plains (42)	June 16 to September 30	110	1300	
Sweetgrass Upland (42l), Milk River Pothole Upland (42n), Rocky Mountain Front Foothill Potholes (42q), and Foothill Grassland (42r)	July 1 to September 30	80	560	165 mg Chl*a*/m^2 and 70 g AFDM/m^2

Table ES-1. Recommended Numeric Nutrient Criteria for Different Montana Ecoregions and Stream Reaches.
Related assessment information is also shown.

		Parameter		
Ecoregion (level III or IV) and number, or Reach Description	**Period When Criteria Apply**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**	**Related Assessment Information***
Northwestern Great Plains (43) and Wyoming Basin (18)	July 1 to September 30	150	1300	
River Breaks (43c)	NONE RECOMMENDED	NONE RECOMMENDED	NONE RECOMMENDED	
Non-calcareous Foothill Grassland (43s), Shields-Smith Valleys (43t), Limy Foothill Grassland (43u), Pryor-Bighorn Foothills (43v), and Unglaciated Montana High Plains (43o)†	July 1 to September 30	33	440	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
INDIVIDUAL REACHES:				
Flint Creek, from Georgetown Lake outlet to the ecoregion 17ak boundary (46.4002, -113.3055)	July 1 to September 30	72	500	150 mg Chl*a*/m^2 and 45 g AFDM/m^2
Bozeman Creek, from headwaters to Forest Service Boundary (45.5833, -111.0184)	July 1 to September 30	105	250	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Bozeman Creek, from Forest Service Boundary (45.5833, -111.0184) to mouth at East Gallatin River	July 1 to September 30	76	270	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Hyalite Creek, from headwaters to Forest Service Boundary (45.5833,-111.0835)	July 1 to September 30	105	250	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Hyalite Creek, from Forest Service Boundary (45.5833,-111.0835) to mouth at East Gallatin River	July 1 to September 30	90	260	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
East Gallatin River between Bozeman Creek and Bridger Creek confluences	July 1 to September 30	50	290	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
East Gallatin River between Bridger Creek and Hyalite Creek confluences	July 1 to September 30	40	300	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
East Gallatin River between Hyalite Creek and Smith Creek confluences	July 1 to September 30	60	290	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
East Gallatin River from Smith Creek confluence to the mouth (Gallatin River)	July 1 to September 30	40	300	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
*Benthic algae density.				

† For the Unglaciated High Plains ecoregion (43o), criteria only apply to the polygon located just south of Great Falls, MT.

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ACRONYMS

Acronym	Definition
AFDM	Ash Free Dry Mass
ARM	Administrative Rules of Montana
Chl*a*	Chlorophyll-*a*
DEQ	Department of Environmental Quality (Montana)
DO	Dissolved Oxygen
HBI	Hilsenhoff Biotic Metric
NB	Natural Background
SOP	Standard Operating Procedure
SRP	Soluble Reactive Phosphate
TDP	Total Dissolved Phosphate
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
USGS	United States Geological Survey
WMA	Wildlife Management Area
WWTP	Wastewater Treatment Plant

1.0 INTRODUCTION

This is the first update to the Department of Environmental Quality (Department) document "Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers" (Suplee et al., 2008). Suplee et al. (2008) addresses methods that were used to derive numeric nutrient (nitrogen and phosphorus) criteria. The science of eutrophication in general and numeric nutrient criteria in particular has continued to advance in the interim years. Thus, this update reflects the most up-to-date nitrogen and phosphorus criteria recommendations for the control of eutrophication in wadeable streams and rivers that the Department has so far provided. With these revisions to the nutrient criteria, it bears repeating that the purpose of water quality criteria and standards is to define a level of a pollutant that will protect beneficial uses. This is the level to which degraded streams need to be restored; streams with water quality better than the criteria are addressed by the state's nondegradation provisions (i.e. ARM 17.30.701 through 17.30.718).

In the 2008 document, the Department used ecoregions (Woods et al., 2002), regional stressor-response studies (nutrient as stressor, impact to stream beneficial use as response), and data from reference streams to derive the criteria. Ecoregions were used to segregate the landscape into zones within which single nitrogen and phosphorus criteria—protective of the streams' beneficial uses and unique to each ecoregion—were recommended. Linkages had been made between harm to beneficial uses and nutrient concentrations which occurred, on average, at the 86th percentile of reference, with a coefficient of variation of ± 13% (i.e., from the 73rd to the 99th percentile; (Suplee et al., 2007). In developing its 2008 criteria recommendations, the Department relied heavily on two percentiles from the ecoregional reference distributions, namely the 75th and 90th (Suplee et al., 2008).

In 2008 the Department used only two different reference percentiles to derive the criteria because there were fewer regional dose-response studies available then. Further, the Department believed that it was best to be consistent in the use of reference-percentiles across broad areas of the landscape, because it would be fair and transparent. However in retrospect this approach had its failings, because in some ecoregions (e.g., the Canadian Rockies) the method resulted in criterion concentrations (6 µg TP/L) which other data and studies have shown to be unnecessarily stringent, while in other ecoregions (e.g., the Middle Rockies) the approach produced criteria concentrations (48 µg TP/L) we now believe to be somewhat too high. The original approach limited the Department's ability to recommend custom-fit criteria for different ecoregions that best reflect the level of water quality needed to protect the beneficial uses of each particular region's streams.

The Department still considers the combined use of ecoregions, stressor-response studies, and reference data to be a sound approach, but it was in need of modification. New stressor-response studies are now available and we believe these can better inform the criteria derivation process. In this update, which documents the Department's revised methods and recommended criteria, there will be less reliance on specified reference-distribution percentiles and much more reliance on regional as well as non-regional stressor-response studies. For clarity, we contrast below the approach taken in 2008 (**Figure 1-1**) vs. the approach taken in this document (**Figure 1-2**).

Another concern pertaining to the earlier work was the degree to which all reference sites within an ecoregion were equitably represented. In some ecoregions, a great deal of data has been collected at one or two reference sites and much less data at other sites. In this updated work, we improved objectivity by using two quantitative methods to assure that each reference site in an ecoregion only contributes a comparable amount of information to the ecoregional dataset; this is reflected in box 4 of **Figure 1-2** and is detailed in **Section 2.4**.

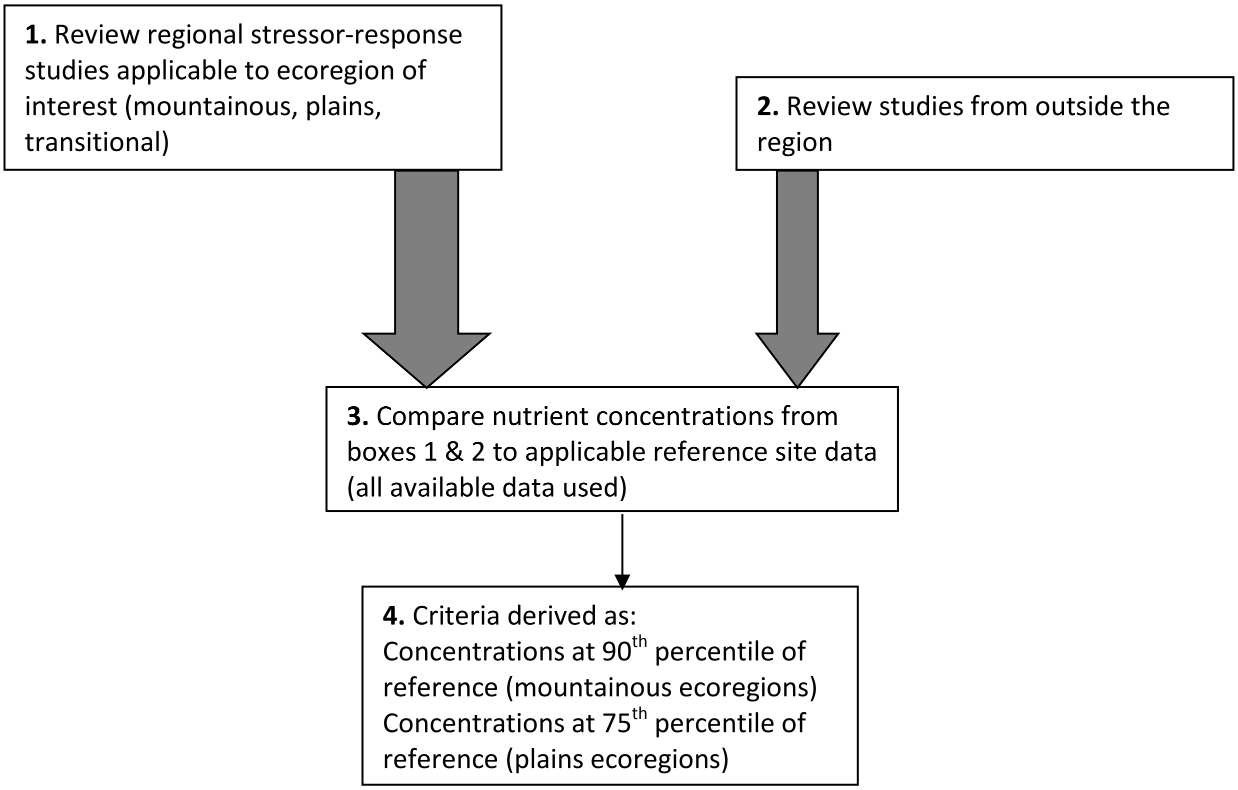

Figure 1-1. Overview of approach used to derive nutrient criteria in 2008 (Suplee et al., 2008).
The size of the large grey arrows near the top of the figure represent the relative importance of the two information sources for deriving regional nutrient criteria.

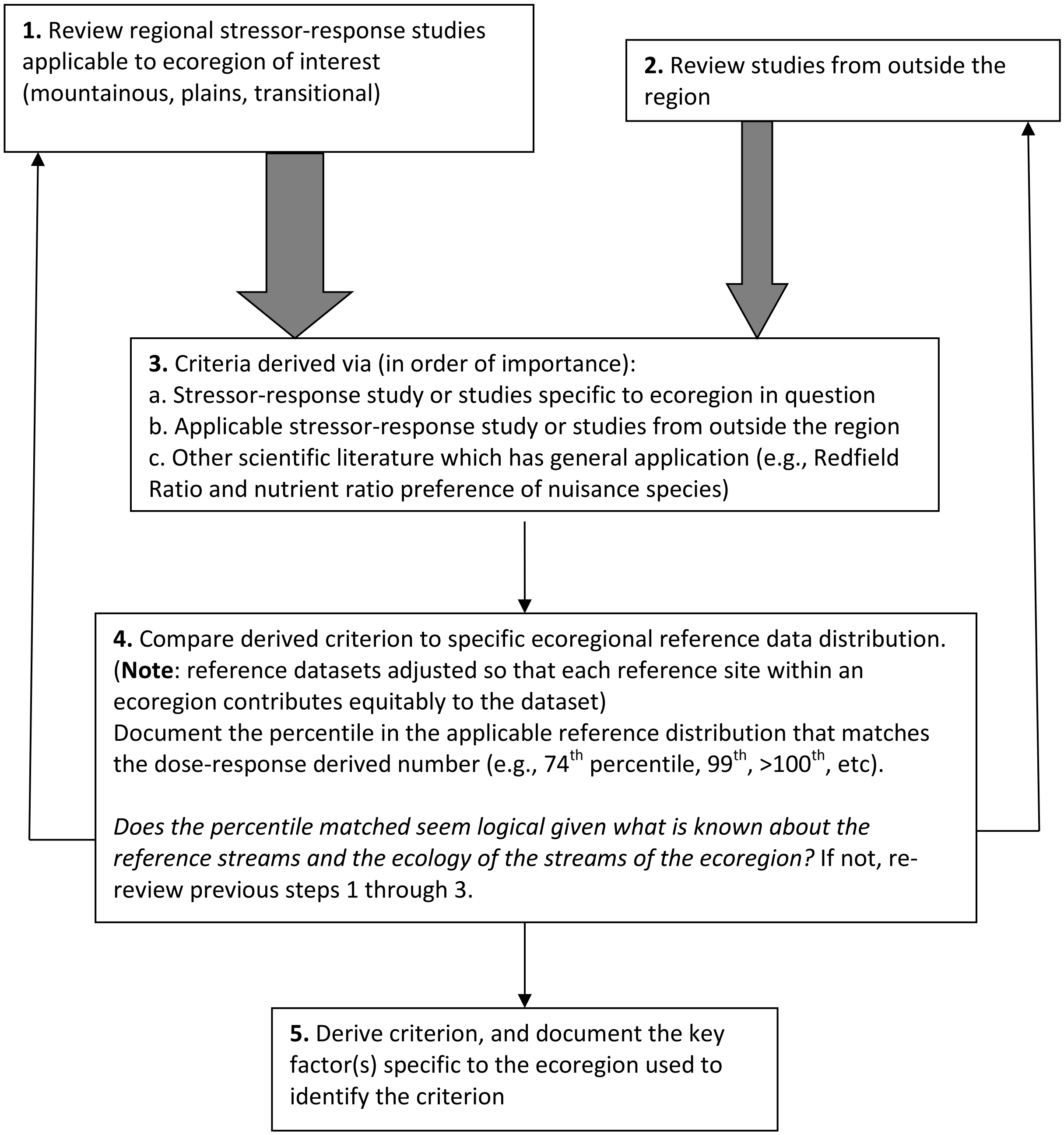

Figure 1-2. Overview of approach used to derive nutrient criteria in this document.
The size of the large grey arrows near the top of the figure represent the relative importance of the two information sources for deriving regional nutrient criteria.

The other major change, relative to the 2008 document, is that the Department will not be recommending nitrate (or nitrate + nitrite) criteria for adoption for the control of eutrophication at this time. Only total phosphorus (TP) and total nitrogen (TN) criteria are provided here. Rapid uptake of soluble nitrogen compounds by aquatic organisms (mainly algae and plants) makes these compounds' concentrations highly variable, and difficult to use as ambient surface water criteria. Total nitrogen and TP have been shown to provide better overall correlation to eutrophication response than soluble nutrients (Dodds et al., 1997; Dodds et al., 2006; Dodds et al., 2002) and, in terms of water quality criteria, total nutrients are more practical than soluble forms for river monitoring and assessment, total maximum daily loads, etc. (Dodd and Welch, 2000). **However**, the Department strongly encourages the collection of nitrate + nitrite when collecting TN and TP data. The soluble data can often point to specific types of nutrient sources, for example. The Department's Water Quality Monitoring Section will continue to include nitrate + nitrite alongside TN and TP for routine monitoring for nutrients and may use some general guidelines from the scientific literature for determining when measured concentrations are clearly too high.

2.0 METHODS USED TO DERIVE THE CRITERIA

In the **Introduction** we presented a general overview of the updated process used to derive the numeric nutrient criteria (**Figure 1-2**). In this section, we delve further into the details of these approaches.

2.1 ECOREGIONS AS THE BASIS FOR ESTABLISHING NUTRIENT CRITERIA ZONES

The Department tested the usefulness of ecoregions (Omernik, 1987) as a means to establish nutrient criteria zones; that work is detailed in Varghese and Cleland (2005) and Section 4.0 of Suplee et al. (2008). The Department will continue to use ecoregions as the basis for establishing nutrient criteria zones. Subsequent analysis has further verified that specific level IV (small scale) ecoregions are significantly different from the larger-scale level III ecoregions in which they reside (Varghese and Cleland, 2008; Varghese and Cleland, 2009). In **Section 3.0** of this document we will detail the criteria derived for individual ecoregions at the level III or level IV scale. In general, a level IV ecoregion will only be broken out for nutrient criteria derivation **if** (1) natural concentrations of nutrients in the level IV ecoregion are elevated above concentrations identified as harming uses per the stressor-response studies pertaining to that region, **or** (2) it is a level IV ecoregion that resides along the Rocky Mountain front (or similar environments) and represents a zone containing mountain-to-prairie transitional streams. In some cases the effect of a particular level IV ecoregion will influence natural nutrient concentrations in downstream waterbodies outside of the boundaries of the level IV. In **Section 4.0** a method to account for this type of influence is detailed and a number of reach-specific criteria are recommended there.

2.2 CRITERIA IN THIS DOCUMENT APPLY TO WADEABLE STREAMS

The scope of the criteria in this document is wadeable streams. The only substantive change since 2008 pertaining to this topic is the definition of specific rivers and river segments which are not wadeable (i.e., the large rivers). Flynn and Suplee (2010) use a wadeability index (product of river depth [in feet] and mean velocity [in ft/sec]) of 7.2 to segregate wadeable from non-wadeable rivers. During summer base flow these large rivers have mean water depths in excess of 3.15 ft and discharges of 1,500 ft^3/sec or greater. In Montana, rivers with these characteristics are almost always 7^{th} order or higher (Strahler, 1964), and this is consistent with earlier definitions of large rivers based on stream order (Welcomme, 1985). **Table 2-1** shows the non-wadeable large rivers of the state to which the criteria in this document ***do not*** apply. The Department is primarily using process-based mechanistic water quality models to identify criteria for large river segments.

Table 2-1. Large river segments within the state of Montana.

River Name	Segment Description
Big Horn River	Yellowtail Dam to mouth
Clark Fork River	Bitterroot River to state-line
Flathead River	Origin to mouth
Kootenai River	Libby Dam to state-line
Madison River	Ennis Lake to mouth
Missouri River	Origin to state-line
South Fork Flathead River	Hungry Horse Dam to mouth
Yellowstone River	State-line to state-line

2.3 Criteria Apply Seasonally, With Exceptions

As before, we recommend that the numeric nutrient criteria for wadeable streams and rivers apply seasonally, during that period when algae growth is peak and ensuing water quality impacts are maximal (i.e., the "Growing Season"). See **Table 2-2** below. For monitoring and assessment purposes, however, a ten day window (plus/minus) on the Growing Season start and end dates is acceptable, in order to accommodate year-specific conditions (e.g., an early-ending spring runoff). Best professional judgment is required to decide if early or later sampling is warranted.

Table 2-2. Start and Ending Dates for Three Seasons (Winter, Runoff and Growing), by Level III Ecoregion.

Ecoregion Name	Start of Winter	End of Winter	Start of Runoff	End of Runoff	Start of Growing Season	End of Growing Season
Canadian Rockies	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Northern Rockies	Oct.1	March 31	April 1	June 30	July 1	Sept. 30
Idaho Batholith	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Middle Rockies	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Northwestern Glaciated Plains	Oct.1	March 14	March 15	June 15	June 16	Sept. 30
Northwestern Great Plains	Oct.1	Feb. 29	March 1	June 30	July 1	Sept. 30
Wyoming Basin	Oct.1	April 14	April 15	June 30	July 1	Sept. 30

Exceptions to the seasonal applicability of nutrient standards will occur when it is known or demonstrated that a stream or river is having a significant influence on a downstream lentic waterbody (lake, reservoir). In such cases, criteria (and nutrient loads) applicable to the lake may apply to a stream draining to the lake, and would apply year round. These situations need to be determined case-by-case, and are beyond the scope of this document.

2.4 Method for Assuring that All Reference Sites are Equitably Represented in an Ecoregion (The All-observations Dataset after applying Brillouin Evenness Index, and the Median dataset)

Assuring that each reference site contributes an approximately equal number of N and P observations to each ecoregional zone has been a Department objective for some years (see Section 6.2.1 of Suplee, et al. (2008)). Since Suplee et al. (2008) was released, the Department continued to target under-represented reference sites and collected data in summer 2009 and summer 2010.

In spite of the targeted field work, there was still a fair amount of inequality in terms of the number of nutrient observations per site in each ecoregion. Therefore, we undertook two different methods in the office using the updated (current through 2010) reference nutrient dataset. In method one we used (as in 2008) the Brillouin evenness index (Pielou, 1966; Zarr, 1999). This method assures that each reference site in an ecoregion contributes equal amounts of information to the nutrient dataset. Distribution statistics (e.g., maximum, median, 75th percentile) are calculated from the resulting dataset, and these statistics provide a means for readers to see the full range of nutrient concentrations that have been observed across reference sites during the growing season. The method assumes each observation from a reference site is independent of the others. Nutrient samples are collected from reference sites in a way intended to maximize independence, and a number of tested case studies show independence is usually maintained (Suplee and Sada de Suplee, 2011). In this document datasets so handled are called "all-observations after applying Brillouin Evenness Index", or simply the "all observations dataset".

In method two, all the growing-season observations from a reference site within an ecoregion are first reduced to a median. Distribution statistics are then calculated on the population of site medians; these are the "median datasets" in this document. Method two addresses the potential for intra-site temporal pseudoreplication (Hurlbert, 1984), but the output masks the full range of nutrient concentrations that have been observed across the ecoregion's reference sites. For this reason only the interquartile range and the 90th percentile are reported for the median datasets.

Because method one is computationally involved, it is detailed here. The Brillouin evenness index (J) for a whole population (Pielou, 1966; Zarr, 1999) is:

$$\mathbf{J} = H \div H_{max}$$

with

$$H = \frac{(\log n! - \sum \log f_i!)}{n}$$

and

$$H_{max} = \frac{\log n! - (k - d) \log c! - d \log (c + 1)}{n}$$

where n is the total number of reference nutrient observations (e.g., TP) in the ecoregion, f_i is the frequency of nutrient observations specific to each reference site in the ecoregion, k is the number of reference sites in the ecoregion, c is the integer portion of n/k and d is the remainder. The index value J will range from zero to one (one being the case where each reference site has been sampled for nutrients exactly the same number of times).

We wanted to achieve an evenness of 90% or better for each ecoregional reference dataset. Applying these equations with a target J value of ≥0.9 required in some cases that a proportion of observations from heavily-sampled reference sites be excluded from use. This was carried out objectively and independently for TN and for TP, as follows. First, the J value was calculated using all data for a given nutrient (e.g., TP) from all reference sites within the ecoregion in question. If the value was ≥0.9, nothing further was done and all the data were used as-is for descriptive statistics. If the value was <0.9, we identified the over-contributing reference sites and calculated how many observations would need to be removed from each in order to achieve a J value of 0.9. Because J measures evenness, reducing many observations from a single over-contributing site was not effective. Instead, a smaller number of observations had to be eliminated from each of the major over-contributors. Once the number of observations to be eliminated from each over-contributing reference site was known, we randomly removed that number of observations from the dataset of each of the specified sites. Finally, the now 'more even' ecoregional dataset (comprising the remaining observations from sites where the random-elimination process was applied plus the observations from the sites where no censoring was applied) was used to generate descriptive statistics.

In **Sections 3.0** and **4.0** we present distribution statistics (e.g., 25th, 75th percentiles) for data processed by both methods one and two. The results are sometimes different for each, and this is a function of the way the data were processed. It is important to note that no inferential statistics were carried out nor are the recommended criteria tied to any specific percentile; the distributional qualities of the reference data are being provided primarily as a means for readers to compare the recommended criteria to regional reference data. It should also be noted that in some level IV (small scale) ecoregions, there were only a few references sites and the number of collected nutrient samples was correspondingly low. We wanted to maintain a sample-size minimum of about 12 (Varghese and Cleland, 2008, Appendix H) for these ecoregions in order to sufficiently characterize the reference condition. Therefore, in level IVs that were near to this sample-size minimum, no sample-size reductions using the Brillouin evenness index were undertaken.

2.5 Literature Consulted

A re-review of the relevant scientific literature cited in Suplee et al. (2008) was undertaken, as well as a search and review of various studies and reports that have been released before and since 2008. The Department completed a whole-stream nitrogen and phosphorus addition study between 2009 and 2011 (Montana Department Environmental Quality, 2009). Findings from that study are incorporated into this work as well. The Department also completed a mechanistic water quality model (QUAL2K) for the lower Yellowstone River and has recommended criteria for that waterbody using the model (Flynn and Suplee, 2013). Although the later work pertains to large rivers, findings from it help define the range of nutrient criteria one might expect for flowing waters of Montana.

Details on the specific literature that was most useful within each ecoregion will be provided in **Section 3.0.**

2.5.1 Literature Pertaining to Nutrient Enhancements in Rivers and Streams

Much of the pertinent scientific literature of the past few decades focuses on the effects of nitrogen (N) and phosphorus (P) over-enrichment. However, there is a smaller but equally valuable body of scientific literature addressing intentional nutrient *additions* to rivers and streams; these actions have usually been carried out for the purpose of enhancing depleted fisheries production (Holderman et al., 2009; Stockner, 2003). As we pointed out in Section 1.3 of Suplee et. al (2008), N and P have an interesting duality in that too much is a problem (cultural eutrophication), but too little can also be a problem (cultural oligotrophication). Thus, the nutrient-addition literature enabled us to have a better understanding of the ecology of nutrient-poor rivers and streams and how that ecology shifts as nutrients increase towards the concentrations that ultimately become "too much of a good thing". Many of the nutrient-addition studies were carried out in the Pacific Northwest in streams and small rivers similar to those found in western Montana (Perrin et al., 1987; Johnston et al., 1990; KOHLER et al., 2008; Perrin and Richardson, 1997; Stockner and Shortreed, 1978). Because salmon die after spawning in the upper tributaries of rivers draining to the Pacific, large quantities of marine-sourced nutrients are relocated to the streams annually. But overfishing, dams, and habitat destruction have greatly reduced many salmon runs, leaving the streams stripped of their annual nutrient source. To boost survival of the few fry and fingerlings that are spawned, nutrient additions have been undertaken by resource managers. These nutrient additions enhance algal growth and secondary production (aquatic insects), which in turn provide a larger food source for the fish which enhances their growth and survival (Stockner and Ashley, 2003). This type of work has included large-scale nutrient additions to the Kootenai River as it flows out of Montana into Idaho. Ambient nutrients in the Kootenai River were greatly reduced after the completion of Libby dam in the early 1970s (Holderman et al., 2009).

The streams to which nutrients are added for fisheries enhancement have very low ambient nutrient levels (ca. 5-10 µg TP/L and < 15 µg NO_3-N/L), and nutrient concentrations are only increased by a few additional micrograms per liter. The studies reviewed provided good incite on the ecological changes that occur in low-nutrient streams once nutrients increased, and how salmonid fisheries react to these small nutrient increases.

2.6 Both Nitrogen and Phosphorus Criteria are Recommended

The concept of nutrient limitation is important in the development of N and P criteria. Relative to N and P, limitation can be defined in a negative sense; a nutrient is *not* limiting if, when increased, one does not observe an effect on plant or algal growth (Gibson, 1971). The scientific literature has many examples of studies and analyses showing that N, or P, or commonly both stimulate algal production in surface waters (Francoeur, 2001; Smith et al., 1999; Tank and Dodds, 2003; Elser et al., 1990; Elser et al., 2007; Lewis et al., 2011). Co-limitation appears to be especially common in flowing waters , where nutrient-addition experiments show that added N and P result in much greater response of algal growth than does N- or P-addition alone (Elser et al., 2007). Regional work using nutrient diffusing substrates (N, P, and N+P) supports these findings (Mebane et al., 2009). Mebane et al.'s experiments were carried out *in situ* in intermontane wadeable streams of Idaho which are comparable to Montana's western streams. Background N and P concentrations in the streams ranged from very low (7 µg TP/L and 50 µg TN/L) to quite elevated (e.g., 91 µg TP/L and 1,820 µg TN/L). Based on the growth of algae on the nutrient diffusers that developed over 21 days, N and P co-limitation was indicated in three streams, N limitation was shown in two streams, and P limitation was found in one stream; one stream with highly elevated ambient nutrients showed no limitation (Mebane et al., 2009). And it should be noted, especially in light of the definition of nutrient limitation given above, that in most of the streams the greatest algal biomass developed on the N+P diffusers (Mebane et al., 2009).

Liebig's Law of the Minimum (Hooker, 1917) is a well-established tenet in the agricultural sciences that states that biomass yield for a particular plant is usually limited by the nutrient that is present in the environment in the least quantity relative to the plant's need for that nutrient to support growth. The law is sometimes used to rationalize the idea that, in most cases, only P needs to be reduced to low concentrations to achieve eutrophication control in freshwaters. But Liebig's Law best applies to single plant species at a given place at a certain time, whereas the numeric nutrient criteria in this document apply to a mixed flora in flowing waters over several months of growing season. These flowing waters receive variable N and P loads over time and are home to mixed populations of algae species—and each species has somewhat different N and P requirements and capability of taking up nutrients (Hecky and Kilham, 1988; Borchardt, 1996). Nutrient limitation of the aggregate algal community is largely a function of the nutrient limitation of the dominant species, but shifting nutrient availability can change the dominant species and potentially the limiting nutrient.

Streams are variable environments where, for example, N and P availability can alternate as a function of stream discharge (Hullar and Vestal, 1989). Stated simply, limiting nutrient levels are not fixed and both nutrients are likely to limit some facet of the algal community at any point in time. If for example P is presently limiting in a stream, that does not mean there is no point in limiting N. If P were to increase, say from summer rain events, or due to the confluence of a downstream tributary with slightly higher P concentrations, the N that was formerly in excess can become the limiting nutrient without any change in its absolute concentration (Gibson, 1971). Similarly, an algal community may be N-limited early in summer, and as surface flows drop, proportionally more N-rich groundwater enters the stream, shifting the community structure and switching the stream to P-limitation. Results from twelve years of monitoring on the Clark Fork River in Montana support the idea that it is best to control both N and P. There, in river locations where both the N standard and the P standard were met (20 µg TP/L and 300 µg

TN/L); algal biomass has usually been reduced to the standard (150 mg Chl*a*/m^2). Locations in the river where these nutrient levels have not been met continue to have elevated algae biomass, and study sites give mixed signals regarding nutrient limitation—some suggesting N limitation, others P; these signals are not consistent across time or location (Suplee et al., 2012).

Water quality standards based on control of only a single nutrient (i.e., P) could result in unwanted ecological consequences in Montana's rivers and streams. Background nutrient levels in our western reference streams are usually quite low (10-18 µg TP/L and 85-190 µg TN/L; Smith et al., 2003; Suplee et al., 2007), and usually have TN:TP ratios at or somewhat higher than Redfield (Redfield = 7:1 by mass; Redfield, 1958). The nuisance diatom alga *Didymosphenia geminata* has, in recent years, spread to and formed nuisance benthic blooms in low-nutrient rivers and streams worldwide (Whitton et al., 2009; Kilroy, 2011; Spaulding and Elwell, 2007). It is found in Montana and, in western U.S. states, probabilistic survey data show that in over half of streams containing *D. geminata* TP is <10 µg/L (Spaulding and Elwell, 2007). Others also report that *D. geminata* usually occurs in streams with very low P (Whitton et al., 2009; Kilroy and Bothwell, 2012), and that it tends to disappear when TP exceeds about 20 µg/L (Lovstad, 2008). Further, *D. geminata* generally thrives in waters where N:P ratios are high (34:1 on average) much of the time (Whitton et al., 2009). *Didymosphenia geminata* blooms in low-P streams are caused by the diatoms' elevated production of polysaccharide stalks which develop as a consequence of phosphorus limitation (Kilroy, 2011; Kilroy and Bothwell, 2012). Stalk production in attached diatoms is considered competitively advantageous because it elevates the cells towards higher light (Hudon and Bourget, 1981), and this also places them in closer contact with available nutrients in flowing water (Kilroy and Bothwell, 2012; Bothwell et al., 2012)[3].

Researchers suggest that an effective way to diminish *D. geminata* blooms is to encourage its algal competitors by assuring that a sufficient (though small) supply of soluble phosphorus is available (Whitton et al., 2009; Kilroy and Bothwell, 2012). Indeed, the Montana Department of Fish, Wildlife and Parks is currently planning low-level phosphate addition experiments in troughs alongside the Kootenai River (where *D. geminata* blooms have become quite severe) to see if the alga can be brought under control via nutrient management actions. The alga is believed to be impacting the salmonid fishery there, where the high algal density appears to be reducing abundance of key aquatic insects which salmonids prey upon (Jim Dunnigan, Fishery Biologist, MT Fish Wildlife and Park, personal communication March 14, 2012).

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[3] Alternative hypotheses exist regarding what controls *D. geminata* blooms in low-P streams. One of them states that the diatoms' polysaccharide stalks have an affinity for iron, which is absorbed to the stalks and in the process forms iron oxyhydroxide (Sundareshwar et al., 2011). Iron oxyhydroxides have a strong affinity for P and will adsorb (co-precipitate) it from the water (Mortimer, 1941; Caraco et al., 1989; Hasler and Einsele, 1948). Sundareshwar et al. (2011) posit that as the mat grows, anaerobic microbial decomposition (by iron- or sulfate-reducing bacteria) of dead diatoms within the mat leads to the reduction of the iron, formation of iron sulfides, and concomitant release of P. The abundant P is then available to the live diatoms at the mat surface, supporting further growth. The geochemical process they describe is well known in marine and freshwater systems (Suplee and Cotner, 2002). But the mechanism that Sundareshwar et al. (2011) propose is unsatisfactory, as it does not explain why the mats grow rapidly and develop to great size in low P streams in the first place. Sundareshwar et al. (2011) note that *D. geminata* produces high levels of alkaline phosphatase at the mat surface because the P sequestered there with iron is *not* bioavailable. Thus, we view their hypothesis as a potential mechanism for mat maintenance, but not necessarily for mat development. Others also find that this geochemical explanation does not jive with findings in *D. geminata* dominated streams, where increases in stream P concentrations lead to *declines* in *D. geminata* (Bothwell et al., 2012; Kilroy and Bothwell, 2012).

Phosphorus reduction is often promoted as the only eutrophication control approach needed to control eutrophication, this being based largely on the more economical removal of P from wastewater (Lewis and Wurtsbaugh, 2008), and the assumption that P can be made to become limiting in the waterbody, *sensu* Liebig's Law. But the facts given above indicate that it would be unwise to recommend only P standards for control of excess algal biomass in our streams and rivers. A P-only approach, in order to work, would require that P standards be set to the background levels observed in our western region's reference sites (e.g., 10 µg TP/L). Total phosphorus concentrations this low are hard to achieve technologically, but if the P standard was not set to this low natural background, then the commonly occurring N-limitation or N and P co-limitation would lead to algal growth stimulation nonetheless. Worse yet, in the long term, a P-only strategy would result in highly skewed (elevated) N:P ratios accompanying the low P levels. These management-induced conditions might control green algae biomass but may lead to nuisance blooms of *D. geminata.*

A balanced and prudent policy would be to reduce both N and P and maintain, as nutrient concentration reductions occur, a roughly balanced (i.e., Redfield) ratio between the two. This is the strategy that has been applied on the Clark Fork River and it appears to be working (Suplee et al., 2012). Other researchers in the field have recommended that both N and P need to be controlled to effectively manage eutrophication (Conley et al., 2009; Lewis et al., 2011; Paerl, 2009). Thus, we will generally be recommending both N and P criteria for wadeable streams and rivers in this document.

One final word on Redfield ratios. Studies of benthic algae show that it is necessary to move some distance above or below the Redfield ratio in order to be strongly convinced that a lotic waterbody is P or N limited (Dodds, 2003). When a benthic algal Redfield ratio (by mass) is <6, N limitation is suggested, and when it is >10 P limitation is indicated (Hillebrand and Sommer, 1999). Thus, there is a range of N:P values between about 6 and 10 where one can state, for practical purposes, that algal growth is co-limited by N and P.

3.0 Ecoregion-specific Numeric Nutrient Criteria Recommendations

In this section are documented the numeric nutrient criteria for each ecoregion. Ecoregional information is arranged as follows: (1) first the level III ecoregion is presented, and (2) if any level IV ecoregions within the level III need to be treated separately, their information follows in a subsection. The same presentation format is followed for each ecoregion, be it level III or level IV, to the degree possible. Data and discussion specific to each ecoregion is presented on three or four pages. A map of Montana showing the ecoregion in which the criteria apply is shown first, followed by criteria recommendations and tables of descriptive statistics for the reference sites in the ecoregion. **The Redfield ratio shown for the reference sites is based on the 50th percentile from the applicable median dataset.** Then readers will find: histograms of the reference data TN and TP distributions based on the all-observations dataset evened using the Brillouin index (in cases where the data were skewed to the right they have been $\log_{10}$ transformed); a discussion of the scientific studies (regional and beyond) that were used to help derive the criteria, any other considerations pertaining to the derivation of the criteria; and a conclusion summarizing final thoughts on the criteria.

Data from reference sites (Suplee et al., 2005) were important in the process of deriving the nutrient criteria (see **Figure 1-2**). **Figure 3-1** below is a statewide map showing the locations of all stream reference sites current through August 2011. There are currently 185 different sites in the network.

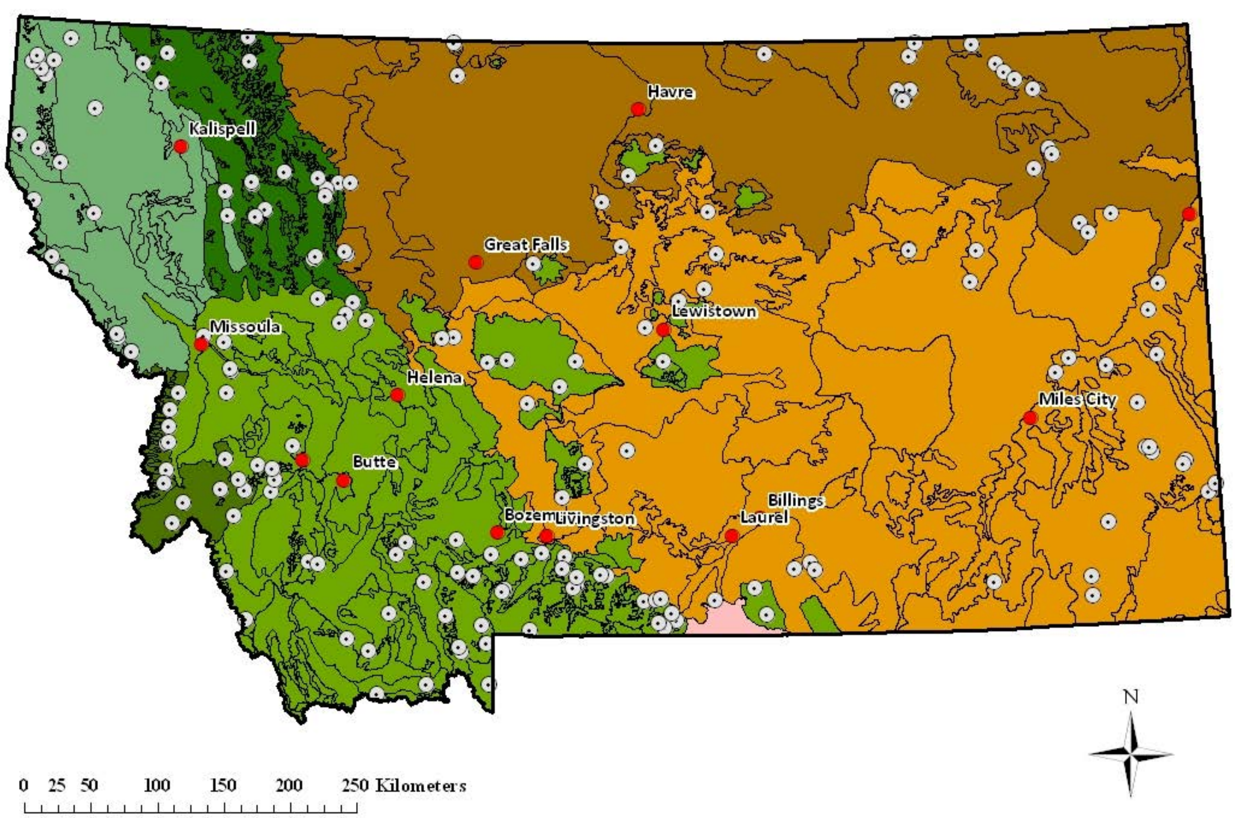

Figure 3-1. Map of Montana showing location of stream references sites (white dots). Colored regions denote level III ecoregions. Red dots show the major towns.

3.1 Level III: Middle Rockies (Ecoregion 17)

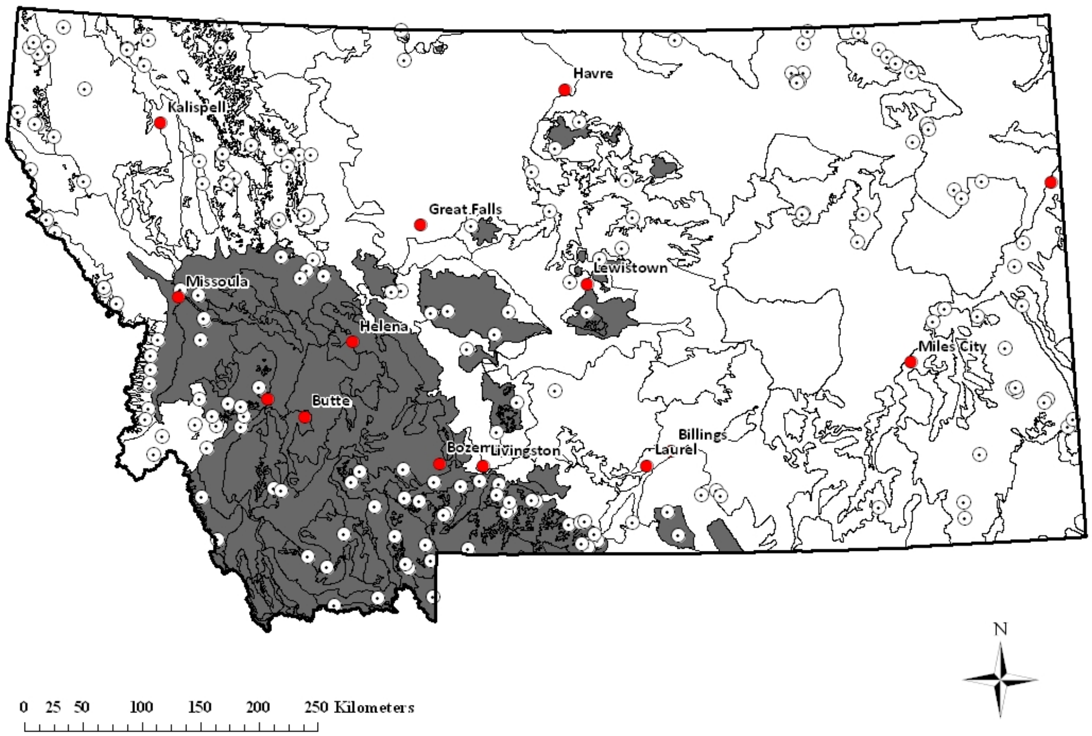

Figure 3-2. Map of Montana showing the Middle Rockies ecoregion in gray.
White dots are the reference sites.

Recommended Numeric Criteria

Total Phosphorus: **30 µg TP/L**
Total Nitrogen: **300 µg TN/L**

N:P Ratio of Criteria: **10:1**
N:P Ratio of Reference Sites: **11:1** (Redfield N:P ratio = 7:1)

Descriptive Statistics of Regional Reference Sites

Table 3-1A. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Middle Rockies ecoregion.
Data are from the all-observations dataset after applying Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	**Number of Reference Sites**	**Number of Samples**	**Min**	**Max**	**25th**	**(Median)50th**	**75th**	**90th**
TN	57	148	3	9580	55	95	141	220
TP	61	245	0.5	840	6	10	20	70

Table 3-1B. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Middle Rockies ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	**Number of Reference Sites**	**25th**	**(Median) 50th**	**75th**	**90th**
TN	57	51	90	136	181
TP	61	4	8	15	43

Criteria Match to Reference Distributions:
The 30 µg TP/L criterion matches to the 80th percentile of the all observations dataset and the 82nd percentile in the median dataset.

The 300 µg TN/L criterion matches to the 93th percentile of reference of the all observations dataset and the 98th percentile in the median dataset.

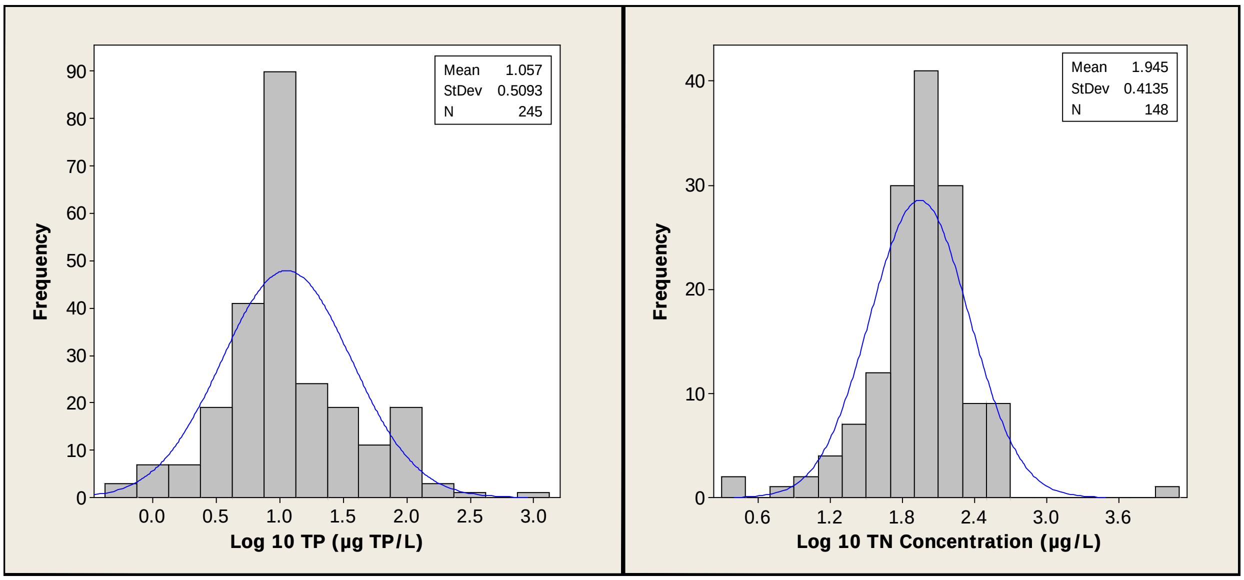

Figure 3-3. Nutrient concentrations from reference streams in the Middle Rockies ecoregion.
Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). Data were collected during the Growing Season (July 1-September 30).

Discussion of the Middle Rockies Ecoregion Nutrient Criteria
Two regional dose-response studies were available that relate TN and TP to stream impacts (Mebane et al., 2009; Suplee et al., 2012). Suplee et al. (2012) show that TP is saturated in the Clark Fork River at 24 µg/L. They also suggest that criteria for the Clark Fork River— upstream of the Flathead River confluence— be set uniformly to about 20 µg TP/L and 300 µg TN/L to meet the algae standard (150 mg Chl*a*/m^2 max). Further, they indicate that both TN and TP criteria should be met to achieve the intended reductions in algal biomass. Suplee et al. (2012) build on earlier work in which nutrient criteria were developed for the Clark Fork River (Dodds et al., 1997), and by doing so provide large-scale confirmation that the original criteria were largely correct. Mebane et al. (2009) carried out a study in southern Idaho. Many of the streams were intermontane and, thus, very similar to intermontane streams of this ecoregion. They recommend 40 µg TP/L and 600 µg TN/L in order to maintain benthic algae growth ≤150

mg Chl*a*/m^2, per Suplee et al. (2009). To maintain 125 mg Chl*a*/m^2, which is generally appropriate for shallow wadeable streams[4], the values would drop to about 35 µg TP/L and 475 µg TN/L.

Beyond the Middle Rockies ecoregion, studies in northern and southern temperate rivers and streams show that nutrient-benthic Chl*a* regressions have breakpoints at 27-62 µg/L for TP and between 367-602 µg/L for TN (Dodds et al., 2006; Dodds et al., 2002). What this indicates is that above the breakpoint concentrations, nutrients are saturated, and benthic algae control via nutrient control become ineffective. Stevenson et al. (2006) show in Michigan streams that the likelihood of reaching bottom coverage by *Cladophor*a of 20-40% increases sharply when TP exceed 30 µg/L and TN exceeds 1,000 µg/L. This level of streambed coverage by *Cladophora* was found to be unacceptable to the Montana public (see Suplee et al., 2009, Table 1). Chambers et al. (2011) derive nutrient criteria for Canadian streams using multiple methods including dose-response relationships between nutrients and algae and macroinvertebrate metrics. They recommend 20 µg TP/L and 210 µg TN/L for the Montane Cordillera, a mountainous region in British Columbia (actually, part of the Northern Rockies ecoregion). Equations relating benthic algal Chl*a* to total nutrients (Dodds et al., 1997, Equation 17; Dodds et al., 2006; Equation 19), were used to calculate TN levels that would maintain 125 mg Chl*a*/m^2 benthic algae given a TP of 30 µg TP/L. These equations resulted in TN concentrations ranging from 466-718 µg TN/L. If the algae level is set instead to 150 mg Chl*a*/m^2, and 30 µg TP/L is again used, the TN values range from 750 to 1,210 µg/L.

Conclusion

Studies that have the most specificity to the Middle Rockies suggest criteria ranging from 20-40 µg TP/L and 300-600 µg TN/L. Studies further afield provide a range of criteria to prevent nuisance algal growth or impacts to aquatic life communities ranging from 20-30 µg TP/L and 210-1,210 µg TN/L. **We recommend for this ecoregion 30 µg TP/L and 300 µg TN/L**. We recommend these values because: (1) these concentrations fall within the ranges provided in the studies, especially studies that are most pertinent to the ecoregion; (2) they maintain an N:P ratio of 10 which is close to the natural condition of regional reference sites (i.e., 11:1) and is fairly close to the upper band of the Redfield ratio and that indicates slight P limitation; and (3) they should generally encourage a balanced and diverse stream flora for this region by keeping nutrient ratios not far from Redfield and TP at a concentration which will help inhibit *Didymosphenia geminata* blooms.

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[4] A nutrient dose-response study carried out by the Department in southeastern Montana showed that in a wadeable stream benthic algae levels of 127 mg Chl*a*/m^2 (33 g AFDM/m^2) led to seasonal exceedances of the dissolved oxygen (DO) standard (Suplee and Sada de Suplee, 2011). Subsequent work—using a model based on Streeter-Phelps (1925) and cooler water temperatures more typical of mountainous streams—showed that DO exceedances would still occur in many (though not all) western MT streams. Therefore, when using Chl*a*-nutrient relationships from Mebane et al. (2009), Dodds et al. (1997; 2006), and others, we also used 125 mg Chl*a*/m^2 as the target algae level. The Department believes this value is a well-supported threshold, giving consideration to both the DO impacts observed in the dosing study and also the recreational threshold (and regarding the later, giving consideration to the known statistical patterns provided in the Department's SOP Chl*a* method). In this document we continue to use 150 mg Chl*a*/m^2 as well, which corresponds to the arithmetic mean of the replicates at the highest level of benthic algae found to be acceptable to the MT public (Suplee et al., 2009).

3.1.1 Level IV Ecoregion within the Middle Rockies: Absaroka-Gallatin Volcanic Mountains (17i)

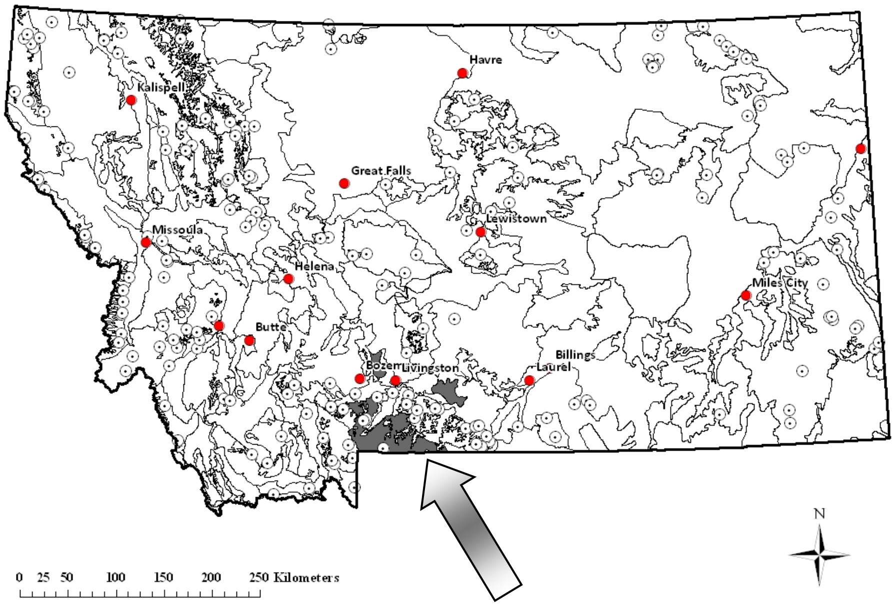


Figure 3-4. Map of Montana showing the Absaroka-Gallatin volcanic Mountains (17i), a level IV ecoregion within the Middle Rockies ecoregion.
White dots are the reference sites.

<u>Recommended Numeric Criteria</u>
Total Phosphorus: **105 µg TP/L**
Total Nitrogen: **250 µg TN/L**

N:P Ratio of Criteria: **2:1**
N:P Ratio of Reference Sites: **0.8:1** (Redfield N:P ratio = 7:1)

<u>Descriptive Statistics of Regional Reference Sites</u>

Table 3-2A. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Absaroka-Gallatin Volcanic Mountains (17i) ecoregion.
Data are from the all-observations dataset after applying the Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	**Number of Reference Sites**	**Number of Samples**	**Min**	**Max**	**25th**	**(Median)50th**	**75th**	**90th**
TN	4	13	7	181	52	80	100	163
TP	4	16	16	144	61	81	105	127

Table 3-2B. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Absaroka-Gallatin Volcanic Mountains level-IV ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	**Number of Reference Sites**	**25th**	**(Median) 50th**	**75th**	**90th**
TN	4	42	65	83	93
TP	4	62	77	90	106

The 105 µg TP/L criterion matches to the 75th percentile of reference in the all observations dataset and the 89th percentile of reference in the median dataset.

The 250 µg TN/L criterion is greater than the 100th percentile of reference in both datasets.

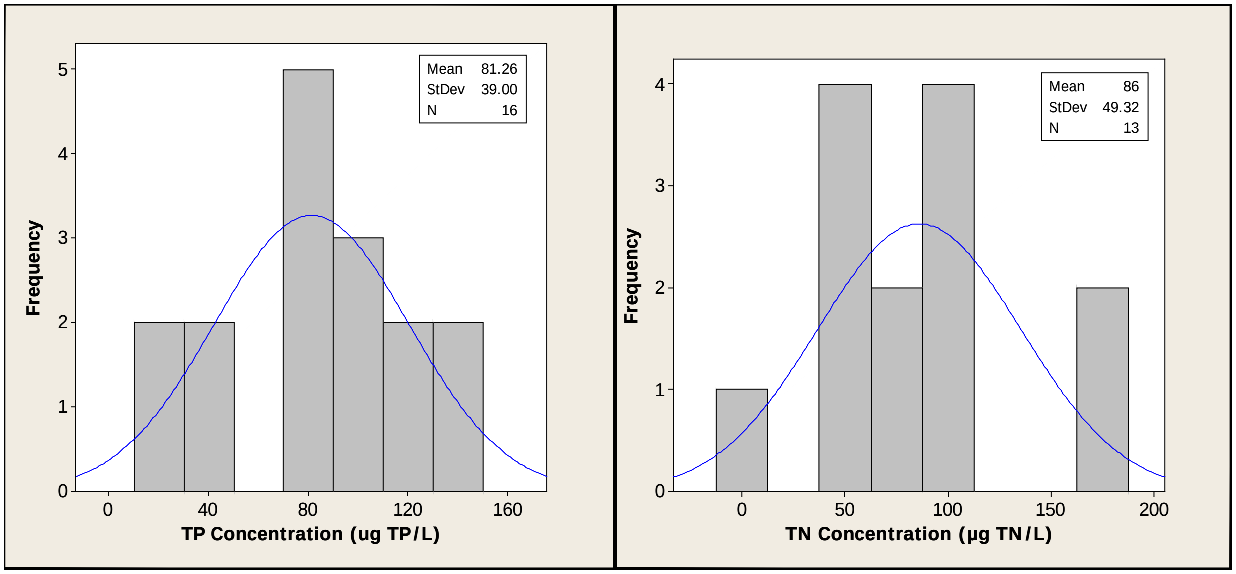


Figure 3-5. Nutrient concentrations from reference streams in the Absaroka-Gallatin Volcanic Mountains (17i) ecoregion.
Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). Data are from the Growing Season (July 1-September 30).

<u>Discussion of the Absaroka-Gallatin Volcanic Mountains Ecoregion</u>
The Absaroka-Gallatin Volcanic Mountains ecoregion (17i) has statistically significantly higher TP concentrations than the rest of the Middle Rockies (Varghese and Cleland, 2008; Varghese and Cleland, 2009). Permian age Phosphoria formations (United States Geological Survey, 1951) outcrop throughout this ecoregion and cause naturally elevated P concentrations. The natural concentrations of TP in 17i exceed harm-to-use thresholds identified for the Middle Rockies (20-40 µg TP/L). The median TP concentration of reference streams in ecoregion 17i is 77 to 81 µg/L (median or all observations datasets, respectively), compared to 8-10 µg/L for the Middle Rockies as a whole, and is therefore already higher than saturation (Dodds et al., 2006).

Observation of the reference streams of this ecoregion indicate that nuisance levels of benthic algae are not developing. This suggests that they are N limited, otherwise one would expect high algae levels at these TP concentrations (as observed in the transitional level IV ecoregions of the Northwestern Glaciated Plains, discussed later on). Natural TN levels in these streams are fairly low, lower than what is observed in the Middle Rockies as a whole. To assure that management-induced changes in TN do not lead to stream impacts, careful consideration of the appropriate TN criterion was essential. Equations relating benthic algal Chl*a* to total nutrients (Dodds et al., 1997, Equation 17; Dodds et al., 2006; Equation 19) were used to calculate TN that would maintain 125 mg Chl*a*/m^2 benthic algae given a TP of 105 µg/L (105 µg TP/L = 75th percentile of reference of 17i). This resulted in TN concentrations from 245 to 287 µg TN/L. If the algae level is set instead to 150 mg Chl*a*/m^2, and 105 µg TP/L is again used, the TN values range from 322 to 483 µg/L. Total phosphorus at the 75th percentile of reference was selected because it assures that the majority of data from the ecoregion's reference sites are below the TP criteria, and it lends itself well to reach-specific criteria derivation in cases where a stream reach further down gradient receives water from both the Middle Rockies and the Absaroka-Gallatin Volcanic Mountains (more on this in **Section 4.2**).

Conclusion

We recommend 105 µg TP/L and 250 µg TN/L as criteria for this level IV ecoregion. The TN criterion is more restrictive here than the 300 µg/L recommended for the Middle Rockies, and more restrictive than the other western ecoregions that will be discussed below; this is to assure adequate control of N in these apparently N-limited streams. The criteria have an N:P ratio of 2:1, however this is acceptable because maintaining a ratio near to Redfield is not realistic (or necessary) since the streams' natural N:P ratios are already low (on the order of 1:1). The reference data for this ecoregion were collected between 1990 and 2009, providing good temporal dispersion. Since there are still only a minimal number of samples (13-16) available for characterizing this ecoregion we recommend continued sample collection to increase the sample size.

3.2 LEVEL III: NORTHERN ROCKIES (ECOREGION 15)

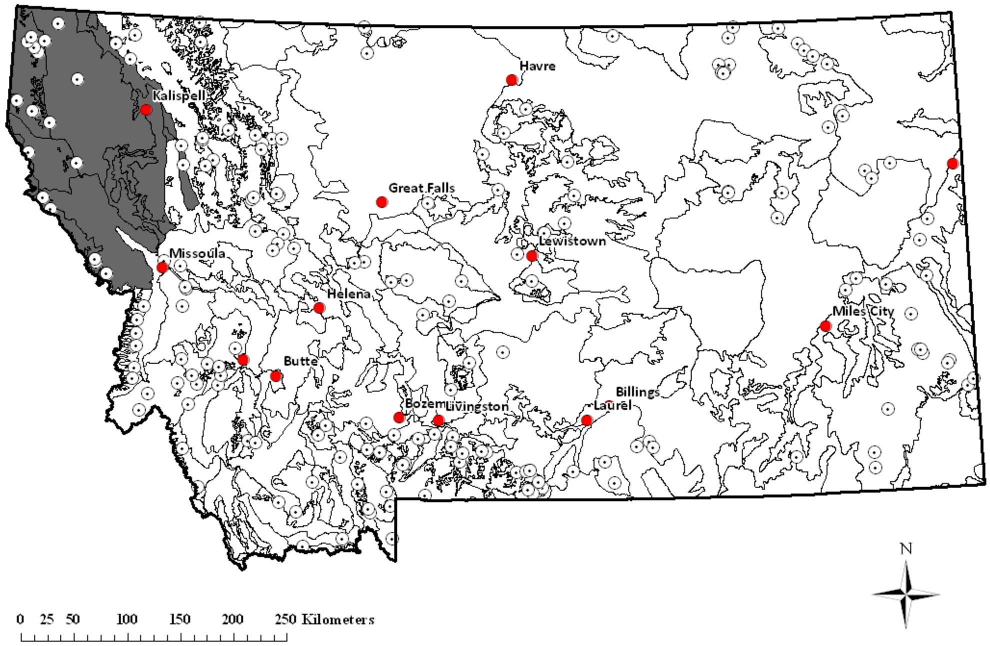


Figure 3-6. Map of Montana showing Northern Rockies ecoregion.
White dots are the reference sites.

Recommended Numeric Criteria
Total Phosphorus: **25 µg TP/L**
Total Nitrogen: **275 µg TN/L**

N:P Ratio of Criteria: **11:1**
N:P Ratio of Reference Sites: **8:1** (Redfield N:P ratio = 7:1)

Descriptive Statistics of Regional Reference Sites

Table 3-3A. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Northern Rockies ecoregion.
Data are from the all-observations dataset after applying Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	Number of Reference Sites	Number of Samples	Min	Max	25th	(Median)50th	75th	90th
TN	22	76	3	360	18	41	94	167
TP	22	81	0.5	18	4	6	9	13

Table 3-3B. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Northern Rockies ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	Number of Reference Sites	25^{th}	(Median) 50^{th}	75^{th}	90^{th}
TN	22	18	39	79	131
TP	22	3	5	9	13

The 25 µg TP/L criterion is greater than the 100th percentile of reference for both datasets. The 275 µg TN/L criterion matches to the 96th percentile of reference (all observations dataset) and the 99.5th percentile in the median dataset.

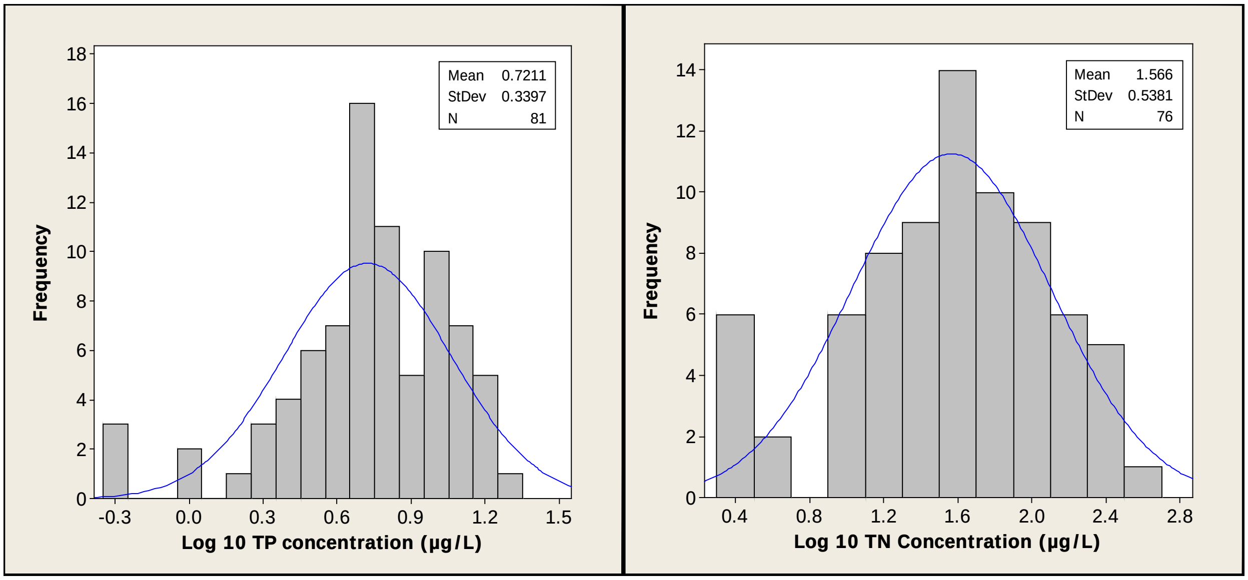


Figure 3-7. Nutrient concentrations from reference streams in the Northern Rockies ecoregion. Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). Data were collected during the Growing Season (July 1-September 30).

__Discussion of the Northern Rockies Ecoregion Nutrient Criteria__

Three regional dose-response studies specific to the Northern Rockies were available that relate nutrients (both soluble and total forms) to stream impacts or changes in aquatic communities. Welch et al. (1989) use a model and an artificial stream study and then adapt them to an open river system (Spokane River, Washington). Their equations indicate that at 10 µg soluble reactive phosphate (SRP)/L, the distance on the river with algal biomass of 150 mg Chl*a*/m^2 would be constrained to 16 km. The Montana public found a mean of ≤150 mg Chl*a*/m^2 acceptable for river recreation[5]. Assuming an SRP:TP ratio of 0.25:1 (as is commonly observed on the Clark Fork River), 10 µg SRP/L equals 40 µg TP/L. Chambers et al. (2011) derive nutrient criteria for Canadian streams using multiple methods including dose-response relationships between nutrients and algae and macroinvertebrate metrics. The study streams were located in the Okanagan Basin (British Columbia) just north of Washington State, and are within the Northern Rockies ecoregion. They recommend 20 µg TP/L and 210 µg TN/L for streams of the region to protect aquatic life.

The third study (Gravelle et al., 2009a; Gravelle et al., 2009b) discusses a Before After Control Impact Paired study in which the authors assess the effects of different timber harvest intensities on nutrient concentrations and aquatic insect metrics in the Mica Creek Experimental Watershed in northern Idaho.

[5] The Spokane River is a 6th order river and is therefore on the large side of wadeable (Flynn and Suplee, 2010), and impacts to dissolved oxygen standards would be less likely in a river this size due to good re-aeration and total river volume. Therefore we only used 150 mg Chl*a*/m^2 in the equation (as opposed to 125 mg Chl*a*/m^2, discussed in **footnote 4**).

In the post-road construction period (1998-2001), summer TP increased to about 40 µg/L, TKN increased slightly to about 150 µg/L, and nitrate+nitrite did not change. Later, in the post-harvest period (2002-2006), TP declined again to 20 µg/L and TKN to about 40 µg/L, but nitrate+nitrite increased markedly to a monthly summer average of 350 µg/L (about 400 µg TN/L). Across this entire ten year period there were very few changes in the aquatic insect metrics monitored, although Ephemeroptera, Plecoptera, and Trichoptera abundance increased over the period (Gravelle et al., 2009b). Among the biometrics, the Hilsenhoff Biotic Metric (HBI) was of particular interest as the Department uses it as part of the assessment of nutrient impacts in mountainous streams (Suplee and Sada de Suplee, 2011). Relative to the control period (1994-1997), HBI scores were essentially unaffected by the nutrient concentration changes observed. Based on the data, it is likely that the streams were N limited in the post-road period and P limited in the post-harvest period.

Beyond this ecoregion, applicable studies are essentially the same as described in **Section 3-1** for the Middle Rockies (excluding Chambers et al., 2011). These studies indicate a range of candidate criteria from 20-30 µg TP/L and 300-1,210 µg TN/L. Work by Mebane et al. (2009) in central Idaho has less direct application here, but note that streams where they observed very low ambient TP and TN concentrations (similar in concentration to Northern Rockies reference streams) N and P co-limitation was the norm.

<u>Conclusion</u>

We recommend 25 µg TP/L and 275 µg TN/L for this ecoregion. The scientific literature most specific to this ecoregion (Welch et al., 1989; Chambers et al., 2011; Gravelle et al., 2009a; Gravelle et al., 2009b) suggests criteria ranging from 20-40 µg TP/L and 210-400 µg TN/L. Some consideration was given to the fact that the natural background concentrations in this ecoregion are quite low relative to the range of potential criteria. The concentrations 25 µg TP/L and 275 µg TN/L result in an N:P ratio of 11, which is higher than the regional reference stream ratio (8:1) but still close to the Redfield range where co-limitation by N and P occurs (the 11:1 ratio suggests slight P limitation).

3.3 LEVEL III: CANADIAN ROCKIES (ECOREGION 41)

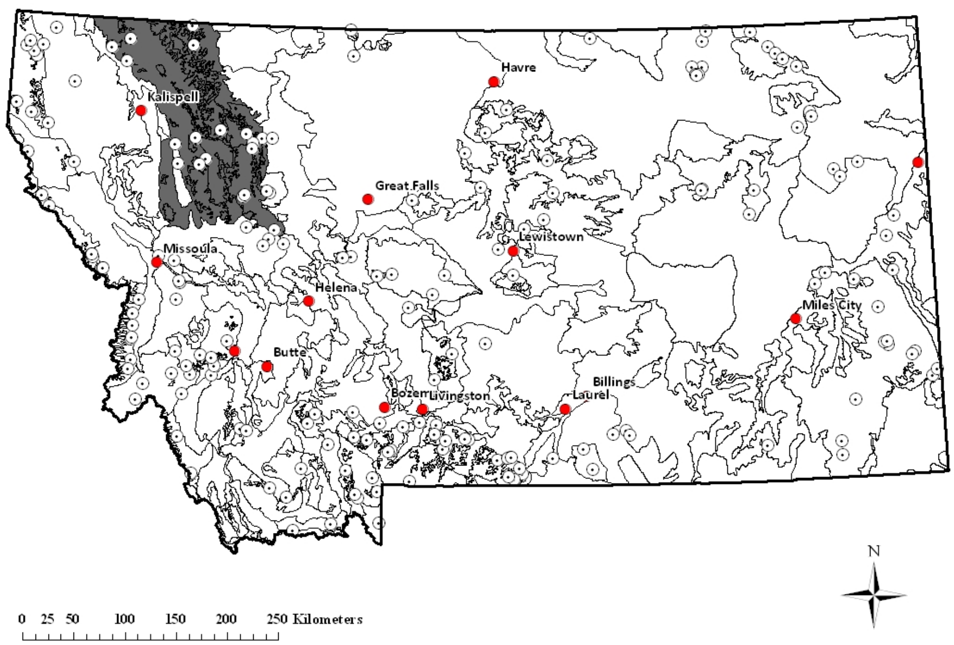


Figure 3-8. Map of Montana showing Canadian Rockies ecoregion.
White dots are the reference sites.

Recommended Numeric Criteria

Total Phosphorus: **25µg TP/L**
Total Nitrogen: **325 µg TN/L**

N:P Ratio of Criteria: **13:1**
N:P Ratio of Reference Sites: **16:1** (Redfield N:P ratio = 7:1)

Descriptive Statistics of Regional Reference Sites

Table 3-4A. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Canadian Rockies ecoregion.
Data are from the all-observations dataset after applying Brillouin Evenness Index.

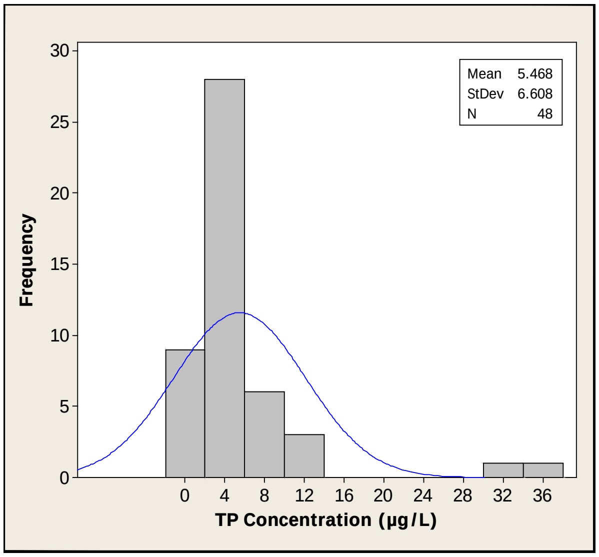
			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	Number of Reference Sites	Number of Samples	Min	Max	25th	(Median)50th	75th	90th
TN	13	39	2.5	413	27	63	156	268
TP	14	48	0.5	35	2	4	6	9

Table 3-4B. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Canadian Rockies ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	Number of Reference Sites	25th	(Median) 50th	75th	90th
TN	13	40	80	156	245
TP	14	4	5	6	7

The 25 µg TP/L criterion matches the 97th percentile of reference in the all observations dataset and the 98th percentile of reference in the median dataset. .

The 325 µg TN/L criterion matches the 95th percentile of reference all observations dataset and the 99th percentile of reference in the median dataset.



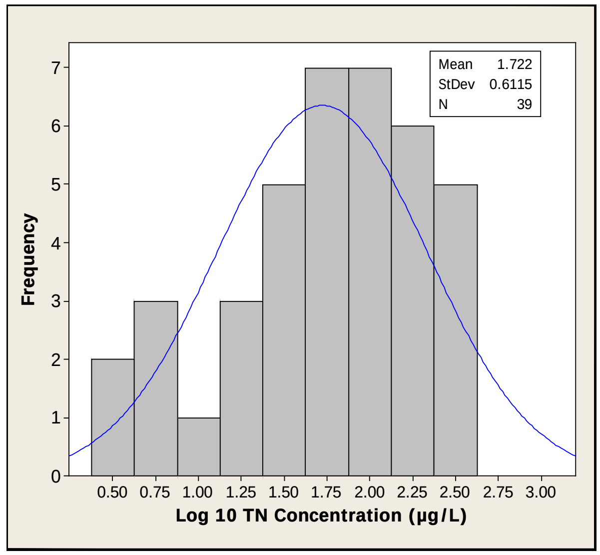


Figure 3-9. Nutrient concentrations from reference streams in the Canadian Rockies ecoregion. Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). Data were collected during the Growing Season (July 1-September 30).

Discussion of the Canadian Rockies Ecoregion Nutrient Criteria

Several studies have direct application to the Canadian Rockies (Sosiak, 2002; Bowman et al., 2007; Scrimgeour and Chambers, 2000). All three were carried out in Canadian rivers in the ecoregion. No model equation between benthic Chl*a* and nutrients was provided in Bowman et al. (2007), however Michelle Bowman graciously provided us the data for the relationship between TP and benthic Chl*a* from their study (personal communication, January 21, 2009). The TP-Chl*a* correlation, though weak, suggests that benthic algal biomass of 150 mg Chl*a*/m^2 equates to 89 µg TP/L, and 125 mg Chl*a*/m^2 equates to 66 µg TP/L (see **footnote 4** for information pertaining to the benthic algae levels used here).

Sosiak (2002) provides a Chl*a* vs. total dissolved phosphate (TDP) + NO_{2+3} multiple-regression equation, and a conversion between TDP and TP concentrations for the Bow River (TP is about 2.8 X TDP). He reports that 150 mg Chl*a*/m^2 equates to 18 µg TP/L (equal to 10 µg TP/L @ 125 mg Chl*a*/m^2). But he assumes that nitrate in the river is essentially saturated (conc. = 267 µg NO_{2+3}-N/L). We reset the NO_{2+3} in the equation to a value (50 µg NO_{2+3}-N/L) that is a much more realistic proportion of any foreseeable TN criterion, with the following results: 150 mg Chl*a*/m^2 corresponds to 41 µg TP/L, and 125 mg Chl*a*/m^2 equates to 23 µg TP/L. Scrimgeour and Chambers (2000) note that, in the absence of human influence, the Wapiti-Smokey rivers are probably P limited, but once alterations to the water quality occur due to kraft mill effluent, N and P co-limitation is most common. Finally, watershed managers on the Bow River are recommending 28 µg TP/L in the central Bow River in order to maintain benthic algae ≤ 150 mg Chl*a*/m^2 (Bow River Basin Council, 2008).

Relevant dose-response studies from outside the ecoregion are essentially the same as those described for the Middle Rockies. Work carried out in northern and southern temperate rivers and streams show that nutrient-benthic Chl*a* regressions have breakpoints at 27-62 µg/L for TP and between 367-602 µg/L for TN (Dodds et al., 2006). Stevenson et al. (2006) show in Michigan streams that the likelihood of reaching bottom coverage by *Cladophor*a of 20-40% increases sharply when TP exceed 30 µg TP/L and 1,000 µg TN/L. This level of streambed coverage by *Cladophora* is unacceptable to the Montana public (see Suplee et al., 2009, Table 1). Chambers et al. (2011) derive nutrient criteria for Canadian streams using multiple methods including dose-response relationships between nutrients and algae and macroinvertebrate metrics. They recommend 20 µg TP/L and 210 µg TN/L for the Montane Cordillera, a mountainous region west and north of Montana in British Columbia (in the Northern Rockies ecoregion). Equations relating benthic algal Chl*a* to total nutrients (Dodds et al., 1997, Equation 17; Dodds et al., 2006; Equation 19), respectively), were used to calculate TN levels that would maintain 125 mg Chl*a*/m^2 given a TP of 25 µg/L. These equations result in TN concentrations ranging from 528-821 µg TN/L. But Bowman et al. (2007) state that nutrient-algae relationships in nutrient-poor lotic systems are harder to predict, and specifically note that Dodds' equations under predict benthic algal biomass of oligotrophic rivers such as those found in the Canadian Rockies. As such, Dodds' equations (and the work of Stevenson et al. (2006)) need to be considered cautiously in this ecoregion.

Conclusion

Total P values derived from Sosiak (2002) and Bowman et al. (Bowman et al., 2007) are in the range of 23 to 89 µg TP/L. The Bow River Basin Council (2008) suggests 28 µg TP/L to maintain river benthic algae at the same levels considered here. None of the equations specific to the Canadian Rockies provide a means to easily derive a TN criterion. Given that the reference sites in this ecoregion have a fairly high TN:TP ratio (16:1, **Table 3-4B**; highest of the western ecoregions), and that others have noted the inherent P limitation of the region (Scrimgeour and Chambers, 2000), it would be prudent to establish TP values that maintain this inherent P limitation. **We recommend 25 µg TP/L, and a corresponding TN criterion (giving consideration to Redfield and the region's natural N:P ratio) of 325 µg TN/L.** Criteria in this ratio (13:1) should induce slight P limitation (as is inherent in the ecoregion), but are shifted somewhat toward a Redfield ratio that would result in co-limitation.

A final note. One level IV ecoregion within the Canadian Rockies, the Southern Carbonate Front (41d), had statistically higher total Kjeldahl N (TKN) concentrations in its reference sites compared to reference sites of the rest of the Canadian Rockies (Varghese and Cleland, 2008; Varghese and Cleland, 2009). Total Kjeldahl N is a close surrogate for TN, therefore we investigated whether or not this level IV ecoregion should have a separate TN criterion. The Southern Carbonate Front's median TN concentration (73 µg TN/L) falls midrange of the Canadian Rockies as a whole (63-80 µg TN/L; **Tables 3-4A, B**), and the Southern Carbonate Front's nitrogen levels are not high enough to warrant separate criteria. The criterion recommended for the Canadian Rockies, 325 µg TN/L, matched the 91st percentile of the Southern Carbonate Front's TN and TKN reference distribution. (When only its TN data were considered, 325 µg TN/L matched the 94th.) Thus, 325 µg TN/L is an appropriate criterion for this level IV ecoregion as well since the great majority of its reference data are less than the criterion.

3.4 LEVEL III: IDAHO BATHOLITH (ECOREGION 16)

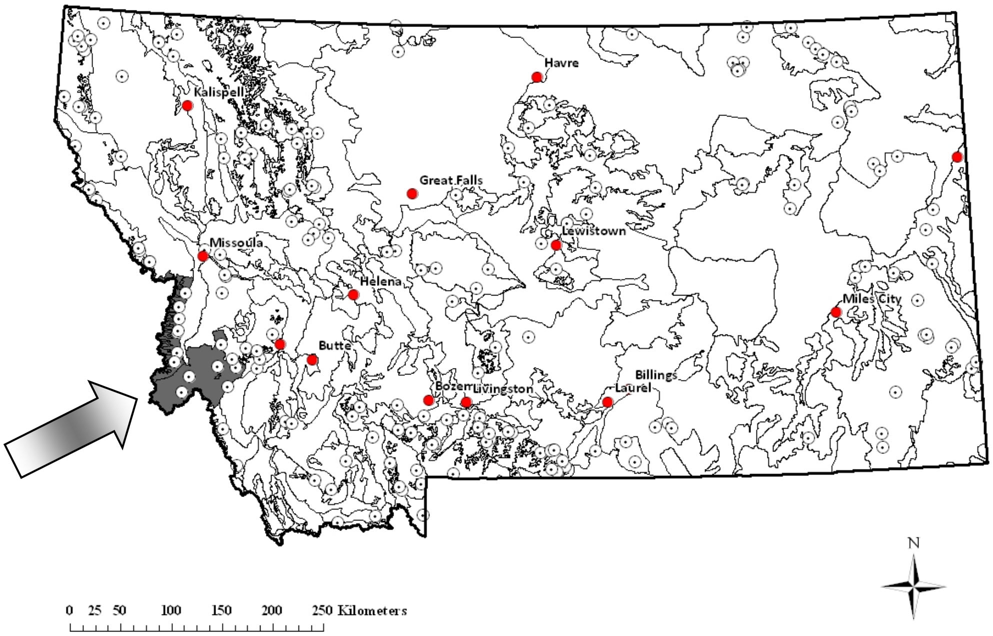


Figure 3-10. Map of Montana showing the Idaho Batholith ecoregion.
White dots are the reference sites.

__Recommended Numeric Criteria__
Total Phosphorus: **25 µg TP/L**
Total Nitrogen: **275 µg TN/L**

N:P Ratio of Criteria: **11:1**
N:P Ratio of Reference Sites: **10:1** (Redfield N:P ratio = 7:1)

__Descriptive Statistics of Regional Reference Sites__

Table 3-5A. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Idaho Batholith ecoregion.
Data are from the all-observations dataset after applying Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	Number of Reference Sites	Number of Samples	Min	Max	25th	(Median)50th	75th	90th
TN	9	28	2.5	238	46	70	95	163
TP	9	28	0.5	19	4	6	8	11

Table 3-5B. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Idaho Batholith ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	Number of Reference Sites	25th	(Median) 50th	75th	90th
TN	9	40	62	72	104
TP	9	6	6	8	11

The 25 µg TP/L criterion is beyond the 100th percentile of reference for both datasets.
The 275 µg TN/L criterion is beyond the 100th percentile of reference for both datasets.

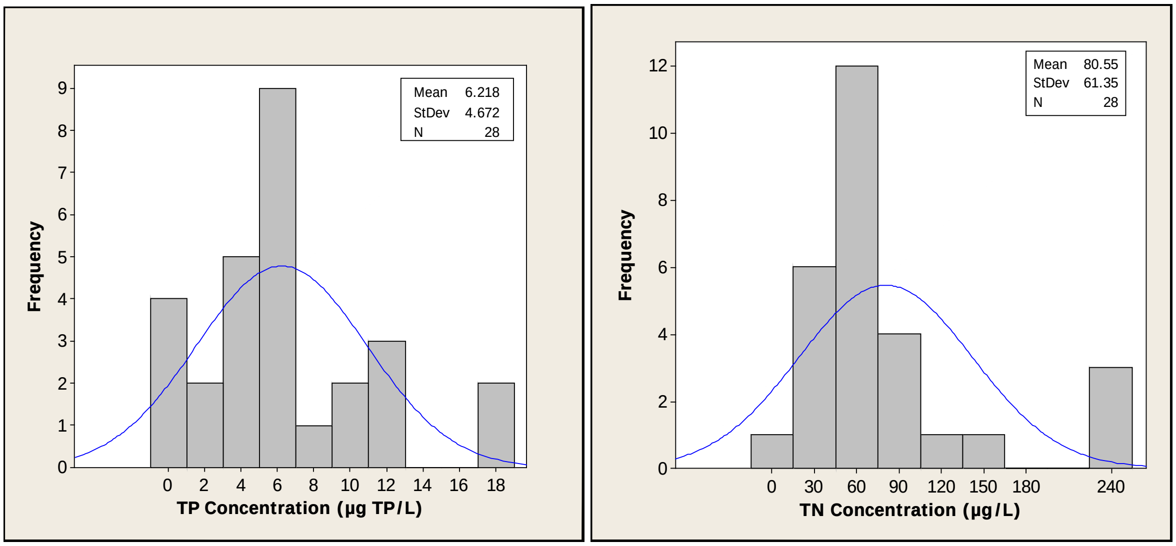


Figure 3-11. Nutrient concentrations from reference streams in the Idaho Batholith ecoregion. Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). Data were collected during the Growing Season (July 1-September 30).

Discussion of the Idaho Batholith Ecoregion Nutrient Criteria

There is a relatively small extent of this level III ecoregion in Montana (most of it is found in central Idaho). Mebane et al. (2009) carried out a multiple approach, dose-response study in wadeable streams, and several of their sites are located in the Idaho Batholith ecoregion of central Idaho. They carried out *in situ* nutrient limitation trials using nutrient diffusers and other approaches. One of their study sites located in the Idaho Batholith (the Big Wood River) had very low ambient nutrients (7-10 µg TN/L and 50-150 µg TN/L), not unlike reference streams of this ecoregion in Montana (**Tables 3-5A,B; Figure 3-11**). The Big Wood River was found to be very strongly N and P co-limited and, in fact, single N- and P-additions alone grew no more algae than did the un-amended control. This indicates that solo N or P control (no control on the other) would need to be maintaining concentrations no higher than natural background (ca. 8 µg TP/L or 100 µg TN/L) to prevent substantial increases in benthic algal growth. Among the study streams, N and P co-limitation or N-limitation was most common. These data demonstrate that coupled N and P criteria are important to maintain desired water quality conditions. Mebane et al. (2009) recommend criteria of 40 µg TP/L and 600 µg TN/L in order to maintain benthic algae growth to ≤150 mg Chl*a*/m^2, per Suplee et al. (2009). But to maintain 125 mg Chl*a*/m^2, which is more appropriate for many wadeable streams (see **footnote 4**), corresponding concentrations are about 35 µg TP/L and 475 µg TN/L. Note that in Mebane et al.'s study a good response curve was obtained between benthic algal Chl*a* and TN. This was less true for TP where, in a number of cases, there were a fair number of observations falling outside the Chl*a*-TP response curve (i.e., higher benthic algal Chl*a* was observed at lower-than-predicted TP levels). Finally, the study of Mebane et al. (2009) included a number of sites in agricultural settings of the Snake River Plain ecoregion (12), so the study's findings should only be carried so far when applying them to the Idaho Batholith.

Relevant dose-response studies from outside the ecoregion would be the same as those described for the Northern Rockies and Canadian Rockies, which have similar, low nutrient concentrations. Chambers et al. (2011) derive nutrient criteria for Canadian streams using multiple methods, including dose-response of nutrients vs. algae and macroinvertebrate metrics. They recommend 20 µg TP/L and 210 µg TN/L for the Montane Cordillera, a mountainous region in the Northern Rockies ecoregion. Other work in the Northern Rockies suggests values of about 40 µg TP/L and 400 µg TN/L (Gravelle et al., 2009a; Gravelle et al., 2009b). In the Canadian Rockies, a TP criterion could fall between 23 and 89 µg/L and, for the Bow River there, we documented a recommendation of 28 µg TP/L (Bow River Basin Council, 2008). Equations relating benthic algal Chl*a* to total nutrients (Dodds et al., 1997, Equation 17; Dodds et al., 2006; Equation 19) were used to calculate TN levels that would maintain 125 mg Chl*a*/m^2 benthic algae, given a TP of 30 µg TP/L. These equations resulted in TN concentrations from 466-718 µg TN/L. But Bowman et al. (2007) note that nutrient-algae relationships in nutrient-poor lotic systems (like the Idaho Batholith) are harder to predict and, specifically, the equations of Dodds et al. (2006) under predict the actual benthic algal growth observed.

Conclusion

We found that a study that has application to this ecoregion (Mebane et al., 2009) indicates strong N and P co-limitation or N-limitation in the ecoregion's streams, apparently due to very low natural background nutrient concentrations. These low background nutrient concentrations are similar to those observed in the Northern and Canadian Rockies ecoregions (**Table 3-3A, B; Tables 3-4A, B**). The work of Mebane et al. (2009) suggests that good control of nitrogen is probably of greater importance in controlling algal biomass than phosphorus. Bowman et al. (2007) state that nutrient-algae relationships in nutrient-poor lotic systems are hard to predict, and that published models (such as Dodds et al., 2006) under predict actual benthic algal biomass. We gave this finding careful consideration when we evaluated the range of potential criteria for this ecoregion. The work of Mebane et al. (2009) suggests values of 35-40 µg TP/L and 475-600 µg TN/L. The broader range of recommended values that are most applicable to the Idaho Batholith are 20-89 µg TP/L and 210-400 µg TN/L. Consideration was given to the fact that the natural background concentrations in this ecoregion are quite low relative to the range of potential criteria. **We recommend 25 µg TP/L and 275 µg TN/L.** The TN and TP criteria we recommend are in a ratio (10:1) which matches the natural background for the Idaho Batholith (11:1; **Table 3-5B**) and which (based on Redfield) would result in slight P limitation or possibly N and P co-limitation (the latter of which is typically for regional streams).

3.5 Level III: Northwestern Glaciated Plains (Ecoregion 42)

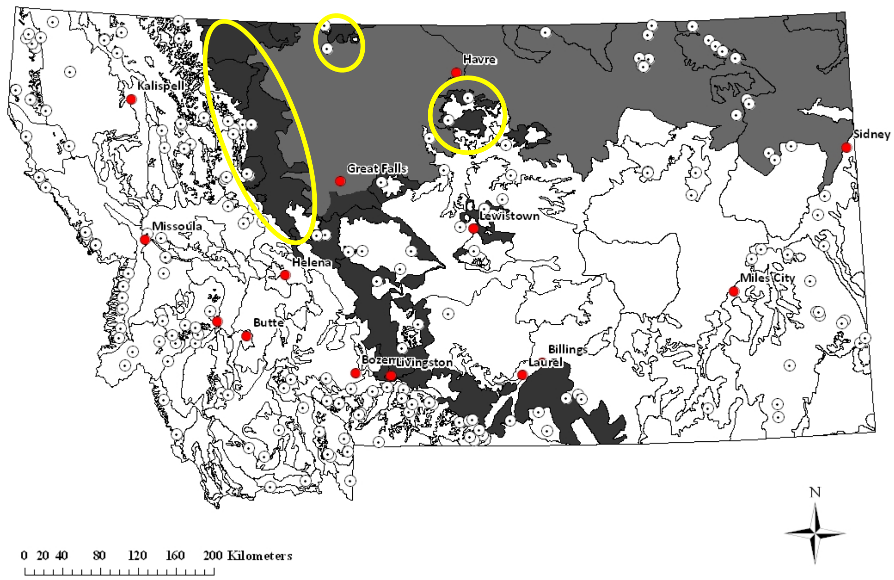


Figure 3-12. Map of Montana showing the Northwestern Glaciated Plains ecoregion (42) in light gray. The dark gray area is a mountain-to-plains transitional zone comprised of level IV ecoregions within ecoregion 42 (and 43, to the south). Mountain-to-plains transitional level IVs that are part of the Northwestern Glaciated Plains (circled in yellow) were not included among the reference data compiled here. White dots are the reference sites.

Recommended Numeric Criteria

Total Phosphorus: **110 µg TP/L**
Total Nitrogen: **1,300 µg TN/L**

N:P Ratio of Criteria: **12:1**
N:P Ratio of Reference Sites: **18:1** (Redfield N:P ratio = 7:1)

Descriptive Statistics of Regional Reference Sites

Table 3-6A. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Northwestern Glaciated Plains ecoregion.
Data are from the all-observations dataset after applying Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	**Number of Reference Sites**	**Number of Samples**	**Min**	**Max**	**25th**	**(Median)50th**	**75th**	**90th**
TN	17	52	55	3891	630	969	1398	1945
TP	18	59	10	638	28	60	111	184

Table 3-6B. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Northwestern Glaciated Plains ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	**Number of Reference Sites**	**25th**	**(Median) 50th**	**75th**	**90th**
TN	17	720	900	1325	2078
TP	18	26	49	89	105

The 110 µg TP/L criterion matches to the 75th percentile of reference in the all-observations dataset and to the 92nd percentile in the median dataset.

The 1,300 µg TN/L criterion matches to the 65th percentile of reference in the all-observations dataset and to the 72nd percentile in the median dataset.

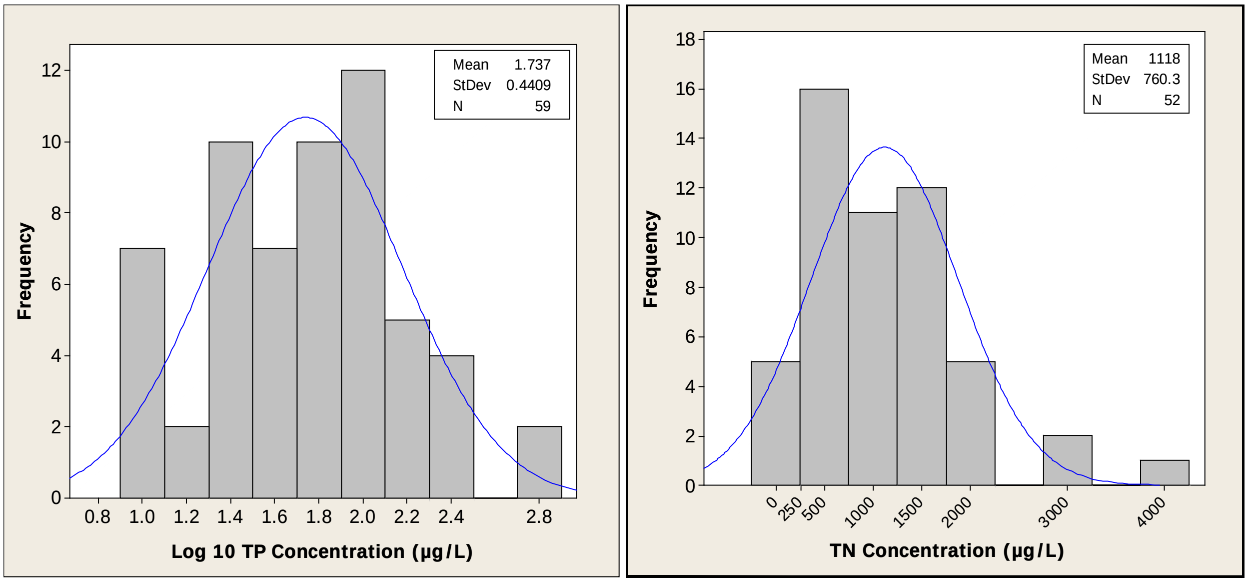


Figure 3-13. Nutrient concentrations from reference streams in the Northwestern Glaciated Plains ecoregion, but excluding data from the level IV ecoregions 42l, 42n, 42q, and 42r.
Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). Data were collected during the Growing Season (June 16-September 30).

Discussion of the Northwestern Glaciated Plains Nutrient Criteria

One study has direct application to the Northwestern Glaciated Plains (Suplee et al., 2008), and there are three additional studies carried out in this ecoregion or in glaciated plains regions further east that have relevance (Wang et al., 2007; Heiskary et al., 2010; Chambers et al., 2011). In Appendix A of Suplee et al. (2008), a relationship is shown between TN concentrations and dissolved oxygen (DO) concentrations as inferred by diatom taxa. Once TN was higher than 1,120 µg TN/L, streams sites had DO concentrations lower than the B-2 DO standard (5.0 mg/L; DEQ, 2010); B-2 streams are widespread throughout this ecoregion. Note in **Tables 3-6A and 3-6B** above that this threshold concentration (1,120 µg TN/L) matches around the 60th percentile of reference; rather low in the reference distribution given that reference sites—by definition— are minimally impacted and support their uses. On close observation of Figure 3.2 in Appendix A of (Suplee et al., 2008), it appears that a concentration between 1,100 and 1,450 µg TN/L could be appropriate as a threshold. Indeed, the 90% confidence interval around the threshold is 780 to 1,480 mg TN/L (Suplee et al., 2008). Giving consideration to the ecoregion's reference distribution (**Table 3-6B; median dataset**), where the 75th percentile of reference equals 1,325 µg TN/L, a TN criterion of about 1,300 µg TN/L appears to be appropriate. It should also be noted that N limitation was strongly indicated in wadeable streams of this region (Suplee, 2004) and was also given consideration in selecting 1,300 µg TN/L. Although the TN:TP ratio of the reference sites (18:1) might suggest the contrary, Redfield ratios are only meaningful when nutrient concentrations are low and generally below saturation (i.e., at concentrations much lower than the natural concentrations found in this ecoregion).

Chambers et al. (2011) derive nutrient criteria for prairie streams of the Northwestern Glaciated Plains ecoregion in Alberta, Canada. They use modeling to relate % agriculture in the watershed and stream nutrient concentrations. In that method, the y-intercept (i.e., zero agriculture) is used to define 'no impact' (Dodds and Oakes, 2004) and can help define a candidate criterion. For the Northwestern Glaciated Plains in Canada this equals 680 µg TN/L. They also use methods involving fixed percentiles of regional reference (and non-reference) sites. Based on all these methods, they provide a range of TN from 680 to 1,110 µg/L, and recommend 980 µg TN/L as a provisional threshold for prairie stream protection. Using the same methods, they also recommend a provisional TP criterion of 106 µg /L.

Fish biometrics have been developed for warm-water plains streams of Montana (Bramblett et al., 2005). The work was carried out exclusively in Montana, in the Northwest Glaciated Plains and the Northwestern Great Plains ecoregion to the south. Fish biometrics provides a means to address harm-to-use for warm-water fish assemblages. One metric, 'proportion of tolerant individuals'[6], shows significant positively correlation with TKN concentrations and significant negative correlation with DO concentrations. Though no specific TN threshold can be drawn from Bramblett et al. (2005), their findings lend support to our work which shows that elevated TN concentrations impact regional DO (and, in turn, the more sensitive taxa in the warm-water fish assemblage).

Patterns in warm-water plains streams in Wisconsin may be roughly comparable to warm-water streams of the Northwestern Glaciated Plains. Wang et al. (2007) examine Wisconsin warm-water streams (e.g., the Southeast Wisconsin Till Plains ecoregion) and relationships between nutrient concentrations and various warm-water fish metrics. One of the metrics ('% individuals considered intolerant')[7] was significantly correlated (negatively) with TP and TN. (This metric is essentially the mirror image of 'proportion of tolerant individuals' of Bramblett et al. (2005).) Wang et al. (2007) also report changepoint thresholds between nutrients and the metric. Nutrient changepoint thresholds represent concentrations above which warm-water fish assemblages are likely to be substantially degraded (Wang et al., 2007). The '% individuals considered intolerant' metric was found to have an ecological threshold at 70-90 µg TP/L and 540 to 1,830 µg TN/L.

Heiskary et al. (2010) recommend numeric nutrient criteria for rivers in different regions of Minnesota, including the southern region of the state which is warm-water and dominated by the Western Corn Belt Plains ecoregion. They examine relationships between DO concentration and fish metrics (including changepoint thresholds), benthic and phytoplankton algae vs. nutrient concentrations, and DO flux (daily maximum minus the daily minimum) vs. invertebrate and fish metrics. Fish metrics include the '% sensitive fish species' metric, which is comprised of most of the same species used in Wisconsin (Lyons, 1992) and suggests there is a fairly consistent bioassessment approach across that plains region. Heiskary et al. (2010) recommend a TP criterion of 150 µg TP/L to protect fish and aquatic life in the southern region of Minnesota.

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[6] The metric is comprised of highly tolerant species, including: goldfish, common carp, fathead minnow, white sucker, black bullhead, and green sunfish (Bramblett et al., 2005).

[7] This metric was specifically designed to assess fish assemblages in perennial warm-water streams of intermediate size (ie., wadeables), and has direct application to southeastern Minnesota as well (Lyons, 1992). The fish comprising the intolerant species list are almost all warm-water species, and a few are even found in this region of Montana (e.g., smallmouth bass, Iowa darter, and silvery minnow; Brown, 1971).

Earlier we presented work showing that in wadeable streams TP is saturated at 24-62 µg TP/L and TN between 367-602 µg/L (Dodds et al., 2006; Suplee et al., 2012); our recommended criteria for western MT generally are set within or below those ranges since nutrient levels must be below saturation breakpoints to achieve improvements in algae levels/eutrophication. Natural background concentrations of nutrients in the Northwestern Glaciated Plains ecoregion are already at or above these concentrations (Chambers et al., 2011) (see also **Table 3-6A and Table 3-6B**), and yet harm-to-use thresholds at even higher nutrient concentrations can still be identified. If benthic algae are nutrient-saturated, how is this so? We believe it is strongly related to the basic ecology of these streams. The region's streams tend towards two scenario endpoints: (1) un-scoured streams dominated by macrophytes and benthic algae, and (2) scoured, more turbid streams where phytoplankton can be dominant (Suplee, 2004; Suplee et al., 2008, Appendix A). In scenario 2 streams, summer scouring events give phytoplankton a competitive edge because they can tolerate higher turbidity by rotating through the water column via wind and flow advection. Higher turbidity surely induces more light limitation, yet summer phytoplankton concentrations can in cases become very high (>70 µg Chl*a*/L; (Suplee, 2004)). Other regional streams have phytoplankton Chl*a* concentrations as high as 516 µg/L (Suplee, 2004). Phytoplankton-dominated wadeable streams are rarely seen in western MT, and were not a meaningful proportion of the datasets used to derive the nutrient saturation levels mentioned above. In the clearer, un-scoured streams (scenario 1) of the Northwestern Glaciated Plains, macrophytes can—if stimulated enough by nutrients—impact DO concentrations, especially when they senesce (Jewell, 1971). Macrophytes can gain nutrients from the water but also from the sediments (Chambers et al., 1989), obscuring the direct water column nutrient concentration vs. DO relationship. Dissolved oxygen problems probably need to become fairly severe before notable impacts to plains fishes occur, because these fish are already naturally tolerant (Bramblett et al., 2005).

So what we are dealing with here are streams (and associated flora and fauna) of a very different (and more low-DO tolerant) nature relative to western MT, which leads to the higher nutrient impact thresholds observed. That is, to discern a harm-to-use impact in plains streams, levels of nutrients well above the saturation point for algae in non-plains streams are needed in order to override the physical, floral, and faunal differences (and accompanying confounding factors).

Conclusion

Work specific to Montana indicates a TN criterion of 1,300 µg TN/L would maintain DO concentrations at standards common throughout the ecoregion. Studies carried out in warm-water streams in the plains of Canada, in Wisconsin, and in Minnesota provide a range of values from 540 to 1,830 µg TN/L and 70 to 150 µg TP/L. **We recommend 1,300 µg TN/L and 110 µg TP/L.** The TN criterion was derived from a study carried out specifically in this ecoregion in Montana and falls within the range of potential values located in the literature. The TP criterion (110 µg TP/L) is very close to that recommended by Chambers et al. (2011), equates to the 75^{th}-92^{nd} percentile of the reference distribution for this ecoregion (**Tables 3-6A, B**), and falls within the range of criteria located in the literature.

3.5.1 Transitional Level IV Ecoregions within the Northwestern Glaciated Plains: Sweetgrass Upland (42l), Milk River Pothole Upland (42n), Rocky Mountain Front Foothill Potholes (42q), and Foothill Grassland (42r)

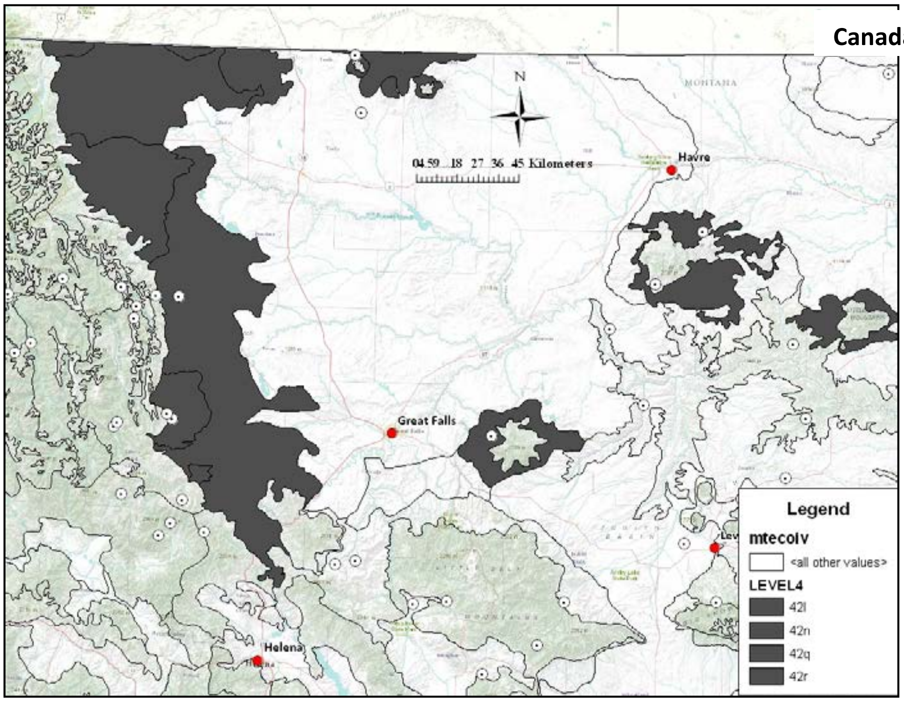

Figure 3-14. Map of Montana showing in gray the transitional level IV ecoregions (42l, 42n, 42q, and 42r) within the Northwestern Glaciated Plains.
White dots are the reference sites.

Recommended Numeric Criteria
Total Phosphorus: **80 µg TP/L**
Total Nitrogen: **560 µg TN/L**

N:P Ratio of Criteria: **7:1**
N:P Ratio of Reference Sites: **10:1** (Redfield N:P ratio = 7:1)

Descriptive Statistics of Regional Reference Sites

Table 3-7A. Descriptive Statistics for TN and TP Concentrations in Transitional Level IV Ecoregions (42q, 42r) of the Northwestern Glaciated Plains. No data were available for 42l, 42n.
Data are from the all-observations dataset after applying Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	**Number of Reference Sites**	**Number of Samples**	**Min**	**Max**	**25^{th}**	**(Median)50^{th}**	**75^{th}**	**90^{th}**
TN	5	20	24	2830	115	253	515	704
TP	5	17	1	380	9	20	78	246

Table 3-7B. Descriptive Statistics for TN and TP Concentrations in Transitional Level IV Ecoregions (42q, 42r) of the Northwestern Glaciated Plains.
No data were available for 42l, 42n. Data are from the median dataset.

		Concentration at given Percentile (μg/L)			
Nutrient	**Number of Reference Sites**	**25th**	**(Median) 50th**	**75th**	**90th**
TN	5	143	176	200	392
TP	5	11	18	35	206

The 80 μg TP/L criterion matches to the 75th percentile of reference in the all observations dataset and the 79th percentile in the median dataset.

The 560 μg TN/L criterion matches to the 80th percentile of reference in the all observations dataset and is greater than the 100th percentile in the median dataset.

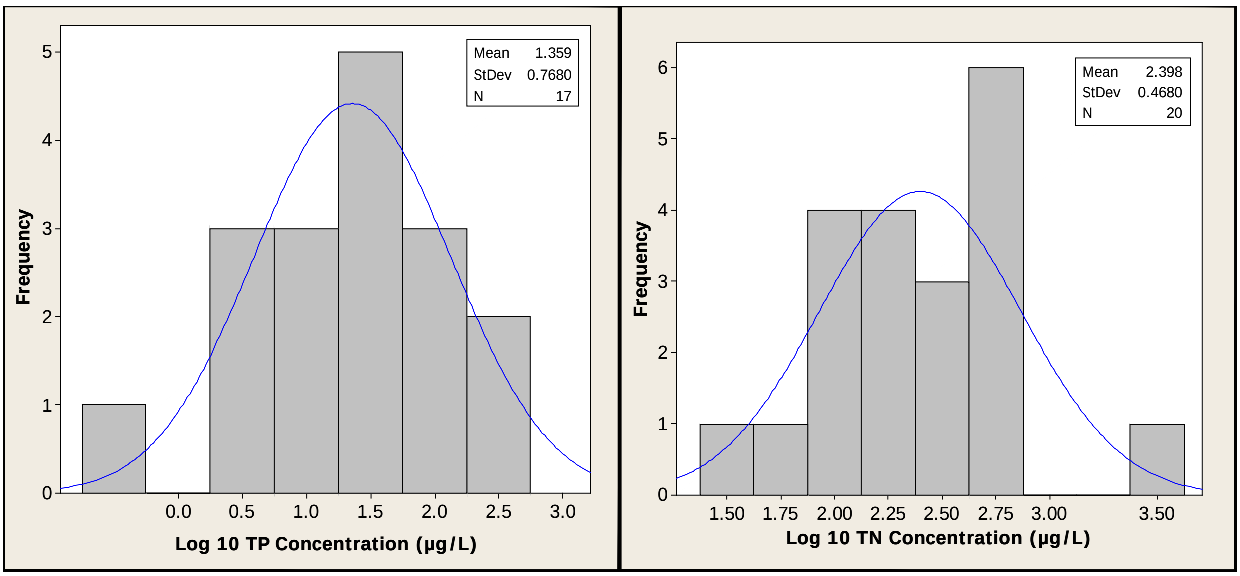

Figure 3-15. Nutrient concentrations from reference streams in the Transitional Level IV ecoregions (42q, and 42r) of the Northwestern Glaciated Plains.
Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). No data were available for 42l and 42n. Data were collected during the Growing Season (June 16-September 30).

Discussion of the Nutrient Criteria for the Transitional Level IV Ecoregions Within the Northwestern Glaciated Plains

In general, streams located in these transitional level-IV ecoregions have more in common with the mountains than the plains. Several lines of information provide support for this. First, although these transitional level IVs form part of the Northwestern Glaciated Plains ecoregion, most of the streams in the transitional level IVs are classified by the state as B-1. This means that they are expected to support salmonid fisheries and are coldwater systems, in sharp contrast to the warm-water streams found further to the east. Clearly, those who developed the stream class system in the late 1950s recognized the strong mountain influences on these streams. Second, floristically they have more in common with mountain streams. Teply and Bahls (2007) carry out a hierarchical cluster analysis on diatom algae (Bacillariophyta) from Montana reference sites, and find that streams of the transitional region are best classified along with the mountain streams. In fact, the level IV ecoregions addressed here (42l, 42n, 42q, and 42r) are being assessed by the Department using diatom metrics designed to evaluate coldwater streams (Teply, 2010; Montana Department Environmental Quality, 2011). Third, although the natural level of nutrients here (**Tables 3-7A, B**) are higher than the most nutrient-rich mountain ecoregion (the Middle Rockies, **Tables 3-1A, B**), they are still much lower than concentrations observed in the Northwestern Glaciated Plains further to the east (**Table 3-6A, B**).

Note that the TP and TN criteria recommended for the four level-III mountain ecoregions to the west (25-30 µg TP/L and 275-325 µg TN/L) fall between the 50^{th} and 60^{th} percentile of reference for these transitional ecoregions (all-observations dataset; **Table 3-7A**). For the median dataset (**Table 3-7B**) the level-III ecoregion criteria ranges fall between the 60^{th} and 67^{th} percentile of reference in this ecoregion for TP, and between the 81^{th} and 85^{th} for TN. Clearly, with these small datasets how the statistical summaries are prepared has a big effect on the distributional statistics reported.

Some reference streams in the transitional ecoregions have benthic algae levels that fall close to or are slightly above the recreationally-based threshold (150 mg Chl*a*/m^2; **Table 3-8**). In reference streams of Montana's mountainous ecoregions (e.g., Middle Rockies, Northern Rockies), we have not measured a site-average benthic Chl*a* level >80 mg/m^2 and the median among sites there is only 14 mg Chl*a*/m^2 (Suplee et al., 2009); also recall that nutrient concentrations of reference streams in the mountain ecoregions are usually well below the recommended criteria. One would expect higher background algae levels in this transitional region given the elevated background nutrient concentrations, and this is what we observe.

Table 3-8. Site-average Benthic Algae Levels Measured in Streams of Ecoregions 42r and 42q.

Site Name	Reference Site No.	Ecoregion (level IV)	Sampling Date	Site-average Chl*a* (mg/m^2)
Clear Creek	ClearCre_121_W	42r	9/20/2001	19
Clear Creek	ClearCre_121_W	42r	8/5/2003	29
Clear Creek	ClearCre_121_W	42r	8/7/2003	17
Clear Creek	ClearCre_121_W	42r	9/7/2003	41
Clear Creek	ClearCre_121_W	42r	8/21/2009	37
Barr Creek lower site at Sun River WMA	BarrCree_504_C	42q	7/9/2009	159
Barr Creek lower site at Sun River WMA	BarrCree_504_C	42q	8/6/2009	84
Barr Creek lower site at Sun River WMA	BarrCree_504_C	42q	9/11/2009	95
Rose Creek upstream from confluence with Barr Creek	RoseCree_518_C	42q	7/10/2009	91
Rose Creek upstream from confluence with Barr Creek	RoseCree_518_C	42q	8/8/2009	148
Rose Creek upstream from confluence with Barr Creek	RoseCree_518_C	42q	9/13/2009	60

WMA – wildlife Management Area

It was necessary to derive benthic algae and nutrient criteria specific to these transitional ecoregions since natural levels of nutrients here are elevated compared to the mountain ecoregions (but still well below concentrations of the plains). Natural algae levels are high enough (i.e., >125mg Chl*a*/m^2) that seasonal DO problems may occur in these streams and may influence the fisheries here. Therefore, the next beneficial use to consider is recreation. Suplee et al. (2009) show that site-average benthic algal Chl*a* levels of 150 mg Chl*a*/m^2 are acceptable to the public, but 200 Chl*a*/m^2 clearly are not. Since benthic algae levels in this region may naturally reach 159 mg Chl*a*/m^2 (**Table 3-8**), but values >200 mg Chl*a*/m^2 have not been observed, a value of 165 mg Chl*a*/m^2 should be an appropriate target. This is about the highest algae level (giving consideration to the statistical confidence around the 150 mg Chl*a*/m^2 average) the public would find acceptable (Suplee and Sada de Suplee, 2011).

Equations relating benthic algal Chl*a* to total nutrients (Dodds et al., 1997, Equation 17; Dodds et al., 2006; Equation 19) were used to calculate TN levels that would maintain 165 mg Chl*a*/m^2 benthic algae based on a TP level of 80 µg/L (80 µg TP/L = 75th-79th percentile of reference here; **Tables 3-7A, B**). These equations resulted in TN concentrations ranging from 445 to 775 µg TN/L. Also, note that nutrient-benthic Chl*a* regressions have saturation breakpoints at 27-62 µg TP/L and between 367-602 µg/L for TN (Dodds et al., 2006), and, the inherent TN:TP ratio of the region is about 13:1 (higher than Redfield).

<u>Conclusion</u>

Giving consideration to the equations of Dodds et al. (2006), saturation thresholds (Dodds et al., 2006), Redfield ratio, and natural background nutrient levels, we **recommend 80 µg TP/L and 560 µg TN/L for the transitional ecoregions 42l, 42n, 42q, and 42r.** The TN criterion is in the range provided by the equations of Dodds et al. (2006), is lower than the higher nitrogen saturation concentration and, along with TP, provides an N:P ratio of 7:1 (at Redfield). Note also that the 560 µg/L TN criterion corresponds to the last commonly-observed concentration in the reference sites (equal to 2.75 log; **Figure 3-15**, right panel).The TP criterion (used to calculate the TN value), having been set to approximately the 75th percentile of regional reference, should prevent unnecessarily high false positive rates (i.e., declaring a reference stream as impaired) when the Department carries out assessments in this region. **The benthic algal biomass criterion for this region is also adjusted up to 165 mg Chl*a*/m^2 to account for natural background levels, with a corresponding Ash Free Dry Mass (AFDM) value equal to 70 g/m^2** (per AFDM, see Suplee et al., 2009, Table 1)**.**

3.6 Level III: Northwestern Great Plains (Ecoregion 43) and the Wyoming Basin (Ecoregion 18)

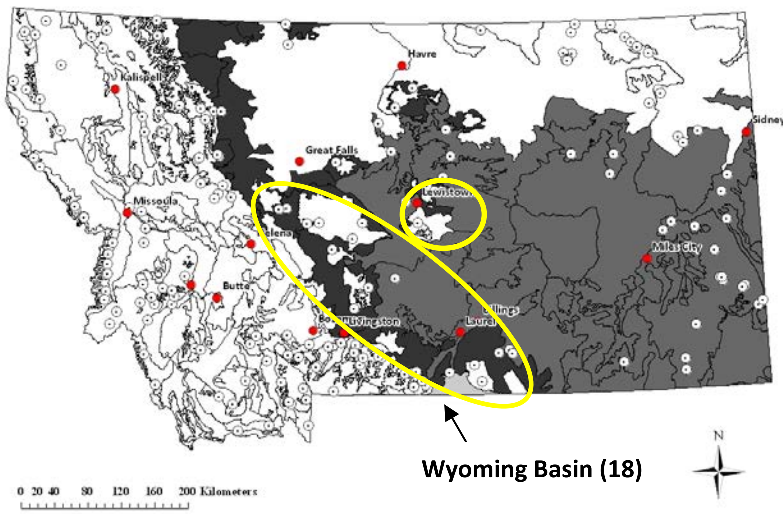

Figure 3-16. Map of Montana showing the Northwestern Great Plains ecoregion (43) in gray, the Wyoming Basin (18) in light gray.
Also, in dark gray, is the mountain-to-plains transitional zone comprising level IV ecoregions in ecoregion 43 (and 42 to the north). Mountain-to-plains transitional level IVs that are part of the Northwestern Great Plains (circled in yellow) were not included among the reference data compiled here. White dots are the reference sites.

Recommended Numeric Criteria
Total Phosphorus: **150 µg TP/L**
Total Nitrogen: **1,300 µg TN/L**

N:P Ratio of Criteria: **9:1**
N:P Ratio of Reference Sites: **13:1** (Redfield N:P ratio = 7:1)

Descriptive Statistics of Regional Reference Sites

Table 3-9A. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Northwestern Great Plains ecoregion.
Data are from the all-observations dataset after applying the Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	**Number of Reference Sites**	**Number of Samples**	**Min**	**Max**	**25^{th}**	**(Median)50^{th}**	**75^{th}**	**90^{th}**
TN	30	100	50	9900	482	792	1389	3141
TP	32	112	1	9911	36	73	137	519

Table 3-9B. Descriptive Statistics for TN and TP concentrations in Reference Streams of the Northwestern Great Plains ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	**Number of Reference Sites**	**25^{th}**	**(Median) 50^{th}**	**75^{th}**	**90^{th}**
TN	30	537	1042	1365	2296
TP	32	48	82	202	288

The 150 µg TP/L criterion matches to the 77th percentile of reference (all-observations dataset) and the 70th percentile of reference in the median dataset (**but see important caveat in paragraph just above the Conclusion, below**).

The 1,300 µg TN/L criterion matches to the 68th percentile of reference (all observations dataset) and the 73rd percentile of reference in the median dataset.

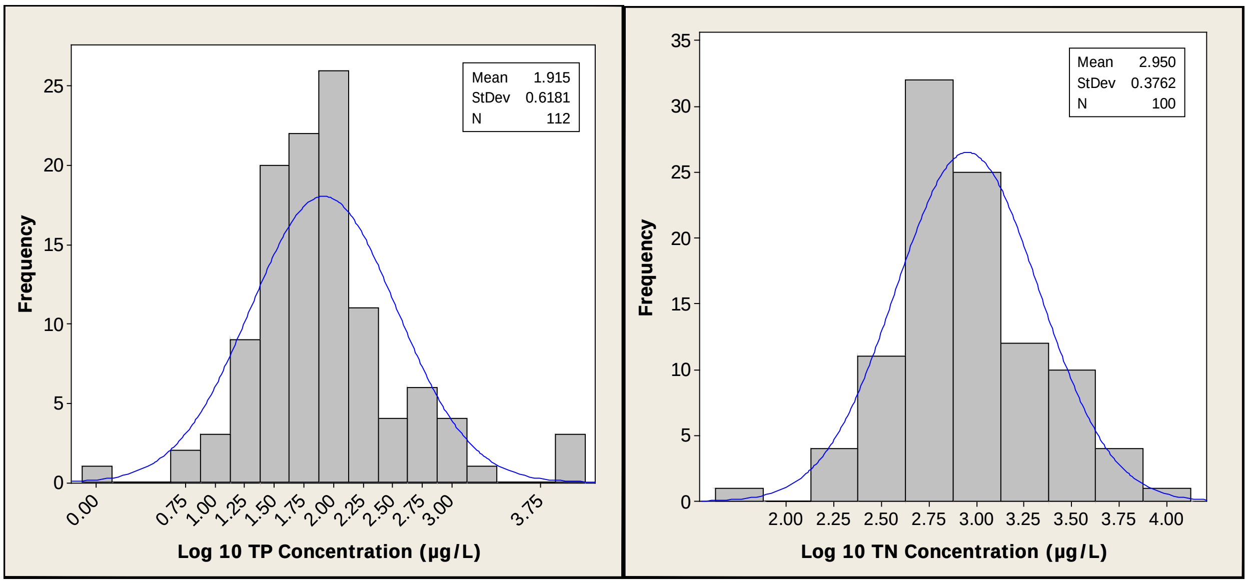

Figure 3-17. Nutrient concentrations from reference streams in the Northwestern Great Plains ecoregion (43), but excluding data from the mountain-to-plains transitional level IV ecoregions 43s, 43t, 43u, and 43v.
Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). Data were collected during the Growing Season (July 1-September 30).

Discussion of the Northwestern Great Plains Nutrient Criteria[8]

The Department (in cooperation with the Carter County Conservation District) carried out a whole-stream nitrogen and phosphorus addition study in Box Elder Creek, a reference stream site located in southeast Montana in the Northwestern Great Plains ecoregion. The full technical report for the study has not yet been prepared, but some of the work has been published (Suplee and Sada de Suplee, 2011). In addition to this study, there are three other studies carried out in plains regions further north and east that should have at least general relevance to this ecoregion (Wang et al., 2007; Heiskary et al., 2010; Chambers et al., 2011).

Appendix B of Suplee and Sada de Suplee (2011) contains key information about the Box Elder Creek dosing study. Additional facts are provided here.

[8] The level III ecoregion Wyoming Basin (18) has a very small extent in extreme south central Montana (Figure 3-16). No reference data are available there and for purposes of recommending regional nutrient criteria it is being lumped with the Northwestern Great Plains.

Nutrient dosing took place in summer 2010 and was preceded by "pre" data collection in 2009 and "post" data collected in 2011, at which times no nutrient additions were made. In the study, dissolved sodium nitrate was used as the N source and dissolved dipotassium phosphate as the P source. The High Dose reach was brought up to (after mixing) 150 µg NO_3-N/L and 23 µg SRP/L, continuously, for 53 days in August and September 2010. (Nutrients were added at a single point at the head of the study reach.) Ambient nutrient concentrations in summer are normally 3 µg NO_3-N/L and 4 µg SRP/L (or, in totals, about 500 µg TN/L and 54 µg TP/L). Loading calculations showed that the sodium and potassium added to the stream as part of the compounds used for nutrient additions increased ambient background of those elements by <0.5% and, therefore, are not considered a significant influence on the results. Benthic algae, stimulated by the nutrient additions, grew to levels far above normal for the stream and led to impacts on DO concentrations when the algae senesced *en masse* in early October when the growing season ended. Dissolved oxygen impacts appear to have occurred in patches longitudinally along the stream, with very low DO (zero mg/L) on the bottom in areas where the heaviest densities of decomposing algae settled. The study showed that there is a direct linkage between elevated inorganic nutrients, increased plant growth and, ultimately, impacts to DO standards. Probably the most surprising aspect of the work was that the DO impacts where out-of-phase with peak algal productivity.

Findings from the Box Elder Creek study are generally consistent with the results of the study in Appendix A of Suplee et al. (2008). In that study, DO concentrations—as inferred by diatom taxa—declined when nutrients (TN) became elevated. But what the Box Elder study adds to our understanding is that DO problems may be seasonal and longitudinally patchy in distribution along the stream bottom; in the most impacted locations, DO at the bottom essentially drops to zero. Bramblett et al. (2005), in developing an index of biotic integrity for this region based on fish, find that the 'proportion of tolerant individuals'[9] biometric is significantly correlated (positively) with TKN concentrations and significantly correlated negatively with DO concentrations. We speculate that when nutrient over-enrichment occurs, the more sensitive fish native to these warm-water streams are harmed by patchy, seasonally low DO, and are replaced by tolerant species that can withstand the changes. Indeed, the common carp (*Cyprinus carpio*) is one of the fish in the 'proportion of tolerant individuals' biometric, and is well known for its ability to tolerate very low DO concentrations (Brown, 1971).

Regarding studies from outside the Northwestern Great Plains ecoregion, Chambers et al. (2011) derive nutrient criteria for prairie streams of the Northwestern Glaciated Plains ecoregion in Alberta, Canada. Models that relate % agriculture in the watershed to stream nutrient concentrations are developed, and the y-intercept of the model (i.e., zero agriculture) defines the 'no impact' level (Dodds and Oakes, 2004). For prairie streams in Canada this equals 680 µg TN/L. They also use other methods involving fixed percentiles of regional reference (and non-reference) sites. Taken together, a range of TN concentrations from 680 to 1,110 µg/L is provided, and they recommend 980 µg TN/L as a provisional threshold. Using the same methods, they also recommend a provisional TP criterion of 106 µg /L.

Wang et al. (2007) examine streams in Wisconsin, including warm-water streams located in plains regions of that state (e.g., the Southeast Wisconsin Till Plains ecoregion). Wang et al. (2007) examine relationships between nutrient concentrations and various warm-water fish metrics, one of which ('% individuals considered intolerant') was significantly correlated (negatively) with TP and TN. (This metric is essentially the mirror image of 'proportion of tolerant individuals' of Bramblett et al. (2005). Wang et

[9] The metric is comprised of highly tolerant species, including: goldfish, common carp, fathead minnow, white sucker, black bullhead, and green sunfish (Bramblett et al., 2005).

al. (2007) report changepoint thresholds between nutrients and the metric. Nutrient changepoint thresholds represent concentrations above which warm-water fish assemblages are likely to be substantially degraded (Wang et al., 2007). The '% individuals considered intolerant' metric was found to have a threshold between 70 and 90 µg TP/L and 540 to 1,830 µg TN/L (the ranges are due to the different threshold identification techniques used).

Heiskary et al. (2010) recommend numeric nutrient criteria for rivers in different regions of Minnesota, including the southern region of the state which is warm-water and dominated by the Western Corn Belt Plains ecoregion. They examine relationships between DO concentration and fish metrics (including changepoint thresholds), benthic and phytoplankton algae vs. nutrient concentrations, and DO flux (daily maximum minus the daily minimum) vs. invertebrate and fish metrics. Fish metrics include the '% sensitive fish species' metric, which is comprised of most of the same species used in Wisconsin (Lyons, 1992) and suggests there is a fairly consistent bioassessment approach across that plains region. Heiskary et al. (2010) recommend a TP criterion of 150 µg TP/L to protect fish and aquatic life in the southern region of Minnesota.

Total P concentrations in the reference sites at the 75th percentile of reference (**Tables 9A, B**) are near or higher than the highest dose-response TP value we could locate as a potential criterion (150 µg TP/L, Heiskary et al., 2010). This would suggest that applying the 150 µg TP/L value in this region might be futile because the natural levels are already higher. However, the datasets shown include the River Breaks level IV ecoregion for which we are not recommending criteria (more on this next section). It has been standard practice throughout this document to include reference data from break-out level IV ecoregions when describing the overall reference condition for the coarser level III (except where noted for the transitional ecoregions). However we will make an exception here. If the River Breaks data are excluded from the median dataset, the 75th percentile for the Northwestern Great Plains drops to 130 µg/L, considerably lower than the 202 µg/L in **Table 3-9B**. This suggests that in areas outside of the River Breaks the 150 µg TP/L value would be meaningful (and matches the 77th percentile of the Northwestern Great Plains ecoregion not including the River Breaks data).

Conclusion

A scientific study (Box Elder Creek nutrient dosing) carried out in the Northwestern Great Plains in Montana shows a direct linkage between elevated nutrient concentrations and declines in DO concentration. These DO changes likely impact warm-water fish assemblages periodically and may lead to the undesirable changes in local fish assemblages observed by Bramblett et al. (2005). A study carried out in the Northwestern Glaciated Plains of Montana (Suplee et al., 2008, Appendix A) suggests a criterion of 1,300 µg TN/L to maintain DO concentrations at state standards. Studies carried out in warm-water streams in the plains of Canada, in Wisconsin, and in Minnesota provide a range of values from 540 to 1,830 µg TN/L and 70 to 150 µg TP/L. **We recommend 1,400 µg TN/L and 150 µg TP/L for this ecoregion.** The TN criterion for the Northwestern Glaciated Plains study (1,300 µg TN/L) is has good application to this ecoregion. This is supported by the similarity in the central tendency of the two regions' reference TN concentrations (**Tables 3-6B and 3-9B**), and a general similarity in the ecology of the streams in the two regions (see the discussion of plains streams ecology in the last paragraph of **Discussion of the Northwestern Glaciated Plains Nutrient Criteria**). A concentration of 1,300 µg TN/L matches the 68th and 73rd percentiles of reference (all-observations and median datasets, respectively).

The suggested TP criterion (150 µg/L) matches that recommended by Heiskary et al. (2010). **Table 3-9B** shows that natural background is higher than this concentration, but the statistics include the River Breaks level IV which has naturally high TP concentrations and for which we will not be recommending criteria (more on that, next section). **If River Breaks TP data are excluded from the Northwestern Great Plains ecoregion median dataset, the 150 µg TP/L criterion then matches the 77th percentile of reference (which makes the selection of 150 µg TP/L more reasonable).** Additional work and analysis is needed to refine the TP criterion for this ecoregion.

Note: No nutrient criteria are being proposed for the River Breaks level IV ecoregion (discussed next). As such, the criteria recommended above for the Northwestern Great Plains would not apply there.

3.6.1 Level IV Ecoregion within the Northwestern Great Plains: River Breaks (43c)

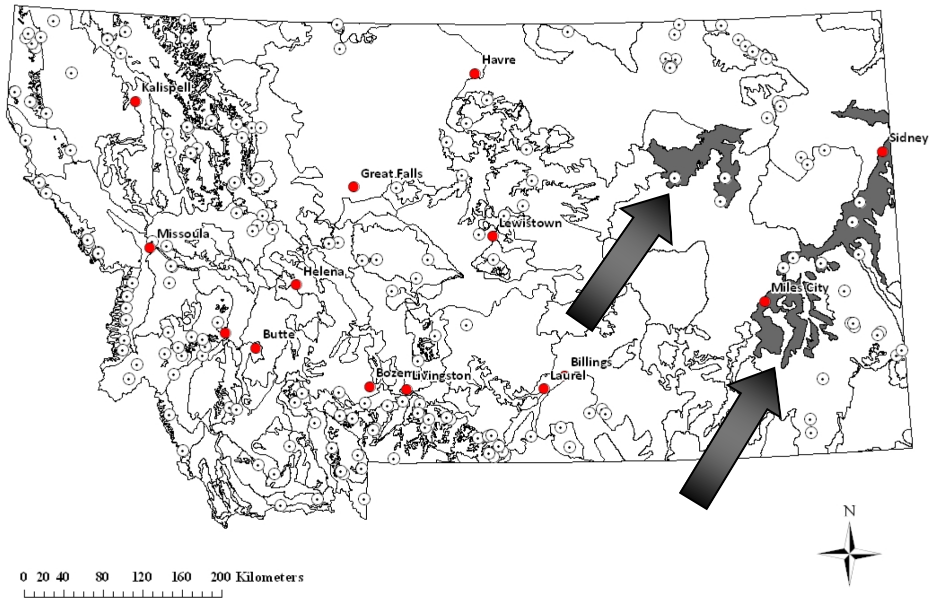

Figure 3-18. Map of Montana showing the River Breaks (43c), a level IV ecoregion within the Northwestern Great Plains ecoregion.
White dots are the reference sites.

Recommended Numeric Criteria
Total Phosphorus: **NONE RECOMMENDED**
Total Nitrogen: **NONE RECOMMENDED**

N:P Ratio of criteria: **n/a**
N:P Ratio of Reference sites: **8:1** (Redfield N:P ratio = 7:1)

Descriptive Statistics of Regional Reference Sites

Table 3-10A. Descriptive Statistics for TN and TP Concentrations in Reference Streams of the River Breaks (43c) level IV ecoregion.
Data are from the all-observations dataset after applying the Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	Number of Reference Sites	Number of Samples	Min	Max	25th	(Median)50th	75th	90th
TN	8	28	480	9900	1005	1333	2486	3792
TP	8	29	33	9911	51	129	293	2123

Table 3-10B. Descriptive Statistics for TN and TP Concentrations in Reference Streams of the River Breaks (43c) level IV Ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	Number of Reference Sites	25th	(Median) 50th	75th	90th
TN	8	1158	1213	2071	2270
TP	8	78	153	257	301

Table 3-10C. Descriptive Statistics for NO_{2+3} and SRP Concentrations in Reference Streams of the River Breaks (43c) level IV Ecoregion.
Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	**Number of Reference Sites**	**25th**	**(Median) 50th**	**75th**	**90th**
Nitrate + nitrite (NO_{2+3})	8	3	7	241	606
Soluble reactive P (SRP)	7	5	6	18	28

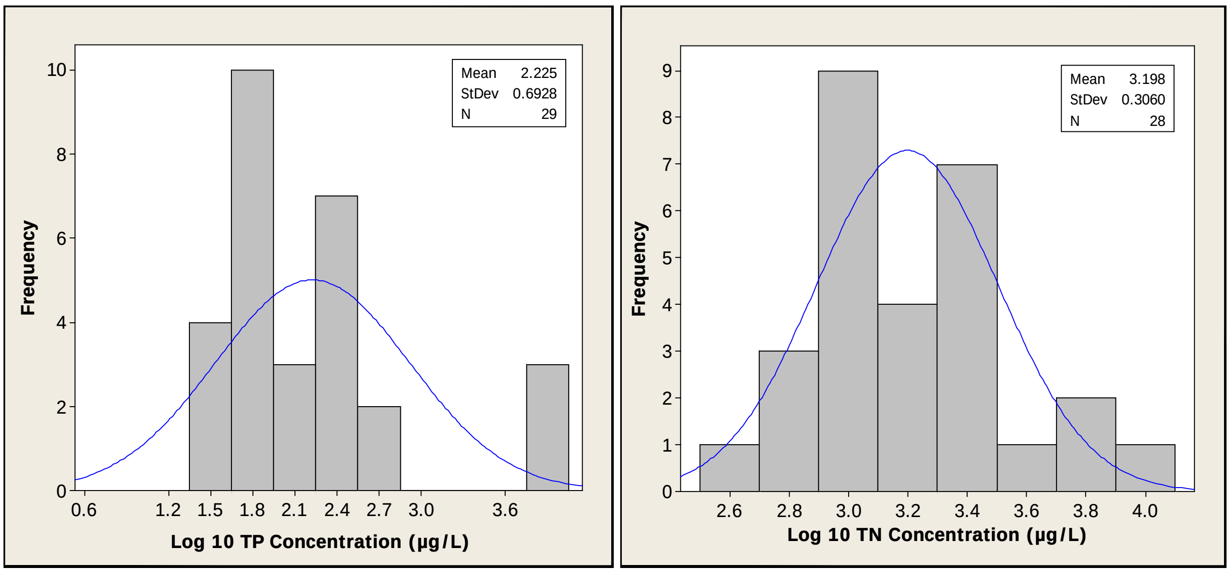

Figure 3-19. Nutrient concentrations from reference streams in the River Breaks (43c) level IV ecoregion.
Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). Data are from the Growing Season (July 1-September 30).

__Discussion of the River Breaks (43c) Nutrient Criteria__
As shown in **Tables 3-10A** and **B**, average TN concentrations in the reference sites of this level IV ecoregion are near to or higher than harm-to-use levels identified for the Northwestern Great Plains and the Northwestern Glaciated Plains (i.e., 1,300 µg TN/L). On the phosphorus side, the highest dose-response TP criterion identified so far for the plains (150 µg TP/L;(Heiskary et al., 2010) corresponds to the 57th percentile in the all-observations dataset and the 50th percentile in the median dataset, i.e., about average for this ecoregion's reference sites.

In this ecoregion we have also presented the soluble nutrient concentrations (**Table 3-10C**). In relation to algal biomass, it results that soluble nutrients here are saturated or nearly so in the reference sites. For NO_2+NO_3, the 75th percentile of River Breaks reference streams was 241 or 631 µg N/L (median or all-observations datasets, respectively; all observations table not shown). Rier and Stevenson (2006) show there is little peak algal biomass increase above 308 µg soluble nitrogen per liter (and peak biomass may actually be saturated closer to 250 µg DIN/L). As such, River Breaks reference sites are often saturated with soluble nitrogen. Soluble P concentrations are also quite high. At the 75th percentile of reference SRP is 18 or 20 µg P/L (median vs. all-observations datasets, respectively; all observations table not shown) and therefore these reference streams are often P saturated for peak algal biomass, or nearly so (Bothwell, 1989; Horner et al., 1983).

Clearly these streams have highly elevated nutrient levels naturally, but they also have characteristics that strongly dampen plant growth as they have not been found to develop a robust benthic flora. Highly dissected and erodible terraces and uplands lead to bottomlands of this ecoregion where the soils have poor permeability (Woods et al., 2002). This results in flashy, sediment-laden flows when summer thunderstorms occur. All eight of the reference sites in this region have been found to be extremely turbid when sampled in summer (e.g., as high as 30,000 mg TSS/L with accompanying turbidity of 4,000 nephelometric turbidity units), although by fall in some cases the water has cleared as summer thunderstorms diminish.

Benthic algal growth in these streams (due to the factors above) is usually low (**Figure 3-20),** lower than what is observed in streams for the plains regions as a whole (see data pertaining to the plains in the July 16, 2009 presentation to the Nutrient Work Group, *available at*: http://deq.mt.gov/wqinfo/nutrientworkgroup/AgendasMeetingsPresentations.mcpx). Macrophyte density also tends to be limited. In half of the eight reference streams from this ecoregion no macrophytes were observed at all, in two streams they were present but extremely sparse, and in two streams they were commonly found at sparse to moderate levels along much of the stream channel. As for benthic algae, flashy turbid flows and lengthy periods of high turbidity in these streams prevent a robust benthic flora from developing in many cases, in spite of abundant nutrient availability.

As has been observed in streams along Montana's Hi-line (Suplee, 2004), plains streams can at times develop high levels of phytoplankton Chl*a*. In this ecoregion, especially at these nutrient levels, this occurs in the reference sites as well. Many phytoplankton Chl*a* observations from the reference streams in the River Breaks are low (e.g., < 10 µg Chl*a*/L) but in quite a few cases they can become high (e.g., 72 µg Chl*a*/L, Hart Creek, 7/30/2006; 44 µg Chl*a*/L, Snap Creek, 8/24/2006). Suplee (2004) found that 95% of the phytoplankton samples from reference streams in the Northwestern Glaciated Plains were <20 µg Chl*a*/L. In comparison to the River Breaks, this suggests that—at least sometimes—River Breaks reference streams have naturally high phytoplankton Chl*a* concentrations due to ecological conditions (and elevated nutrients) prevalent in the ecoregion. Whether or not these high levels of phytoplankton Chl*a* affect the region's fish fauna is unknown. In all probability, the more severe physical constraints here (i.e., flashy conditions with extreme levels of suspended sediment) are a far greater constraint on the fish fauna and other aquatic life.

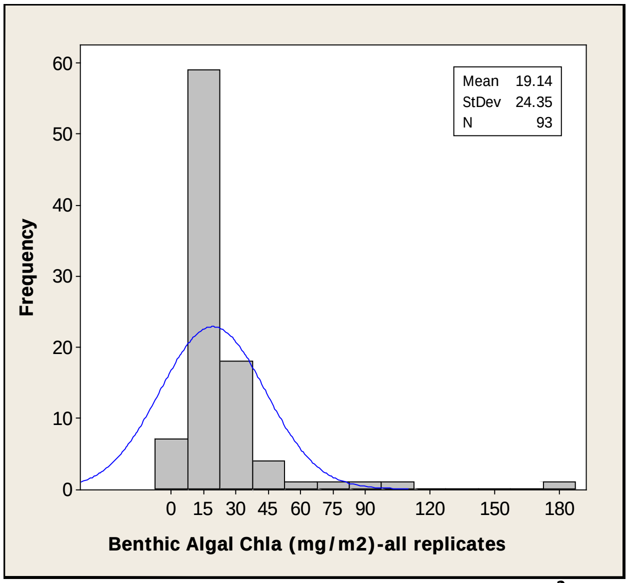

Figure 3-20. Benthic algal density (mg Chl*a*/m^2), all replicates, from reference streams in the River Breaks (43c) ecoregion.
Data are from the Growing Season (July 1-September 30).

<u>Conclusion</u>
This level IV ecoregion has highly turbid, flashy streams with naturally elevated TP and TN levels and soluble nutrient concentrations at or above saturation levels needed to support maximum algal biomass. Concentrations observed in the region's reference sites indicate that nutrient concentrations here are already near to or elevated above the harm-to-use thresholds identified for the plains region as a whole. **As such, no nutrient criteria are recommended for streams within this level IV ecoregion.** Readers should note that the nutrient criteria recommended for the Northwestern Great Plains (level III), discussed previously, would apply across that ecoregion, except here in the River Breaks.

3.6.2 Transitional Level IV Ecoregions within the Northwestern Great Plains: Non-calcareous Foothill Grassland (43s), Shields-smith Valleys (43t), Limy Foothill Grassland (43u), Pryor-Bighorn Foothills (43v), and Parts of the Unglaciated Montana High Plains (43o)[10]

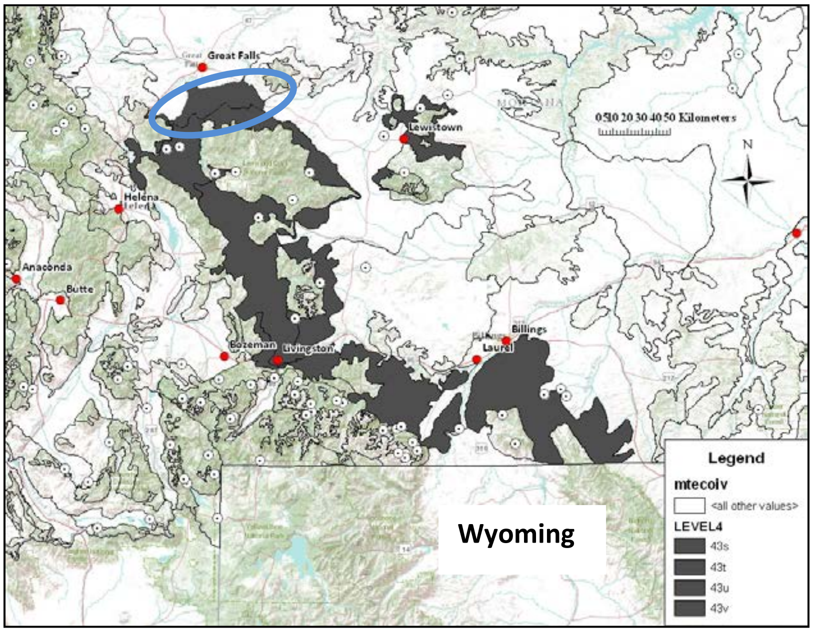

Figure 3-21. Map of Montana showing in gray the transitional level IV ecoregions (43s, 43t, 43u, and 43v) within the Northwestern Great Plains.
The sub-section of ecoregion 43o which is grouped with these level IV ecoregions is located just south of Great Falls (circled in blue). White dots are the reference sites.

Recommended Numeric Criteria
Total Phosphorus: **33 µg TP/L**
Total Nitrogen: **440 µg TN/L**

N:P Ratio of Criteria: **13:1**
N:P Ratio of Reference Sites: **13:1** (Redfield N:P ratio = 7:1)

Table 3-11A. Descriptive Statistics for TN and TP Concentrations in Transitional Level IV Ecoregions (43s, 43t, 43u) of the Northwestern Great Plains.
No data were available for 43o, 43v. Data are from the all-observations dataset after applying the Brillouin Evenness Index.

			Nutrient Concentration (µg/L)					
					Conc. at given Percentile			
Nutrient	Number of Reference Sites	Number of Samples	Min	Max	25th	(Median)50th	75th	90th
TN	12	40	50	753	78	112	174	224
TP	12	40	3	108	6	10	22	34

[10] For the Unglaciated Montana High Plains ecoregion, only the polygon located just south of Great Falls, MT is associated with this transitional region.

Table 3-11B. Descriptive Statistics for TN and TP Concentrations in Transitional Level IV Ecoregions (43s, 43t, 43u) of the Northwestern Great Plains.
No data were available for 43o, 43v. Data are from the median dataset.

		Concentration at given Percentile (µg/L)			
Nutrient	**Number of Reference Sites**	**25th**	**(Median) 50th**	**75th**	**90th**
TN	12	98	134	202	222
TP	12	8	10	27	33

The 33 µg TP/L criterion matches to the 87th percentile of reference (all observations dataset) and the 90th percentile of reference in the median dataset.

The 440 TN/L criterion matches to the 98th percentile of reference in the all-observations dataset and is greater than the 100th percentile in the median dataset.

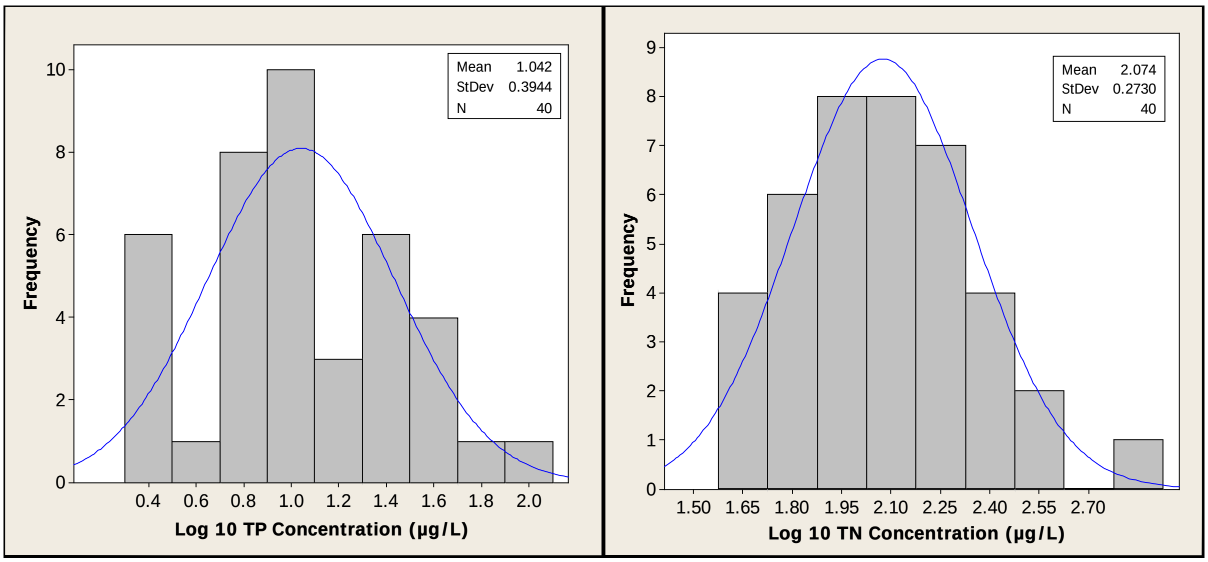

Figure 3-22. Nutrient concentrations from reference streams in the transitional level IV ecoregions (43s, 43t, 43u) of the Northwestern Great Plains.
Data shown are from the all observations dataset (after application of the Brillouin Evenness Index). No data were available for 43o or 43v. Data were collected during the Growing Season (July 1-September 30).

<u>Discussion of the Nutrient Criteria for the Transitional Level IV Ecoregions Within the Northwestern Great Plains</u>

In general, streams located in these transitional level-IV ecoregions have more in common with the mountains than the plains. Several lines of information support this. First, although these transitional level IVs form part of the Northwestern Great Plains ecoregion, virtually all streams in the transitional level IVs are classified by the state as B-1. This means that they are expected to support salmonid fisheries and are generally coldwater systems, in sharp contrast to the warm-water streams found in the Northwestern Great Plains further to the east. It is clear that when the state's stream-class system was developed in the late 1950s, its developers recognized the strong mountain influences on these streams. Second, floristically they have more in common with mountain streams. Teply and Bahls (2007) carried out a hierarchical cluster analysis on diatom algae (Bacillariophyta) from Montana reference sites, and find that streams of the transitional region are best classified along with the mountain streams. In fact, level IV ecoregions addressed here (43s, 43t, 43u, 43v and part of 43o) are being assessed by the Department using diatom metrics for coldwater streams (Teply, 2010; Montana Department

Environmental Quality, 2011). Third, the natural levels of nutrients here (**Tables 3-11A, B**) are not unlike the most nutrient-rich mountain ecoregion (the Middle Rockies, **Tables 3-1A, B**), but they are much lower than the Northwestern Great Plains further to the east (**Tables 3-9A, B**).

In contrast to nutrient concentrations, which are similar to the Middle Rockies, site-average benthic algae levels in these transitional region's reference sites are somewhat higher than the Middle Rockies (**Figure 3-23**). Of specific interest is a high average value (170 mg Chl*a*/m^2) from the Elk Creek reference site (reference site No. ElkCreek_511_C; ecoregion 43u). The high benthic algae density likely resulted from the stream's naturally elevated TP, where summer concentrations ranged from 29 to 41 µg/L (average: 31 µg TP/L). However, the data do not suggest that elevated nutrients are a common factor across the level IV ecoregion in which Elk Creek resides (Limy Foothill Grasslands, 43u). Another reference site there (Middle Fork Judith River; MFJudith_513_C) has TN and TP concentrations well below the median of the aggregate reference sites shown in **Table 3-11A**.

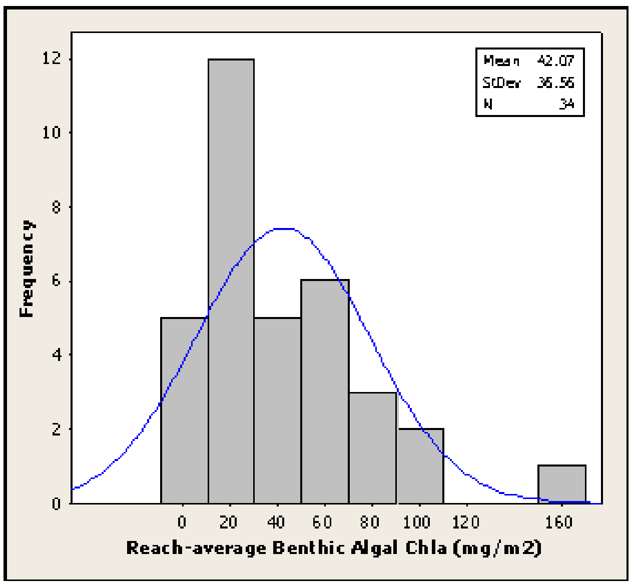

Figure 3-23. Site-average[11] benthic algae density (mg Chl*a*/m^2) from reference streams in the transitional level IV ecoregions (43s, 43t, 43u) of the Northwestern Great Plains ecoregion.

The transitional ecoregions of the Northwestern Great Plains (43s, 43t, 43u) have natural nutrient concentrations with a central tendency roughly comparable to the Middle Rockies. Further, most reference streams (9 of 10) of 43s, 43t, and 43u have benthic algae levels lower than the thresholds considered throughout this document (125 and 150 mg Chl*a*/m^2). However there is one exception to this, Elk Creek, where benthic algae was above the thresholds. This finding is in contrast to the transitional ecoregions of the Northwestern *Glaciated* Plains (see **Section 3.5.1**), where 2 of 3 reference sites had benthic algae levels above the thresholds; there, higher algae levels (and nutrients) seem to be the norm. Because elevated algae levels seem to be the exception and not the rule here in the transitional ecoregions of the Northwestern Great Plains, we used the 125 and 150 mg Chl*a*/m^2 thresholds to help derive the nutrient criteria. We also gave consideration to the naturally higher TP

[11] A site average is the arithmetic mean of (normally) 11 sample replicates collected along a short stream reach ≥ 150 m in length using an unbiased systematic approach. See DEQ Standard Operating Procedure manual for benthic Chl*a* (Montana Department of Environmental Quality, 2011).

observed in the Elk Creek site. Elk Creek's average summer TP matches the 85th percentile of the aggregate reference distribution (**Table 3-11A**).

Conclusion

Equations relating benthic algal Chl*a* to total nutrients (Dodds et al., 1997, Equation 17; Dodds et al., 2006; Equation 19) were used to calculate TN levels that would maintain 125 and 150 mg Chl*a*/m^2 benthic algae, respectively, based on a TP concentration of 33 µg/L (about the 90th percentile of reference [**Tables 3-11A, B**] and a bit higher than the average concentration observed in the Elk Creek reference site). These equations resulted in TN concentrations ranging from 439 to 1,125 µg TN/L. Also, we bore in mind the fact that nutrient-benthic Chl*a* regressions have saturation breakpoints at 27-62 µg TP/L and between 367-602 µg/L for TN (Dodds et al., 2006). And, as for other ecoregions, we took into account the TN:TP ratio of the region, which is about 13:1 (suggests slight P limitation of benthic algae). **We recommend 33 µg TP/L and 440 µg TN/L as criteria for the transitional ecoregions 43s, 43t, 43u, 43v, and part of 43o**. The TN criterion is within the range provided by the equations of Dodds et al. (2006), and within the range of saturation thresholds provided by the same authors. Together, the TN and TP criteria provide an N:P ratio of 13:1 (which matches the ratio in the regions reference sites).

4.0 REACH-SPECIFIC NUMERIC NUTRIENT CRITERIA RECOMMENDATIONS

In **Section 3.0**, ecoregions were used as the ecologically-based system for segregating nutrient criteria for different geographic zones. However, The Department recognizes that within each ecoregional zone there are streams with unique characteristics where numeric nutrient criteria must be considered on a case-by-case basis. Conditions that could render the ecoregional criteria inappropriate include, for example, the presence of a large dam-regulated lake or reservoir upstream[12], or the upstream influence of a level-IV ecoregion known to have elevated TP concentrations. A few cases have already been identified, and these are presented here. Readers should note that outside of the specified reaches described here, the ecoregion-wide criteria would apply. The Department recognizes that other reach-specific exceptions to the ecoregional criteria may be identified in the future; these can be addressed on a case-by-case basis going forward.

4.1 FLINT CREEK: FROM THE GEORGETOWN LAKE OUTLET TO THE BOUNDARY OF ECOREGION 17AK AT LATITUDE 46.4002, LONGITUDE -113.3055

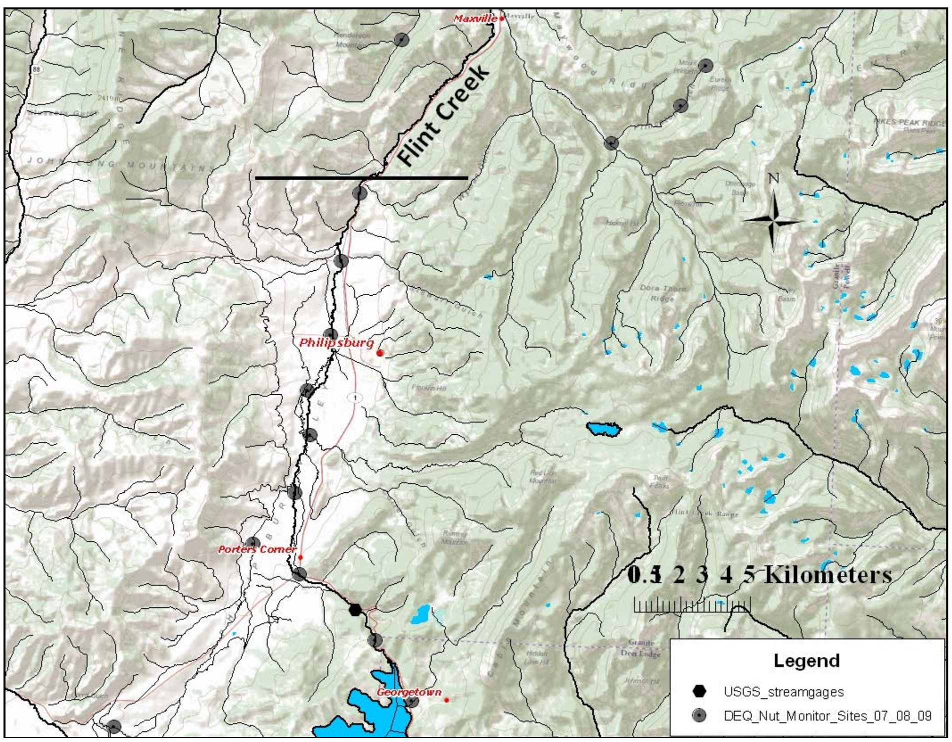

Figure 4-1. Map showing the Flint Creek watershed below the Georgetown Lake outlet.
The criteria presented here would apply from the lake outlet downstream to the black horizontal line, which is the boundary of ecoregion 17ak (Deer Lodge-Philipsburg-Avon Grassy Intermontane Hills and Valleys).

[12] When it comes to reservoirs and dam-regulated lakes, specific state laws must be considered. Conditions resulting from the reasonable operation of dams on July 1, 1971 are natural (§75-5-306[2], MCA). There may exist reasonably operated dams that, due to the nature of the water releases, characteristics of the reservoir, etc., result in nutrient concentrations (and possibly benthic algal densities) that are higher than the ecoregionally-based criteria recommended. These situations will generally be considered by the Department on a case-by-case basis.

Recommended Numeric Criteria
Total Phosphorus: **72 µg TP/L**
Total Nitrogen: **500 µg TN/L**

N:P Ratio of criteria: **7:1**
N:P Ratio of Flint Creek's Water Source: **9:1** (Redfield N:P ratio = 7:1)

Table 4-1. Descriptive Statistics for TN and TP concentrations in Flint Creek just Below Georgetown Lake Outlet (July through September).

		Nutrient Concentration (µg/L)					
				Conc. at given Percentile			
Nutrient	Number of Samples	Min	Max	25th	(Median)50th	75th	90th
TN	15	75	1200	239	340	419	585
TP	18	5.0	161	19	36	72	99

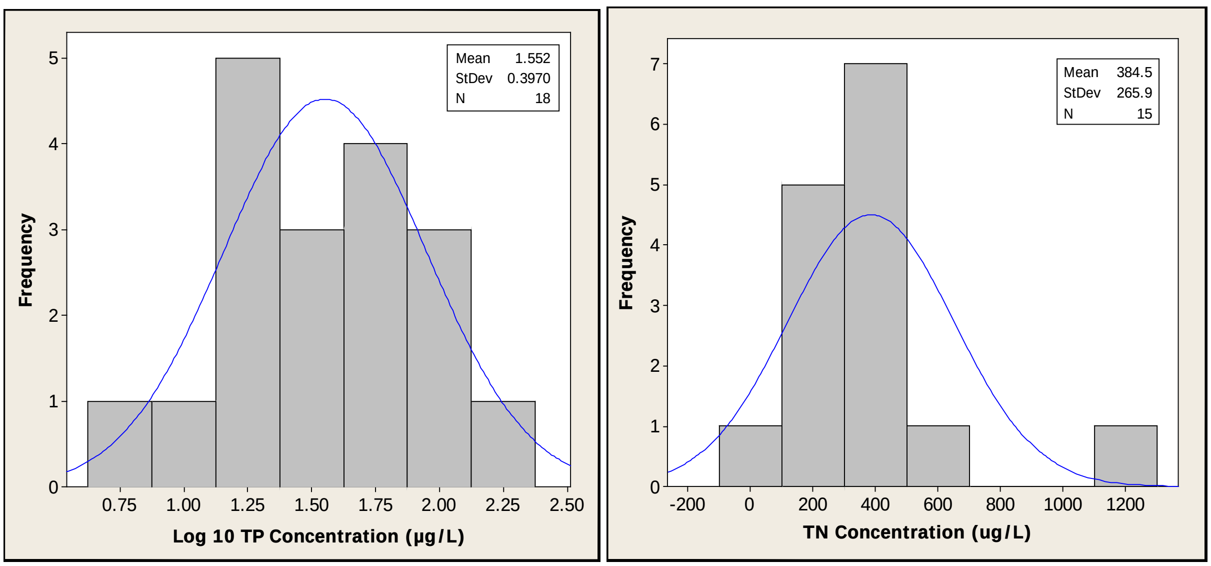

Figure 4-2. Nutrient concentrations observed in Flint Creek just downstream of the point where water exits Georgetown Lake through the dam.
Data were collected during the Growing Season (July 1-September 30).

Discussion of the upper Flint Creek Nutrient Criteria
Assessments of Georgetown Lake by the Department in the late 1990s indicated that the lake was fully supporting its beneficial uses. The lake has high levels of internal nutrient loading, particularly for phosphorus, and nutrient concentrations in the hypolimnion can become quite elevated especially during periods when hypolimnetic dissolved oxygen becomes low (Mortimer, 1941). The lake's dam forms the headwaters of Flint Creek and the stream receives water from the lake through a 36 inch diameter pipe. Flow into the pipe is controlled by a slide gate at the upstream end of the pipe on the dam. Data collected by the Department indicates that, at times, the intake intersects the lake's hypolimnion and can introduce elevated nutrients to the stream below. State law requires that reasonable dam operations be considered natural. Therefore we consider the dam operation effects when we developed nutrient criteria for upper Flint Creek.

Table 4-1 and **Figure 4-2** show nutrient concentrations measured in Flint Creek very near to where water comes out of Georgetown dam. These data (collected by the Department and others) were collected between 2005 and 2009 during the July-September period. The data show that the ambient concentrations coming out of the lake are, during the summer when nutrient criteria apply, elevated compared to natural background for the Middle Rockies (**Tables 3-1A, B**). The extent to which these conditions persist downstream was also evaluated. This was difficult to determine precisely, but the data suggest that by Flint Creek station 9 the stream has returned to "normal" Middle Rockies nutrient concentrations. (Station 9 is the gray dot immediately south of the horizontal line dividing Flint Creek in **Figure 4-1.**) Coincidently, station 9 is very near the boundary of ecoregion 17ak so, for ease, we have set the termination point of Flint Creek's reach-specific criteria there.

Equations relating benthic algal Chl*a* to total nutrients (Dodds et al., 1997, Equation 17; Dodds et al., 2006; Equation 19) were used to calculate TN levels that would maintain 125 and 150 mg Chl*a*/m^2 benthic algae, respectively, based on a TP concentration of 72 µg/L (equal to the 75^{th} percentile of the data for Flint Creek just below Georgetown Lake). These equations resulted in TN concentrations ranging from 290 to 637 µg TN/L (290-380 µg TN/L @ 125 mg Chl*a*/m^2, and 394-637 µg TN/L @ 150 mg Chl*a*/m^2). Also, we considered that nutrient-benthic Chl*a* regressions have breakpoints at 27-62 µg TP/L and between 367-602 µg/L for TN (Dodds et al., 2006). (Because of the location of the water out take from Georgetown Lake, Flint Creek starts with TP concentrations already above saturation.) And, we took into account the TN:TP ratio of water in Flint Creek just below the dam, which is about 9:1 (still in the co-limitation range).

<u>**Conclusion**</u>

We recommend 72 µg TP/L and 500 µg TN/L as criteria for the reach of Flint Creek between Georgetown Lake dam and the boundary of ecoregion 17ak, which is located at 46.4002 latitude, -113.3055 longitude. The TN criterion matches the 88^{th} percentile of the water-quality data coming out of the dam into Flint Creek. The TN concentrations calculated for 125 mg Chl*a*/m^2 could not be consistently achieved without requiring that the Georgetown Lake outtake be raised above the level of the hypolimnion (**Table 4-1**). 500 µg TN/L is within the range provided by the equations of Dodds et al. (2006) for 150 mg Chl*a*/m^2, and within the range of saturation thresholds provided by the same authors. Together, the TN and TP criteria provide an N:P ratio of 7:1 (at Redfield). **The benthic algal biomass criterion for this region is set at 150 mg Chl*a*/m^2 to account for the dam-elevated nutrient levels, with a corresponding AFDM value equal to 45 g/m^2.**

4.2 Bozeman Creek, Hyalite Creek, and East Gallatin River

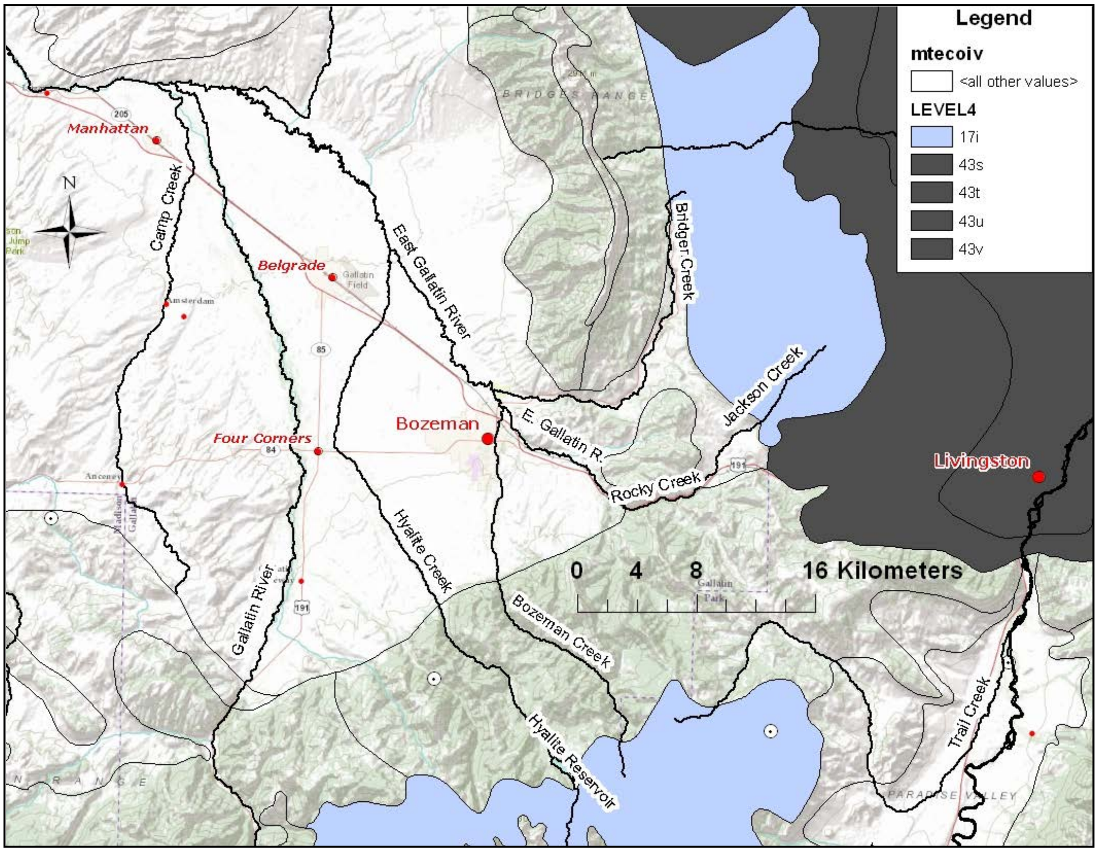

Figure 4-3. Map showing the East Gallatin River watershed, including Bozeman Creek and Hyalite Creek.
Blue shaded areas denote the level IV ecoregion 17i (Absaroka-Gallatin Volcanic Mountains).

__Recommended Numeric Criteria__

Table 4-2. Recommended Criteria for Reaches of Bozeman Creek, Hyalite Creek, and the East Gallatin River.

Stream Name	Reach Boundaries	TP Criterion (µg/L)	TN Criterion (µg/L)	TN:TP Ratio	Benthic Algal Biomass Criterion
Bozeman Creek	Headwaters to Forest Service Boundary (45.5833, -111.0184)	105	250	2:1	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Bozeman Creek	Forest Service Boundary (45.5833, -111.0184) to mouth at East Gallatin River	76	270	4:1	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Hyalite Creek	Headwaters to Forest Service Boundary (45.5833,-111.0835)	105	250	2:1	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
Hyalite Creek	Forest Service Boundary (45.5833,-111.0835) to mouth at East Gallatin River	90	260	3:1	125 mg Chl*a*/m^2 and 35 g AFDM/m^2

Table 4-2. Recommended Criteria for Reaches of Bozeman Creek, Hyalite Creek, and the East Gallatin River.

Stream Name	Reach Boundaries	TP Criterion (µg/L)	TN Criterion (µg/L)	TN:TP Ratio	Benthic Algal Biomass Criterion
East Gallatin River	Reach of East Gallatin River between Bozeman Creek and Bridger Creek confluences	50	290	6:1	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
East Gallatin River	Reach of East Gallatin River between Bridger Creek and Hyalite Creek confluences	40	300	8:1	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
East Gallatin River	Reach of East Gallatin River between Hyalite Creek and Smith Creek confluences	60	290	5:1	125 mg Chl*a*/m^2 and 35 g AFDM/m^2
East Gallatin River	Reach of East Gallatin River from Smith Creek confluence to the mouth (Gallatin River)	40	300	8:1	125 mg Chl*a*/m^2 and 35 g AFDM/m^2

Discussion of the Nutrient Criteria for Bozeman Creek, Hyalite Creek, and the East Gallatin River

In **Section 3.1.1**, we recommended TP and TN criteria specific to the Absaroka-Gallatin-Volcanic Mountains (17i), a level IV ecoregion with naturally elevated phosphorus concentrations. 'Elevated' means that the phosphorus levels in the ecoregion's reference streams were higher than the Middle Rockies (17) as a whole, and are naturally higher than concentrations that dose-response studies (phosphorus as cause, impact to stream beneficial use as effect) applicable to western Montana indicate are protective of beneficial uses.

The Hyalite Creek and Bozeman Creek watersheds contain parts of 17i, have documented elevated TP concentrations in surface water, and mapped Phosphoria formations within their boundaries (United States Geological Survey, 1951). Hyalite Creek and Bozeman Creek are in adjoining drainages and flow northward before joining the East Gallatin River (**Figure 4-3**). Bozeman Creek flows into the East Gallatin River at Bozeman, MT and Hyalite Creek joins the East Gallatin River northeast of Belgrade, MT. The headwaters of Jackson and Bridger creeks also fall within 17i, but that particular area does not have identified geologic sources of phosphorus or water quality data that suggest elevated phosphorus concentrations in surface water, and are not included in this discussion.

Reach-specific Methods

Nutrient data at the 75th percentile of reference for the Absaroka-Gallatin-Volcanic Mountains (17i) and the Middle Rockies (17) were used to determine the potential natural background of streams that flow through both ecoregions, and for waterbodies that receive drainage from both ecoregions (**Table 4-3**). Relative flow contributions were calculated from available discharge data from the USGS and from flow sampling projects conducted by the Department and its contractors. These flow estimates were used to determine the relative contribution from each ecoregional zone and, in turn, determine the potential natural background nutrient concentrations of each stream or stream segment using the following equation:

$$NB_{\mathbf{NEW}} = \frac{(NB_1 * Q_1) + (NB_2 * Q_2)}{Q_1 + Q_2}$$

Where NB_1 is the nutrient concentration (either N or P; µg/L) at the 75^{th} percentile of the reference sites for ecoregion 17i, NB_2 is the nutrient concentration (either N or P; µg/L) at the 75^{th} percentile of the reference sites for ecoregion 17 (Middle Rockies), Q1 and Q2 are the average summer flows (L/sec) that can be allocated to each ecoregional zone, and $NB_{\mathbf{NEW}}$ is the calculated natural-background nutrient concentration (µg/L) for the stream after having accounted for the mixing of the two water sources.

If the calculated natural background concentration (NB_{new}) in a given stream was equal to or greater than the recommended N or P criteria for the ecoregion in which the stream resides, a site-specific analysis was used to calculate the new criterion based on the estimated flow contributions from the different ecoregions. The new criterion was then derived using the mixing equation given above and using the draft ecoregional criteria (**Table 4-3**).

Table 4-3. Ecoregion-specific Reference Conditions and Numeric Nutrient Criteria for TN and TP. Data are from the all-observations dataset after applying the Brillouin Evenness Index.

	75th percentile - Reference Condition		Draft Numeric Nutrient Criteria	
	TN	**TP**	**TN**	**TP**
Level III Middle Rockies	141	20	300	30
Level IV Absaroka-Gallatin-Volcanics	100	105	250	105

All values are in µg/L

For example, in Bozeman Creek discharge records established that 63.4% of the flow at the mouth (1313.86 L/sec) originates upstream of the forest boundary (green area **in Figure 4-3**) where ecoregion 17i's TP concentrations are above the natural background for the Middle Rockies ecoregion. The balance of flow (36.6%; 481 L/sec) originates from below the forest boundary and is therefore associated with the Middle Rockies. Natural background (NB) for TP was calculated as:

$$([105 \text{ µg TP/L} * 833 \text{ L/sec}] + [20 \text{ µg TP/L} * 481 \text{ L/sec}]) \div (833 + 481 \text{ L/sec}) = 74 \text{ µg TP/L}$$

Because 74 µg TP/L exceeds the Middle Rockies criterion of 30 µg TP/L, a reach-specific criterion was then calculated for TP using the ecoregional numeric criteria:

$$([105 \text{ µg TP/L} * 833 \text{ L/sec}] + [30 \text{ µg TP/L} * 481 \text{ L/sec}]) \div (833 + 481 \text{ L/sec}) = 76 \text{ µg TP/L}$$

Criteria for reaches further downstream are then a function of concentrations and the proportion of flow coming from the upstream reach and the concentrations and flow from the tributary that demarcates the upper bound of the new reach. As before, it is a two-step process where estimated natural background is first calculated, and if the result exceeds the local ecoregional criterion, a reach-specific criterion is determined as a function of the criteria already derived for the two upstream waterbodies. This process can be carried downstream as far as needed. For example, for the reach "East Gallatin River between Hyalite Cr and Smith Cr" the TP criterion was calculated as follows:

$$(80 \text{ µg TP/L} * 0.325) + (30 \text{ µg TP/L} * 0.675) = 46 \text{ µg/ TP/L}$$

Where 80 µg/TP and 0.325 are the calculated natural background for lower Hyalite Creek and its proportional contribution to flow in the new reach, respectively, and 30 µg TP/L is the calculated natural background concentration for the East Gallatin River just upstream of Hyalite Creek and 0.675 is its proportion of flow contribution to the new reach (**Table 4-4**). Since the calculated value of 46 µg TP/L exceeds the Middle Rockies regional criterion of 30 µg TP/L, the reach-specific criterion is then calculated:

(90 µg TP/L * 0.325) + (40 µg TP/L * 0.675) = 56.3 µg/ TP/L (rounds to 60 µg TP/L).

Results are shown below in **Table 4-4** for a subset of stream reaches in the area.

Table 4-4. Total Phosphorus Natural Background and Derived Nutrient Criteria for Example Stream and River Reaches in the East Gallatin River Watershed.

	Bozeman Creek (Forest Service boundary to mouth)	East Gallatin R. between Bozeman and Bridger Creeks	East Gallatin R. between Bridger and Hyalite Creeks	Hyalite Creek (Forest Service boundary to mouth)	East Gallatin R. between Hyalite Cr and Smith Cr
Natural Background	74	40	30	80	50
Reach Criterion	76	50	40	90	60

All values are in µg/L

Total phosphorus concentrations are directly affected by natural sources from ecoregion 17i in the Hyalite and Bozeman creek drainages (**Table 4-3**). Natural background for TP is at or above the numeric standard for the Middle Rockies ecoregion in every reach downstream of the phosphorus source area.

Data collected in 2008 and 2009 below the Bridger Creek confluence with the East Gallatin River but above the City of Bozeman WWTP discharge (*n*=5) had a mean of 22 µg TP/L with a maximum of 26 µg TP/L. These data generally support the calculation in **Table 4-4** where it was estimated that natural background for the reach in question would not be above 30 µg TP/L.

For waterbodies receiving significant flows from ecoregions with natural sources of phosphorus, adjusted downstream criteria for TN may be slightly lower based on the same equations and process described above (**Table 4-5**).

Table 4-5. Total Nitrogen Natural Background and Derived Criteria for Example Stream and River Reaches in the East Gallatin River Watershed.

Total Nitrogen	Bozeman Creek (Forest Service boundary to mouth)	East Gallatin R. between Bozeman and Bridger Creeks	East Gallatin R. between Bridger and Hyalite Creeks	Hyalite Creek (Forest Service boundary to mouth)	East Gallatin R. between Hyalite Cr and Smith Cr
Natural Background	120	130	140	110	130
Reach Criterion	270	290	300	260	290

All values are in µg/L

Note that for Bozeman and Hyalite creeks, the criteria applicable to ecoregion 17i (105 µg TP/L and 250 µg TN/L) apply to those streams from their respective headwaters down to the Forest Service boundary.

<u>Conclusion</u>

Equations relating benthic algal Chl*a* to total nutrients (Dodds et al., 1997, Equation 17; Dodds et al., 2006; Equation 19) were used to calculate the benthic Chl*a* biomass that would occur at the criteria levels shown for the stream and river reaches shown in **Table 4-2**. In all cases, benthic algae were maintained at ≤ 125 mg Chl*a*/m^2, therefore that value (and the accompanying AFDM value) is an appropriate and realistic level for these stream segments. The nutrient criteria are adequate to protect the coldwater fisheries use by assuring that dissolved oxygen levels always remains above standards at all times.

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APPENDIX A – PEER-REVIEW COMMENTS AND RESPONSES

Montana Department of
ENVIRONMENTAL QUALITY

MEMO

1520 E 6th Avenue
PO Box 200901
Helena MT 59620-0901
(406) 444-6697 • FAX: (406) 444-3836

To: Tina Laidlaw, U.S Environmental Protection Agency

CC: Eric Urban, Head, Water Quality Standards Section

From: Michael Suplee, Ph.D., Water Quality Standards Section; Vicki Watson, Ph.D., University of Montana

Date: 5/9/2013

RE: MT DEQ's response to peer-review comments on "Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers: Addendum 1"

Reviews were received by three anonymous peer reviewers on the document referenced above in August, 2012. The reviewers were selected by the U.S. EPA in conjunction with the Nutrient Scientific Technical Exchange Partnership & Support (NSTEPS) service. One of the services NSTEPS provides is review of state-developed numeric nutrient criteria.

Section 1.0 below addresses comments that were common to two or all three reviewers; MT DEQ's response is provided in each case. **Section 2.0** lists salient comments from individual reviewers. **Section 3.0** summarizes changes to the "Addendum 1" document[1] that will be made as a result of reviewers' comments. Some comments were minor or editorial in nature and these have simply been addressed during the finalization of the document.

1.0 Comments from Peer Reviewers Addressing EPA's Six Core Questions

EPA posed six questions to the reviewers. The first queried their overall impression of the approach MT DEQ took to derive the numeric nutrient criteria. There was universal agreement among the three reviewers that the approach taken was thorough, scientifically sound, and an effective use of available and relevant information. There were, of course, concerns and recommendations as well. The five remaining EPA questions are addressed in each of the sections below (**Section 1.1** to **Section 1.5**) and most of the reviewer's comments/concerns are covered in these sections. **Sections 1.6 and 1.7** address other issues raised by the peer reviewers.

[1] The draft document was called "Addendum 1" because we considered it an extension of methods and ideas put forth in Suplee et al. (2008). However, enough material has changed and the document is now sufficiently stand-alone that in final form it has been named "Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers: Update 1".

1.1 Concern the MT DEQ has not Provided Nutrient Criteria Recommendations for the Level IV Ecoregion "River Breaks"

Two reviewers were concerned that MT DEQ did not provide draft criteria for this ecoregion. They wanted to see more reference sites, recommended that MT DEQ discuss soluble nutrients from the ecoregion's reference sites, and discuss the potential impact on downstream uses if no criteria were adopted in this area. A third reviewer was apparently very familiar with western plains environments, and understood our reasoning, but was still concerned about downstream use impacts.

RESPONSE: The basic tenant of MT DEQ's approach is to apply appropriate stressor-response studies to a region and then compare the harm-to-use thresholds derived from the studies to the reference distribution (Suplee et al., 2007). Although there are just eight reference sites in the River Breaks, it is not so small a dataset as to preclude reasonable comparisons to dose-response studies. Compiling the reference data by site medians (see discussion on this topic in **Section 1.6** below) did not substantially alter the plains region dose-response-to-reference matches (equal to the 48^{th} percentile for the median dataset and the 53^{rd} percentile for the all-observations dataset, for TP; 60^{th} and the 53^{rd},respectively, for TN). These data show, regardless of how the reference data are summarized, that harm-to-use concentrations applicable to the plains align with nutrient concentrations which are about average in the River Breaks' reference streams (i.e., River Breaks streams are naturally eutrophied or may have already responded to global increases in nitrogen loading [Vitousek et al., 1997]).

Per reviewers' recommendations, we have included soluble nutrients in the final report for the River Breaks. In relation to stream algal growth, it results that soluble nutrients in the River Breaks reference streams are already saturated, or are nearly so. For NO_2+NO_3, the 75^{th} percentile of River Breaks reference streams was 241 or 631 µg N/L (median or all-observations datasets, respectively). Rier and Stevenson (2006) show there is little peak algal biomass increase above 308 µg soluble nitrogen per liter (and peak biomass may actually be saturated closer to 250 µg DIN/L). As such, River Breaks reference sites are often saturated with soluble nitrogen, which is the nutrient most likely to be added to these streams if future development were to occur. Soluble P concentrations are also high. At the 75^{th} percentile of reference SRP is 18 or 20 µg P/L (median vs. all-observations datasets, respectively) and therefore these low-gradient reference streams are often P saturated for peak algal biomass, or nearly so (Horner et al., 1983; Bothwell, 1989).

The absence of numeric nutrient standards in the River Breaks does not mean there will be no nutrient controls whatsoever applied to new permitted sources. Aquatic life ammonia standards still apply year-round. Median pH in the River Breaks reference streams is 8.8, and with typical summer temperatures of 20 to 25°C, the ammonia criterion would be about 360 µg NH_{3+4}-N/L (DEQ-7, 2012) and would provide protection from the toxic effects of ammonia on early fish life stages. If all this ammonia were oxidized to nitrate the resulting nitrate concentration would be well within the nitrate range observed in the River Breaks reference sites. The human health standards of 1.0 mg NO_2-N/L and 10 mg NO_3-N/L would apply year round as well. Thus, Montana's existing water quality standards would preclude the River Breaks from becoming an 'industrial dumping ground', a concern expressed by one reviewer.

Downstream uses will be addressed in permitting situations via application of nondegradation. The River Breaks ecoregion basically drains directly into the mainstem Missouri and Yellowstone rivers. In spite of the elevated nitrates and total N and P coming from the River Breaks, MT DEQ has not observed nutrient problems in the lower Yellowstone River, i.e., algal levels at unacceptable levels or DO and pH that violate state quality standards. Summertime concentrations in the Yellowstone River near Glendive (in the heart of River Breaks country) during low-flow years average 490 µg TN/L and 55 µg TP/L, and are well below our recommended numeric nutrient criteria for the lower Yellowstone River during low flow(815 µg TN/L and 95 µg TP/L; Flynn and Suplee, 2013). In establishing any permit which would allow an N or P discharge that is likely to reach the Yellowstone or Missouri River, nondegradation would be considered.

We conclude that there is no scientifically-defensible way to derive numeric nutrient criteria for the control of eutrophication for streams of the River Breaks. The streams are highly turbid, flashy, have low levels of benthic algae and macrophytes, and have soluble nutrient concentrations at levels that saturate algal growth much of the time. Other MT DEQ programs will address impacts to downstream uses. We will not be recommending nutrient criteria for these streams in the final report.

1.2 Peer Reviewers' Views Concerning the Allowable 20% Exceedence Rate Associated with the Criteria (Pertains to Assessment Methodology[2])

RESPONSE: This topic closely ties to the topic in **Section 1.5** below, and is addressed there.

1.3 Peer Reviewers' Comments on MT DEQ's Use of Benthic Chlorophyll *a*, How the Chlorophyll *a* Threshold (125 mg Chl*a*/m^2) was Derived, and Thoughts on Other Biological Measurements Used to Support Eutrophication Assessment

Two reviewers found the use of benthic chlorophyll *a* to be an excellent tool for assessing eutrophication, while the third did not like it. Derivation of the chlorophyll *a* thresholds were considered appropriate although two reviewers felt that the threshold of 125 mg Chl*a*/m^2 may be too close to the harm-to-use threshold. (The third reviewer found it acceptable.) One reviewer notes that macroinvertebrates are a poor indicator of eutrophication.

RESPONSE: MT DEQ has had good success with measuring benthic chlorophyll *a* and does not believe the concerns of one reviewer (it's too variable, affected by grazers) apply to the physiographic regions where it is used. Note also that MT DEQ collects benthic ash free dry mass (as g/m^2), which can provide good indication of heavy benthic algal growth even if chlorophyll *a* levels have declined due to senescence. As pointed out by one reviewer, MT DEQ has a long tradition of measuring benthic algae density and diatom taxa and both of these are main features of the nutrient assessment method. We agree that macroinvertebrates are not an ideal tool for pinpointing eutrophication problems, which is why they are used secondarily, i.e., only after the better tools (benthic chlorophyll *a*, diatom metrics) have already been played out. At a recent conference of academic experts on stream ecology (April 16-18, 2013, Washington, D.C.), which one of the authors was fortunate enough to attend, there was wide agreement that macroinvertebrates have generally poor predictive power for eutrophication assessment.

MT DEQ had extensive internal discussion about where to set the benthic algae density after the results from the dosing study (Appendix B, Suplee and Sada de Suplee, 2011) showed that average levels of 127

[2] MT DEQ's assessment methodology for assessing eutrophication in wadeable streams (Suplee and Sada de Suplee, 2011) was completely revised in 2009-2010, went through public comment (including EPA review), and was finalized prior to the time that it was provided to the peer reviewers here.

mg Chl*a*/m^2 could result in seasonal DO problems. To inform that discussion, a mechanistic model was built to simulate the DO impact observed in the dosing study and the model showed that higher gradient streams in western Montana would not develop low DO due to their reaeration; lower gradient streams, however, would be impacted. In streams with good re-aeration, therefore, harm-to-use would not occur until 150 mg Chla/m^2 (the recreational threshold). To avoid creating an overly-complex application of the algae threshold, involving not only ecoregions but different beneficial uses and different benthic algae levels for different stream gradients, it was decided that one algae threshold would be established (125 mg Chl*a*/m^2) that should be largely protective of both aquatic life and recreation. Monitoring staff with experience using the thresholds understood the rationale and indicated that they were comfortable with it because, in most cases, streams' algae densities are well below or well above the thresholds, precluding borderline decisions. MT DEQ is measuring diel DO concentrations much more frequently now and will continue to evaluate the 125 mg Chl*a*/m^2 threshold; it can be readjusted if needed in the future.

1.4 Peer Reviewers' Assessment of MT DEQ's Reach-specific Nutrient-criteria Derivation Method

All three reviewers supported the approach taken.

RESPONSE: We are delighted that all three peer reviewers were very supportive of the approach that was taken.

1.5 Peer Reviewers' Views Concerning the use of the Binomial Test, its 20% Allowable Exceedence Rate, and the Student's T-test (Pertains to Assessment Methodology)

The reviewers' main questions and thoughts/observations pertaining to these subjects are summarized as follows. Questions: (a) By using an allowable exceedence rate of 20% and an effect size of 15%, is the exact binomial testing whether 35% of observations must exceed the criterion to be considered non-compliant? (b) Is it appropriate to use an effect size in the T-test? Thoughts/observations: (a)The T-test is a parametric test with assumptions of a normal distribution which are not the norm for the datasets being evaluated, and so it will be less likely to detect a difference in the mean of a nutrient dataset relative to the criterion, and (b) for the T-test to establish non-compliance, the average concentration of a test stream would need to be substantially above the 75th percentile of reference (the reviewer's presumed level at which protection of uses is assured) and this is under protective.

In addition, there was general confusion among reviewers on how MT DEQ has defined an observation when assessing nutrients, how reaches are delineated, and how statistical tests and biological information all fit together in the final assessment.

RESPONSE: MT DEQ's statistical assessment of nutrient concentrations in a stream segment can be reduced to two simple ideas: (1) a test (exact binomial) to determine the proportion of samples that exceed the criterion, and (2) a test (one sample Student's T-test for the mean) to help to identify when the average nutrient concentration has been pulled above the criterion, which may result because most samples are above the criterion or because just a few high outliers are. Each test is discussed below, followed by an overall conclusion.

<u>Binomial Test</u>. Excellent empirical data were available to MT DEQ to derive the allowable exceedence rate used in the test (more on this in a moment). Besides the all-important exceedence rate, MT DEQ had to give consideration to other factors for statistical testing including the realistic number of independent samples that could be collected in a stream reach (restrained by cost/time), and the desire to balance type I and II error rates, i.e., give roughly equal weight to the importance of error of over-regulation vs. failure to protect the environment (Mapstone, 1995). Realistic sample sizes were about 10 to 15, and as such it was impossible to have alpha and beta error both around 0.05 (95% confidence) because sample size would then need to be around 75. So MT DEQ opted for less confidence, i.e., alpha and beta error rates both ≈0.25 (75% confidence).

Allowable exceedence rate (number of samples allowed above the criterion while assuring the river supports beneficial uses) was empirically derived from long-term work on the Clark Fork River—a river where adopted nutrient standards are virtually identical to those proposed for western Montana streams. **The Clark Fork River analysis shows that a defensible criteria exceedence rate could range from 5-31%.** Twenty percent was identified as the most reasonable value. <u>To date, MT DEQ has not found or been made aware of another dataset by which an allowable exceedence rate for numeric nutrient criteria could be determined</u>. Because of this, MT DEQ will continue to use the 20% exceedence rate.

MT DEQ uses a 15% effect size. By establishing 15% effect size, MT DEQ is saying that this is the range of true exceedence rates where the consequence of decision errors is relatively minor. As a point of comparison, if there were a pollutant for which the allowable exceedence rate is set at 10% and it is known that virtually no impact will occur at 9% exceedence, but terrible impacts occur at 11% exceedence, then the effect size would have to be set very close to zero, because the consequence of decision error is huge. And as a result, very large numbers of samples may need to be collected to discern with accuracy that fine a cut on the exceedence rate. But for nutrients, the state-of-the science is still limited and what we do know tells us there is a fairly wide range (5-31%) where decision error impacts are minor; MT DEQ addressed this by selecting a somewhat wide (15%) effect size.

In the binomial—with 20% allowable exceedence rate and 15% effect size—MT DEQ is establishing that streams with <5% exceedence will always PASS (be found compliant with) the binomial test, and streams with >35% exceedence will always FAIL (be found non-compliant with) the binomial. (The reviewer is correct that the 20% exceedence rate and 15% effect size are additive.) Streams falling in between will sometimes PASS, sometimes FAIL (depends on **n**). It could be reasonably argued that 35% exceedence is too high, but sample-size reality then enters the picture: if we lower the effect size to 10%, i.e. streams with 30% exceedence rate will always FAIL the binomial, we would have to collect 25 samples to roughly balance alpha and beta error; too many samples to institute for routine stream nutrient monitoring. Other combinations of exceedance rates and effect sizes within defensible ranges (and again balancing alpha, beta error) also led to **n**'s in the low 20s or higher. In the end, MT DEQ settled on the exceedence rate and effect size we are currently using. However, note that in borderline situations (i.e., the assessment decision is not clear) MT DEQ will collect more data, and may very well end up with sample sizes closer to 20.

<u>T-test.</u> Per the reviewer's question, no, effect size is <u>not</u> included in the T-test. MT DEQ believes that at this point the EPA-recommended T-test is satisfactory for its purpose within the assessment methodology. It is robust against moderate deviations from normality (and many of the small datasets that are considered are essentially normally distributed). The reviewer is correct that the T-test loses power when datasets are highly skewed (and some of the datasets are skewed). But in actual cases where there are a few very large outliers among the 12 or so samples (this is a common scenario), the T-test still FAILS (indicates non-compliance, as we would want it to) even if the exact test statistics (p value, etc.) may not be particularly accurate. Staff who routinely carry out eutrophication assessments have expressed that the T-test results are largely in alignment with the totality of information provided by the binomial and biological measurements.

Regarding the idea that the average concentration in a test site would have to be much greater than the 75^{th} of reference in order to FAIL the T-test, two points can be made. (1) The same reviewer stated that nutrient criteria are best if based upon dose-response studies. MT DEQ has found that dose-response studies often show concentrations >75^{th} of reference are protective of legally-defined beneficial uses (Suplee et al., 2007; Suplee et al., 2008). Thus, PASSING the T-test because the average concentration in a test site is >75^{th} percentile of reference is not necessarily under protective. (2) MT DEQ uses a different test hypothesis depending on the stream's 303(d) listing history for nutrients. Already-listed streams have the null as "stream is impaired" and the alternative as "not impaired". Thus, MT DEQ has the most control on alpha error which is defined upfront in the test. This approach is more protective.

MT DEQ Procedures and Assumptions. Regarding clear explanations of MT DEQ procedures, MT DEQ laid out the entire assessment approach and its assumptions in Suplee and Sada de Suplee (2011), including a number of examples that can be followed (see Section 3.2.4 of that document). However it appears that the final element of the method, the data-review matrix contained in the Excel spreadsheet "NtrntAssessFramework.xlsx", may not have been seen by some reviewers. Lacking this final piece would have led to confusion for sure. In any case, MT DEQ believes that "Addendum 1" (now Update 1) is not the place to detail assessment methodologies that are well covered in other documents. Going forward, Update 1 will continue to focus on nutrient criteria and their derivation.

Conclusion. The binomial test and the T-test in MT DEQ's assessment methodology will continue to be used as configured. As noted by a reviewer, MT DEQ compensates for the higher-than-ideal FAIL threshold of 35% in the binomial test by establishing different null hypothesis depending on if the stream is (or is not) already listed on the 303(d) list, by including the T-test, and by lowering the chlorophyll *a* threshold to 125 mg $Chla/m^2$ (instead of 150 mg $Chla/m^2$). As noted by another reviewer, "Some of the quibbling on these values may never be resolved (including mine), and Montana needs to use best judgment supported by its analysis and other scientific results." We couldn't agree more.

1.6 Number of Reference Sites, Manner by which MT DEQ Characterizes the Reference Condition

One reviewer felt there were too few reference data. Reviewers felt that MT DEQ's novel use of the Brillouin Evenness Index should be (at a minimum) clearly spelled out, and include the equations. One reviewer felt that the Brillouin method was "interesting", but that it did not directly address the issue of temporal pseudoreplication which may arise in repeated measurements of nutrients at reference sites. The reviewer recommended a more traditional approach to summarize reference data, whereby each reference site's nitrogen and phosphorus observations are reduced to a site median, and then distribution statistics on the population of medians is calculated.

RESPONSE: Regarding the number of reference sites, MT DEQ believes it has a good reference site network and has been actively identifying and sampling reference sites for the past twelve years. (Limited work was also carried twenty years ago by Bahls et al. [1992].) From 2000 to 2009 much effort went towards identifying new sites. In some parts of the state (e.g., eastern Montana) staff has gone over the landscape several times and we are at the point where few if any new sites can readily be identified. As it stands, there are 185 different reference sites across the state and all major ecoregions are represented. Because of the relatively large overall number of sites, MT DEQ management indicated that the Reference Project should focus on resampling the network rather than seeking new sites. Some level III ecoregions (e.g., Idaho Batholith) would benefit from additional sites but it is unlikely that will occur in the near future.

We agree that the Brillouin Evenness Index formula should be provided in Update 1 with an explanation of why this approach was taken. This has been included in the final report. Regarding our use of the Brillouin Evenness Index vs. site medians to summarize the reference data, we offer the following. By taking the Brillouin Evenness Index approach, MT DEQ made the assumption that each nutrient observation in the dataset was independent even if collected from the same site. The vast majority of sample observations from the reference sites were collected a month apart, and MT DEQ has shown that such samples are usually temporally independent (Appendix A.3, Suplee and Sada de Suplee, 2011). We believe the data, after application of the evenness index to assure equitable representation of each site, provide a very valuable characterization of reference condition especially when a reader wants to know the true range of nutrient observations (minimum, maximum) in Montana reference sites during baseflow.

Stakeholders from the Montana Nutrient Work Group had earlier indicated that this was important to them. And as pointed out by one reviewer, the exact manner by which reference data are summarized is not terribly important because we do not carry out inferential statistics with the data, nor are criteria tied to a specific reference percentile. We agree with the reviewer that with the approach we used we cannot assure that there is no intra-site temporal pseudoreplication, an issue discussed at length in Hurlbert (1984). In response, we have now provided two summary statistics tables for each ecoregion; the original (derived, as before, using all observations and the Brillouin Evenness Index), and a 2nd table which shows the frequency distribution (25th through 90th) based on the median nutrient concentrations from each site. We believe this approach will provide readers the maximum amount of information and will make comparison to other work easier, since reduction of site data to medians is common in the literature (e.g., Robertson et al., 2001; Wang et al., 2007; Stevenson et al., 2012).

1.7. Concern that MT DEQ has Recommended Nutrient Criteria Concentrations in some Ecoregions Beyond the Applicable Reference Distribution

Two reviewers were concerned that nutrient criteria concentrations had been set at levels beyond any single observation collected in the regional reference streams. The Northern Rockies and Idaho Batholith are good examples. Although one reviewer agreed with MT DEQ that one should not use reference condition nutrient concentrations *alone* to set criteria, at the same time the idea of setting a criterion higher than the highest observation in the regional reference sites was clearly troubling to reviewers.

RESPONSE: The reviewers comments can be summarized as (1) concentrations beyond the reference distribution 75th percentile may be linked to known harm-to-use thresholds (e.g., via benthic algae density), but they will not be protective of sensitive, low-nutrient adapted organisms, and (2) in the ecoregions with naturally-low nutrients the harm-to-use concentrations derived from the dose response studies always had a range, and MT DEQ should have picked the lower concentration threshold given that we are operating beyond the bounds of the regional reference condition. Regarding point 1, we agree with the reviewer that if stream concentrations rise to the criteria and the criteria are beyond the reference condition, some organisms—like low-nutrient diatom taxa—would be displaced. The difficulty with establishing criteria to protect microscopic organisms like this is that there is no definitive harm to the beneficial uses established in Montana law. Studies generally show that with some additional nutrients ultra-oligotrophic streams will have more of the same macroivertebrates (as evidenced by O/E scores >1.0) and more robust populations of some fish. Fish—and to a somewhat lesser degree macroinvertebrates—link directly to Montana's beneficial uses. But as Montana state law is currently written, it would be difficult to defend a criterion based on protecting low-nutrient diatoms (as suggested by one reviewer).

Regarding the 2nd point, there is definitely merit to the idea that if there are several dose-response studies for an ecoregion and the concentrations from them generally fall beyond the reference distribution, greater weight should be given to the study or studies with the lower concentrations. As a result, in the final draft we have somewhat lowered the criteria recommendations in several ecoregions where this occurred. We have still kept an eye on maintaining the reference Redfield ratio, and in some cases the final concentrations are still beyond the reference distribution, but they are closer to it.

2.0 Selected Comments from Individual Reviewers

Here are important comments unique to individual reviewers.

2.2 How a Stream Reach is Delineated (Pertains to Assessment Methodology)

One reviewer was concerned that the flexible manner by which a stream reach can be delineated could make it difficult for any stream reach to ever be found impacted by nutrients, because data from impacted sites would be lumped with data from unimpacted sites and would, in effect, dilute the signal. The reviewer also noted that because of the flexibility in establishing assessment reaches, intentional manipulation of reach lengths could drive the outcome.

RESPONSE: The potential for unethical actions to manipulate analysis outcomes is always present in assessment work, but the high level of professionalism in the MT DEQ staff is such that this issue has not arisen. Regarding the flexibility of assessment reach lengths, this was done purposefully as discussed in detail in Appendix A.2.0 of Suplee and Sada de Suplee (2011). A basic assumption of the method is that reaches should be relatively homogenous in time (over the past 10 years) and in space, and observations collected within the reach should be largely independent. One reviewer was concerned about sample independence but MT DEQ has demonstrated independence in similar nutrient-concentration datasets using standard statistical tests (Durbin-Watson, Rank von Neumann). From these results and earlier experience, temporal and spatial independence guidelines were defined to make sure data collection maintains sample independence to the degree possible (nutrient and biological samples have to be collected a month apart at a site, for example).

If an assessor concludes that a reach is really *not* adequately homogenous (e.g., it comprises a distinctly impacted segment and an unimpacted segment[3]) it is incumbent upon the assessor to subdivide the reach and make an independent assessment of each new segment. This stratification allows maximal precision of estimates for minimal sampling effort (Norris et al., 1992). What remains constant is the minimum number of water quality and biological samples that need to be collected in each of these new assessment reaches in order to make a final compliance decision. The reviewer seemed to suggest that fixed reach lengths, numbers of sites, etc. along streams would be better, but experience has shown that this is highly impractical in applied assessment. If MT DEQ were to carryout assessments using fixed-length reaches, results would be far more arbitrary then the approach currently found in the SOP.

3.0 Summary of Changes Resulting from the Peer Review

1. We have included soluble nutrient data in the final report (Update 1) for the River Breaks.

2. The Brillouin Evenness Index formula is provided in Update 1 with a better explanation of why this approach was taken. We have also characterized reference using median datasets, further described below in 3.

3. Reference condition within an ecoregion has been characterized by first reducing data from each reference site to a site median, then calculating distribution statistics (25th, 50th, 75th, and 90th) for the ecoregion based on the population of site medians.

4. We agree that there is merit to the idea that if there are several dose-response studies for an ecoregion and the concentrations from them are generally beyond the reference distribution, greater weight should be given to the study or studies with the lower concentrations. As a result, in the final document we have somewhat lowered the criteria recommendations in several ecoregions where nutrient concentrations are naturally very low. We have still kept an eye on the Redfield ratio of the regional reference streams and, in some cases, the final criteria recommendations are still beyond the reference distribution, but they are closer to it.

4.0 References

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[3] Temporal non-uniformity must also be considered. For example, if a new feedlot with several permit violations was built alongside the stream three years ago, the assessor would not be including data in the analysis collected six years ago.

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Review of MT Nutrient Criteria Documents.

Response to Questions:

MT – Wadeable Streams Draft Peer Review Questions

1. MDEQ is considering two approaches for the derivation of numeric nutrient criteria in wadeable streams: (1) eco-regional reference condition, and (2) regional and non-regional stressor-response studies. Compare and contrast the ability of each approach to provide a sound scientific basis for numeric nutrient criteria derivation. Please provide documentation on any identified ranges protective of aquatic life based on similar studies. If possible, please provide alternate methodologies using available data and tools, and describe the corresponding advantages and disadvantages.

Many of my specific concerns are detailed separately, following my responses to #1-6, some of which address this question. My overall assessment of the MDEQ dual approach was that it was generally well thought out and appeared to have protection of streams in mind. There were a few exceptions where it appeared the criteria were set a bit too high (well beyond the 100% of reference distribution), and I commented on those decisions and provided citations where available. However, MDEQ did a commendable job of reviewing scientific literature and applying peer-reviewed literature in support of developing defensible, numeric criteria.

2. In Section 3.6.1., Montana suggests that no nutrient criteria are needed for streams in the 1 Level IV Ecoregion within the Northwestern Great Plains: River Breaks (43c). The MDEQ rationale for this decision is: "This level IV ecoregion has highly turbid, flashy streams with naturally elevated TP and TN levels. Concentrations observed in the region's reference sites indicate that nutrient concentrations here are already naturally elevated above the harm-to-use thresholds identified for the plains region as a whole. As such, no nutrient criteria are recommended for streams within this level IV ecoregion." Please comment on whether the state has provided a sufficient scientific basis that 1) these levels are naturally elevated, 2) additional increase in nutrients would not cause harm to aquatic life, and 3) that, therefore, criteria are not needed. Is the reviewer aware of any additional information that could be provided to either support the State's assessment of natural background or that could be used to derive site specific criteria?

I struggled with this decision. I cannot concur that additional increases in nutrients would not cause harm to aquatic life, mainly because there are not sufficient data to support this conclusion. I commented that there needs to be some consideration of dissolved nutrients, or at least a thorough discussion about them relative to TN and TP. Potential sources of nutrients to these streams are likely to be primarily dissolved in form and, without knowing whether there are high levels of dissolved N and P in reference sites, I cannot determine whether inputs are likely to harm aquatic life. Stream flow is also an important consideration. Extended periods of low flow during droughts coupled with over-enrichment from anthropogenic sources of N and P would likely result in biological responses that could harm aquatic life. I also noted that there was a very wide range of TN and TP values among different streams, suggesting that even level 4 ecoregions may not sufficiently capture the variability in geology and natural nutrient concentrations. In sum, I suggest this decision needs further consideration.

3. MDEQ is proposing to allow TN and TP criteria to be exceeded 20% of the time and be considered supporting aquatic life uses. This frequency was derived based on analysis of the Clark Fork River chl-a data. Please comment on the proposed exceedance frequency and whether allowing the stated magnitudes to be exceeded 20% of the time would not result in adverse effects on aquatic life. This information is discussed in the State's Assessment Methodology.

Allowing TN and TP criteria to be exceeded 20% of the time brings with it uncertainty about both the timing and the magnitude of exceedance. For example, exceeding 2 months in a row, in the middle of summer when flow is low is quite different than exceeding two distinct periods during a wet year with higher than average runoff. Similarly, exceeding a criterion by an order of magnitude just one time would obviously have different implications to aquatic life than if the criterion were exceeded by a few parts per billion.

The use of the Student's t-test to compare means to the criterion is MDEQ's approach to considering magnitude of exceedance. Although I am encouraged that MDEQ recognizes magnitude and frequency as important, I am not certain the t-test method is optimal. I outline my concerns with the sampling method, samples, and analysis of data for this test under point #6, below.

4. MDEQ's criteria approach includes a Chl-a value of 125 mg/m^2 to be used as part of the related assessment information. Please comment on the selection of chlorophyll as the primary response variable, the derivation of the chlorophyll threshold, and its application as a statewide assessment indicator.

Benthic CHLA is a widely used indicator of nutrient over-enrichment so it is defensible for MDEQ to include it as a measurement endpoint. However, benthic CHLA is not a reliable indicator of nutrient overenrichment because it is highly variable temporally due to periodic sloughing/senescing, grazing (Taylor et al. 2012) and scouring by high flows. In two years of sampling wadeable streams in central Texas, we found benthic CHLA to be one of the least reliable indicators of nutrient enrichment when compared to periphyton carbon: CHLA ratios, CNP ratios, enzyme activity, primary and bacterial production, and species composition (Scott et al. 2008, King et al. 2009, Scott et al. 2009, Lang et al. 2012). The observed frequency of exceeding 125 mg/cm2 CHLA could be highly variable depending upon the natural flow regime of a stream, interannual variability in precipitation, and timing of site visits, even though a stream may be vulnerable to dense periodic blooms that result in harm to aquatic life.

I also found the use of piecewise regression models (Dodds et al. 2002) to infer chlorophyll a values at particular nutrient levels to be questionable for a few reasons. It did not appear the confidence limits were considered. The fitted mean value falls within a highly variable cloud of points, indicating that 125 mg/m2 is exceeded in many streams possibly as much as half the time. If the goal is to keep CHLA below 125 mg/m2 a certain percentage of the time, quantile regression splines (Anderson et al. 2008) or other nonlinear quantile regression method would more closely match the objective. For example, if the goal was to keep CHLA < 125 mg/m2 80% of the time, the TP or TN value that aligns with the lower 5% CI of the 20% quantile would be a more appropriate number. Thus, risk of exceeding 125 mg/cm2 seems to be potentially high, or at least high uncertain, given the approach to derived the TN/TP thresholds and the high variability in benthic CHLA during the growing season.

5. Section 4.0 outlines a process for determining reach-specific nutrient criteria. Please comment on MDEQ's proposed approach for deriving reach-specific values.

The rationale and methods for setting criteria for this reach seem defensible. The process was consistent with the process used among ecoregions. Overall it is hard to find many suggestions on how they could improve their approach for setting criteria in these rivers, however see my previous comments about using CHLA as a biological endpoint, the use piecewise regression models to identify TP and TN criteria, and several point of concern about the sample design and statistical methods (see #6, below)..

6. Montana is proposing to interpret the numeric criteria using the Students t-test and binomial test to determine whether a stream segment is impaired. Please comment on the State's rationale for this approach.

It is obvious MDEQ has given this process a great deal of thought. Overall I am encouraged by the level of detail in the process and what appears to be a sincere attempt to develop criteria and a process for assessing criteria that is protective of aquatic life in the waters of Montana. This section is particularly important because it describes the nuts-and-bolts of how criteria are used to assess compliance.

There are several moving parts in this process that have the potential to strongly influence the outcome of an assessment. First, the manner in which reaches are delineated is flexible such that it seemed a bit ambiguous to me. Because sample "sites" allocated within reaches are used to assess criteria, how reaches are delineated could be manipulated to influence the outcome of assessments.

The use of multiple sampling sites within a reach to assess criteria is reasonable, but the scale of nutrient overenrichment required to fail a reach seems to be quite large. For example, under low, summer flow conditions, one site within a reach could conceivably fail during both visits whereas downstream reaches pass each time. The use of multiple downstream sites, some of which could be many kilometers away, to calculate exceedance frequency and mean nutrient levels ignores the local impairment and effectively "dilutes" the problem at this location, despite the fact it could span > 1 mile of stream (minimum distances between sites was 1 mile, correct?).

Another factor is the manner in which sampling locations and repeated measurements from those locations are used in the binomial test and t-test as if each sample unit reflects a measurement from the same population. There are two levels of organization being mixed here. Spatial and temporal sample units are being thrown in together in a haphazard way that ignores the distinct components of variance. If there were a clearer definition of reaches, site locations, and sample frequency from those sites, I would feel a little less uneasy about it, but as it stands, I get the impression that reaches may differ wildly in length, number of sites per reach will thus differ, and sample frequency may also differ.

The Clark Fork example illustrates the problem: 15-20 individual CHLA samples were collected per date and each "sample" was treated as a repeated measure, when in fact these are subsamples that are nested within a single observational unit (a site? I can't follow the sampling design very well). The total CHLA "samples" were 285-333 per site over a multiple-year period, but there were far fewer sampling events than 285-333, and far fewer TN and TP "samples" as well because those were composite grab samples. There also were different numbers of "samples" taken per site within the Clark Fork reach, as well as different numbers of samples within a site among dates. This type of analysis would not likely hold up in a peer-reviewed journal because each CHLA measurement is subsample of a single observational unit (site). In sum, I'm not necessarily saying that the approach will lead to wildly inaccurate assessments but I do believe that there are better ways to account for multiple measurements within a site and multiple dates per site within a reach to arrive at an estimate of exceedance frequency.

I also do not really see this as a hypothesis testing problem, but rather a risk assessment or probability of exceedance problem. There is a burgeoning literature on misuse of hypothesis testing statistics for ecological risk assessment and environmental assessment. The use of this approach for this particular application does not strike me as ideal.

I also am not certain about appropriateness of a t-test to detect magnitude of exceedance relative to the criterion. The t-test is a normal-distribution statistic that will be less likely to detect a difference in the mean relative to the criterion when data are skewed, and skewed data (infrequent but large departures from the criterion) are exactly why the statistic is being computed in the first place. Several other methods could be considered, ranging from computing empirical confidence limits using the bootstrap, to more sophisticated Bayesian approaches where an appropriate sample distribution is used and the test computes the probability that the sample mean differs from the criterion.

Specific comments, Addendum:

p2-2: Equitability of sample representation. I agree this is an important consideration but do not understand how the evenness statistic was applied to address the problem. How was J computed, specifically in terms of the observations in the nutrient database? The data are nested by sample unit (site),with each observation representing a distinct date, correct? More detail is needed here.

Section 2.5.1. This paragraph is interesting and I don't have any particular problems with the content except that it does not seem to have any direct applicability to criteria development in Montana. How was the information from sites that were intentionally enriched with N and/or P used to support criteria development? The section ends by suggesting this information was valuable for establishing a "lower bounds" for nutrient concentrations, but how was the information used? What are "lower bounds"?

Section 2.6. This section is an important addition to the document. I think the idea that differential nutrient limitation among different algal and other microbial species is not sufficiently acknowledged in the development in numerical nutrient criteria. This section does an excellent job of describing why managing for 2 nutrients is critical. However, I think a couple of ideas are used interchangeably and might need to be distinguished a bit.

The most important reason for differential nutrient limitation is that different species have different relative N and P demands thus one may be predominantly limiting to an aggregate endpoint such as benthic chlorophyll but in most circumstances at least some species are limited by another resource. This appears to be particularly true of photoautotrophs and heterotrophic microbes growing together in a periphyton community (Scott et al. 2008, 2009, Lang et al. 2012). In this paragraph, the idea of different species being limited by different nutrients is introduced, but later is conflated with the idea of communities switching back and forth between N and P limitation. These are 2 distinct ideas and should be parsed as such.

It is also unclear what is meant here by limitation. Limitation of accrual of benthic chlorophyll or something else? There are numerous indicators of limitation that may not manifest themselves as an increase in standing stocks if other factors are controlling accumulation in the short run. Enzyme activities, in particular, may reveal dual limitation of different subsets of species in the community whereas total biomass remains unchanged with enrichment of N, P or both because of the decoupling of heterotroph and autotroph recycling of carbon, N and P. I say this mainly to encourage a more explicit definition of limitation and acknowledgment that biomass accumulation may not be a good indicator of limitation in all situations.

The discussion about Redfield ratios is fine to include, but again it seems to be lumping responses into one large bin of either N or P limited, when in fact differential limitation means that each species in an attached community of photoautotrophs and heterotrophs has a different N and P demand, hence a community-level N:P ratio target is naive and potentially dangerous. I think ratios are a lot less important than concentrations and supply rate (velocity). Nutrient criteria should emphasize maintaining concentrations that fall below levels of individual nutrients that are known to overstimulate algae and/or microbes; the ratios at those levels may or may not be near "Redfield" because it is the supply rate of ions to the cells that ultimately determines whether a nutrient is limiting to growth or other physiological process.

In sum, I like the fact that Montana is thinking about these details but am a bit concerned about some of the overgeneralizations about nutrient limitation and nutrient ratios in driving decisions to manage for both N and P. The decision to manage for N and P need not be any more complicated than the fact that differential limitation probably occurs in most stream ecosystems and thus both nutrients are likely to limit some facet of the community at any point in time.

Section 3.0

I like the introduction to this section, detailing how the criteria are organized and presented in the forthcoming pages.

Fig 3-1 is a nice illustration of the distribution of reference sites. I noticed here and in the 2005 document that reference sites are spatially contagious. Large areas within each ecoregion are largely unrepresented by reference locations whereas other areas have high densities of them. This is a common problem, given that human activities tend to be clumped and thus the remaining "good" places are also clumped, away from human activity. However, given that there is some mention of the need for Level IV ecoregional criteria in some Level III ecoregions, it would be helpful to know whether there are some level IV ecoregions that contain few or no reference sites.

Fig 3-2. Red dots are cities? Not all red dots are labeled.

Section 3-1. Middle Rockies

Again, noting the Redfield ratio in the criteria recommendations. I don't think there is sufficient justification for including this number given it was derived for marine phytoplankton (i.e. is the the N:P ratio of marine phytoplankton). I worry about other states focusing on this ratio as they plod forward in their development of criteria. Also, the ratios reported are based on mass not moles so if ratios are to be reported please specify that they are based on mass.

p3-3, last paragraph: The interpretation of the breakpoint regression is correct, but more specifically the level of chla/m2 has reached its maximum (the bottom is effectively covered in filamentous algae). The first section of the breakpoint regression line is a quasi-linear increase with quite a bit of scatter. I don't like the interpretation of this type of regression because in reality what is happening is that the growth rate of Cladophora is faster at higher nutrient levels but with sufficient N and P will nevertheless grow until most of the channel is covered or until a high flow event knocks it back. The problem with assuming that a certain level of N or P will keep chla/m2 below a certain level is that it assumes that on average there are sufficiently frequent spates/high flow events that will keep the growth in check. In low water years or very dry summers I highly suspect that any level of N and P that is sufficient to promote filamentous algae will lead to unacceptable levels of chla/m2 (e.g. see experimental results in King et al. 2009). . If the goal is keeping chla/m2 below a certain level, other variables (particularly frequency/timing of storm events or high flows) are needed to better estimate the likelihood of failing to meet biological criteria. As currently written, I think it is overly simplisitic.

p. 3-4 Conclusions: The section acknowledges that TP as low as 20 is associated with undesirable outcomes. The use of N:P ratio as is further used to support 30 ug/L as a TP criterion because it maintains a 10:1 NP ratio, consistent with reference streams. Are we to presume that 200 ug/L TN is also associated with undesirable biological consequences as well? The justification for using the ratio as a basis for choosing 30 ug/L instead of 20 ug/L based on biological responses is warranted here. I feel there is too much emphasis on ratios without sufficient scientific documentation of it being as or more important than concentration/supply rate by ion. I am particularly concerned about the repeated reference to Redfield ratios.

Section 3.1.1 Level IV Ecoregion within the Middle Rockies: Absaroka-Gallatin Volcanic Mountains (17ia). There are only 4 reference sites in this region. The 4 sites span a huge range of TP, with as little as 16 ug/L. I find it hard to find support for a numerical criterion that would allow a stream with 16 ug/L TP to increase to 105 ug/L TP. I am confident there would be biological consequences. How realistic would it be to set basin-specific criteria for this subregion, given that it is relatively small?

Another concern is the selection of 250 ug/L TN despite the fact that this exceeds the highest reference site by almost 100 ug/L. It seems that given the high levels of TP that are naturally available in many of these streams, that any, small input of N could lead to nuisance growth of algae. In this region, it would be helpful to know the dissolved N levels because I suspect that most of the TN is particulate.. An addition of +100 ug/L NH4-N or NO3-N could lead to a substantial biological response.

3.2. Northern Rockies: Comments re: section 3.1 apply here as well.

3.3 Canadian Rockies: The very tight, extremely low TP values among all but one of the "reference" sites suggest that the selection of 25 ug/L TP for a criterion is too. high. It is far beyond the 75th percentile of reference as well as above the 20 ug/L TP number identified by other stressor response studies from the region. Again, I struggle with the use of explanatory models for predicting mean chla/m2. In most situations if TP is elevated it will be elevated by phosphate; adding 15+ ug/L TP above the highest reference sites has a high risk of impairing streams.

3.4. Idaho Batholith. Similar thoughts—TP is < 20 across all samples in reference sites. The literature review and discussion of previous results provides reasonable support for 30 ug/L TP, but not defensibly so. Setting the criterion at 30 ug/L seems to leave the door open for a minimum of 50% increase in P loads to these streams. Given this is far beyond the 75th and 90% reference site quantiles, I think greater justification is needed, especially considering the previous ecoregion was set at 25 ug/L TP despite similar reference distributions for TP.

Section 3.5. More detail on the sources of TN and TP in this region would be helpful. Is alder or another nitrogen fixing plant abundant in the uplands here? We see high natural concentrations of N in high alder streams in glaciated portions of Alaska but very low N when alder is low (Shaftel et al. 2012). As for P, is the source volcanic? What explains the high P levels in reference sites?

Also note that the discussion justifying the choice of criteria is long and somewhat speculative, although I appreciate the level of detail.

Section 3.6. The nutrient dosing study seems like it was not used directly supporting numerical criteria in this region beyond demonstrating that dissolved N and P additions stimulate algae. The amount of dissolved nutrients added was not particularly great despite the large algal and DO response, so it concerns me to see such high recommended levels of TN and TP for the region. However, the distribution of values among reference sites does support the recommended levels, assuming the reference sites are indeed representative of streams with minimal anthropogenic nutrient inputs. The large range of values among reference sites suggests that level IV ecoregions may be needed to parse out natural variability or there are streams that probably shouldn't be considered reference sites.

Section 3.6.1. River breaks. I follow the rationale for concluding that no criteria are necessary for this region. The lack of dissolved nutrient information makes it difficult to know whether all of the nutrients, particularly P, are bound to sediment or whether there is abundant dissolved N and P. I agree that dissolved nutrients can be variable due to biological uptake but in these systems I would suggest considering dissolved N and P. Without any criteria, it still seems these streams could be vulnerable to animal waste discharges, future wastewater discharges, or other sources likely to contribute very high levels of dissolved nutrients as well as organic matter. The streams are reportedly flashy, but this suggests there are periods of extended low flows between periodic flood events that permit blooms of phytoplankton and/or shallow wate r attached algae/plants. As explained in this document, I don't think the state has presented sufficient justification for electing not to set criteria for these streams.

Section 4. Reach specific criteria.

4.1 Flint Creek. The rationale and methods for setting criteria for this reach seem defensible.

4.2 Bozeman Creek et al. Overall it is hard to find many suggestions on how they could improve their approach for setting criteria in these rivers. The approach used is consistent with how it was done among ecoregions.

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Review of Montana Department of Environmental Quality 2012 "Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers: Addendum 1"

General comments:

In preparation for the review below, I read Suplee et al. 2012 "Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers: Addendum 1", Suplee et al. 2011 "Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels", and reviewed Suplee et al. 2005 " Identification and Assessment of Montana Reference Streams: A Follow-up and Expansion of the 1992 Benchmark Biology Study." In addition, I reviewed considerable literature to refresh my memory about details and look for additional information.

Overall, the "Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers: Addendum 1" was scholarly, thorough, and as scientifically sound as any state nutrient criteria document that I've reviewed. I think the approach is sound, information is relatively sufficient, and this work provides a very good next step for stakeholders of Montana and the development of their nutrient criteria to protect their resources. I have great respect for the originality and scientific rigor of the research conducted by MDEQ and its application in water policy. With that said, my responsibility is to indicate strengths and weaknesses in the approach and results, as well as address a set of specific questions. My review includes recommendations for additional approaches and sources of information for deriving benchmarks for nutrient criteria and selecting benchmarks for criteria that I hope MDEQ will find useful in revisions of this document or in their future work. Much research remains to refine the information needed for states and tribes to establish nutrient criteria that will adequately protect designated uses of their waters without overprotection. MDEQ is a leader in that effort and that effort serves the state of Montana well.

Below I've addressed the specific review questions and commented on related issues.

MT – Wadeable Streams Draft Peer Review Questions

1. **MDEQ is considering two approaches for the derivation of numeric nutrient criteria in wadeable streams: (1) eco-regional reference condition, and (2) regional and non-regional stressor-response studies. Compare and contrast the ability of each approach to provide a sound scientific basis for numeric nutrient criteria derivation. Please provide documentation on any identified ranges protective of aquatic life based on**

similar studies. If possible, please provide alternate methodologies using available data and tools, and describe the corresponding advantages and disadvantages.

MDEQ used ecoregion specific stressor-response relationships and ecoregional reference condition to derive numeric nutrient criteria for wadeable streams. Stressor-response relationships were used to determine the nutrient concentration at which undesirable effects in stream condition occurred. Ecoregional reference condition was used to determine the range of nutrient conditions at groups of sites with minimally impacted condition (*sensu* MDEQ 2005), that meet designated uses, and that have similar natural determinants of ecological condition (based on ecoregion constraints). Combining information from stressor-response relationships and ecoregional reference condition, nutrient criteria were then proposed for nutrient concentrations (both TP and TN) that were: 1) related to negative effects in biological condition that were predicted by stressor-response relationships and 2) greater than or equal to the 75th percentile of nutrient concentrations observed at reference sites. If sufficient knowledge is available for characterizing responses of valued ecological attributes (e.g. biological condition) to nutrient enrichment and minimally impacted nutrient concentrations at reference sites, and these characterizations are done appropriately, then I would argue that this is the best framework for deriving nutrient criteria. So an appropriate question to ask is, "Has MDEQ appropriately characterized nutrient concentrations in minimally impacted condition and responses of valued ecological attributes (e.g. biological condition and other indicators of designated use support) to nutrient enrichment?" I'll get to that question later after I briefly defend the MDEQ approach.

I have argued that nutrient criteria (and other stressor criteria) for a site should be derived with at least three steps (e.g. Stevenson et al. 2004, 2008; Soranno et al. 2008), given sufficient information:

1. determine expected conditions[1] for a site (which can be reference or desired conditions) based on management goals (which can be designated uses);
2. determine effect of nutrient concentrations on valued ecological attributes related to management goals for the site (e.g. biological condition or other indicators of designated uses) and select benchmarks in nutrient concentrations for possible criteria;
3. select benchmarks in nutrient concentrations that are greater than or equal to minimally disturbed condition and at concentrations with acceptable risk to impairment of valued ecological attributes (i.e. often measures or indicators of designated uses).

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[1] Expected condition can be defined as minimally disturbed, least disturbed, best available, or desired condition (Stevenson et al. 2004). Here I use definitions of, least disturbed, best available from Stoddard et al. (2006) such that: minimally disturbed is "the condition of streams in the absence of significant human disturbance;" least disturbed is "found in conjunction with the best available physical, chemical, and biological habitat conditions given today's state of the landscape;" and best attainable condition is *"equivalent to the expected ecological condition of least-disturbed sites if the best possible management practices were in use for some period of time."* Desired condition is related to natural resources management and specifically addresses situations in which we management for attributes that may not be greatest in minimally disturbed conditions

In step 1 we should characterize the reference or desired (=expected) condition for the site that should include all physical, chemical and biological conditions that are related directly or indirectly to our management goals (e.g. designated uses) and that occur within the water, the riparian zone, the watershed, regionally, and even globally for contaminants transported through the atmosphere from distant sources. In some special cases, our goals may be to manage for desired condition (*sensu* Stevenson et al. 2004), such as more productive fisheries that are not characteristic of minimally disturbed conditions with high levels of biological condition (*sensu* Davies and Jackson 2006) in naturally low productivity ecosystems. Thus, tradeoffs between managing for productive fisheries and high levels of biological condition (biological integrity) are likely and should be addressed with tiered uses and different tiered uses for different waters within a region that meet the needs of regional stakeholders (Stevenson and Sabater 2010). Also, natural variation in climate, geology, hydrology, and water chemistry cause variation in minimally disturbed condition among ecoregions and among sites (e.g. Cao et al. 2007; Hawkins et al. 2010). So expected condition and nutrient critera, eventually, should be derived separately by ecoregion or by sites (e.g. Herlihy and Sifneos 2008; Soranno et al. 2008; Suplee et al. 2012 (the MDEQ document being reviewed)).

In step 2, we determine relationships between valued ecological attributes indicating designated and desired use support and nutrient concentrations. Nutrient concentrations are not a valued attribute because most people do not value them directly and only perceive risk from them if they cause problems to ecosystem services they do care about. There is little public support for managing nutrients independently of the effects that nutrients have on valued ecological attributes. We should not use reference condition nutrient concentrations alone to derive nutrient criteria because: 1) without stressor-response relationships we cannot be sure that nutrients affect valued attributes of the ecosystem and 2) we don't know the effects of incrementally increasing nutrient concentrations and at what nutrient concentrations risk of losing attributes become unacceptable. In evaluating stressor-response relationships, nutrient concentration benchmarks for potential criteria should be identified at the highest levels of nutrient concentrations at which an acceptable risk of losing valued attributes occurs. Thresholds in stressor-response relationships are highly valuable for delineating levels of nutrient concentrations at which risk levels change dramatically, thereby generating consensus among stakeholders for establishing criteria at specific nutrient concentration benchmarks.

In step 3, we determine what responses in valued ecological attributes change have acceptable risk benchmarks at nutrient concentrations greater than or equal to expected (usually reference) and then determine which benchmarks should be selected for nutrient criteria. In general, it's impractical (although not impossible) to manage a resource for nutrient concentrations lower than minimally or least disturbed condition, so nutrient criteria are usually at least as high as nutrient concentrations in reference conditions[2]; and criteria may be higher than reference conditions if valued attributes are not affected by nutrient concentrations less than or equal to reference conditions.

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[2] Nutrient concentrations characteristic of reference conditions and supporting conditions of reference conditions are not any concentration within the range of nutrient concentrations observed at reference conditions. This will be discussed later in the text.

Now to the question, "Has MDEQ appropriately characterized nutrient concentrations in minimally impacted condition and responses of valued ecological attributes (e.g. biological condition and other indicators of designated use support) to nutrient enrichment?" Here I will also address elements of the review question:

- Please provide documentation on any identified ranges protective of aquatic life based on similar studies.
- If possible, please provide alternate methodologies using available data and tools, and describe the corresponding advantages and disadvantages.

I'll address the question and review question elements by criteria development step, and change the order of steps to correspond to the MDEQ methodology (characterizing stressor-response relationships and reference condition, and then deriving criteria).

Characterizing stressor-response relationships. MDEQ relies heavily on the relationships between nutrient concentrations and chlorophyll a, chlorophyll a and DO stress, and chlorophyll and aesthetics to related nutrient concentrations to support of designed uses. The nutrient-chlorophyll relationship is therefore the primary determinant of DO stress (e.g. Stevenson et al. 2012), which is an important stress on aquatic biota. The nutrient-chlorophyll relationship is also a primary determinant of aesthetics issues. Suplee et al. (2009) show reduced desirability of rivers for recreations use with chlorophyll a exceeding 125-150 mg chl a m^{-2}. The stressor-response relationships that they use are peer-reviewed and scientifically sound, or they have been developed by their own research in regions in which they have particular concern that that existing nutrient-response relationships would not apply. They consider different stressor-response relationships for different ecoregions, which is appropriate, because we would not expect high gradient streams, as in the mountains or foothills, to respond the same to nutrient pollution as in the low gradient streams of the prairies (see Stevenson et al. 2006 for example or ecoregion specific relationships). As an aside, I tried to compare the nutrient concentrations required to produce 125 mg chl a m^{-2}, but I could not determine which equation in Dodds et al. 2006 was equation 19. Comparing predicted nutrient concentrations at chlorophyll management targets using models in Mebane et al. (2009), Dodds et al. (1997 and 2006), and Stevenson et al. (2006) would be informative. Providing these models in the report would have been valuable for establishing the basis for the range in nutrient concentrations that were reported as required to maintain 125 mg chl a m^{-2}. Also, although results of experiments are based on soluble nutrients, Bothwell's experimental work with P and the N and P experimental work of Rier and Stevenson (2006) could be used to support determination of nutrient benchmarks for regulating chlorophyll a accrual.

While MDEQ's approach is scientifically sound, there are other relationships between nutrients and elements of stream ecosystems that may be important for determining whether nutrient pollution threatens designated uses of Montana waters. MDEQ definition of minimally impacted condition[3] indicates that more than chlorophyll and DO stress on invertebrates

[3] MDEQ (2005, p 2) defines minimally impacted condition as "Tier 2 — Minimally Impacted Condition" as "The characteristics of a waterbody in which the activities of man have made small changes that do not affect the completeness of the biotic community structure and function and the associated physical, chemical, and habitat conditions, and all numeric water quality standards are met and all beneficial uses are fully supported unless

should be included in stressor-response relationships. Since I did not find reference to the attributes specifically used to characterize designated uses of MT waters, I will mention some additional information that may be valuable to consider and which might not have been considered by MDEQ.

Relationships between nutrient concentrations, chlorophyll a, DO stress, and aesthetics likely cover most designated uses related to recreation, but may not protect biological condition of invertebrates, algae, and ecosystem function. Stevenson et al. (2008) observed very sensitive response of benthic diatom assemblages in the high gradient streams of the mid-Atlantic highlands with loss of sensitive species and deviations in species composition from reference condition at nutrient concentrations well below the 30 µg TP/L benchmark used for several MT ecoregions. With the abundance of periphyton data in the Western EMAP, the STAR reference site projects (Hawkins et al.), and now the National Rivers and Streams Assessment, generating informative stressor-response relationships for biological condition of periphyton and nutrients should be very practical.

In addition, Miltner and Rankin (1998), Yuan (2004), Smith et al. (2007), and Wang et al. (2007) describe invertebrate responses to nutrient concentrations that could be used to justify benchmarks for protecting biological condition of invertebrate communities. The mechanisms causing changes in species composition at relatively low nutrient concentrations are not well understood. DO and pH stress with nutrient enrichment are two likely mechanisms (Stevenson et al. 2012). In addition, release of streams and rivers from nutrient limitation enables invasion of habitats by taxa requiring higher productivity levels to survive and may shift competitive hierarchies in ways that cause loss of sensitive taxa adapted to naturally stressful low nutrient concentrations (Stevenson et al. 2008). Finally, release of aquatic ecosystems from nutrient limitation may enable invasion and reproduction of aquatic bacteria and fungi that could stress all other biota.

I applaude MDEQ's use of both TN and TP criteria because either can be limiting algal growth in streams with different geological conditions and resulting water chemistry, and at different times of years in some watersheds. I think this is largely done correctly given the amount of information available, where in high P reference regions MDEQ proposes low N criteria to constrain algal accrual. I think selected concentrations will be protective of high biomass in most cases where low N is used to constrain algal accrual. However, I do want to caution that we need to learn more to accurately quantify algal nutrient relationships with both TN and TP in the model, as was used by Dodds et al. (2002, 2006) and MDEQ. Such models violate Leibig's Law of the Minimum. MDEQ does address this in their report, but in reality, those justifications may not be sufficient. There is evidence in recent research that Leibig's Law of the Minimum does not hold, which makes me think algal biomass models with TN and TP linked are appropriate. Even though the science is a bit soft here, I would recommend using

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measured impacts are clearly linked to a natural source. Minimally impacted conditions can be used to describe attainable biological, chemical, physical, and riparian habitat conditions for waterbodies with similar watershed characteristics within similar geographic regions and represent the water body's best potential condition."

the linked models and unlinked models as a multimodel approach for getting a range of conditions that would probably constrain algal biomass below the 125-165 µg chl a m^{-2} targets.

Characterizing reference condition. MDEQ's characterization of nutrient concentrations at reference sites suffered from a low sample size in three ways: 1) for all but a couple ecoregions, there were very small numbers of sites; 2) ror a couple ecoregions, there were fewer than 30 observations of nutrient concentrations at reference sites; and 3) repeated measures of nutrient concentrations at the same site are not independent. In the truest sense of pseudoreplication, the characterization of central tendency and variation in nutrient concentrations at reference sites suffers from some level of dependence in the samples.

The pseudoreplication issue should be addressed in a straightforward manner and put into a broader context so that it does not become overly important as a distraction from the relatively sound science that does underpin MDEQ's efforts. Although Suplee et al. (2011) address the pseudoreplication issue in another report, the key point is that it should be addressed. The broader context should include the following points. First, precise characterizations of percentiles are not that important because reference condition was used as a point of "reference" for nutrient benchmarks in stressor-response relationships where undesireable conditions developed. Second, the relative independence of repeated measures in reference condition is probably pretty low, given other sources of variability in estimates of nutrient concentrations in a stream: spatial and temporal variability in nutrient concentrations of streams and analytical error. I have argued this myself (Stevenson et al. 2006). However, repeated measures statistics can be calculated relatively easily to determine the relative dependence of measurements from the same site given overall variability and to correct estimates of variance among sites for dependency in repeated measures to more accurately characterize the central tendency and variation in nutrient concentrations at reference sites. The evenness approach (calculating evenness of measures among sites) that MDEQ uses is interesting, but it does not address pseudoreplication and dependent measurement issue directly.

Modeling expected nutrient concentrations at sites with land use-nutrient relationships is another method for characterizing the central tendency and variation in nutrient concentrations in minimally disturbed conditions. Modeling reference condition is valuable when the number of reference sites is low or quality of references sites varies between regions, which may have been the case in MT. Examples of different approaches for this kind of modeling can be found in Dodds and Oakes (2004), Herlihy and Sifneos (2008), Stevenson et al. (2008), and Soranno et al. (2008).

Typically, if an endpoint of management is used in criteria development, or as pseudocriteria, as chlorophyll a, then reference condition of that parameter is also described. Reference conditions were reported consistently for TP and TN concentrations. I'd recommend that chlorophyll a, diatom decreasers metric, and Hilsenhoff's biotic index (HBI) be described for reference conditions.

Selecting nutrient benchmarks for criteria. In general, if valued ecological attributes (direct indicators of designated use support) respond sensitively within the range of nutrient conditions at reference conditions, it is difficult to justify higher nutrient benchmarks than the

75^{th} percentile of reference condition, assuming reference condition supports designated uses as described by MDEQ. I remember three distinct exceptions to this rule in MDEQ's proposed criteria. One is several Rockies ecoregions, in which proposed nutrient criteria were substantially above background concentrations, the other was in an ecoregion in which P was high and the TN criterion was well above the 75^{th} percentile of reference condition, and the other was in the River Breaks region in which no criteria were proposed because no known ecological responses to nutrients were known for concentrations that high. I'll address the River Breaks situation below with the specific question asked for the review.

I'm concerned about selecting nutrient criteria above background concentrations in the Rockies ecoregions because proposed criteria would not protect sensitive, low nutrient diatom taxa, ecosystem functions of low productivity systems, and likely corresponding biodiversity of other groups whose response to low nutrient concentrations are poorly understood (bacteria, meiofauna, even benthic macroinvertebrates species). In Stevenson et al. (2008) we observed substantial changes in species composition of diatom assemblages at low nutrient concentrations and substantial loss of sensitive, low nutrient taxa (from counts) across the range of nutrient conditions. I've seen the loss of sensitive, low nutrient taxa (from counts) with low levels of nutrient enrichment in the extensive ecological assessment work that I've done around the country. Yes, this is just loss of taxa from counts, and we're not quite sure what that means (although my students and I are trying to understand that more), but we may actually be losing more taxa from the habitat (not just counts), as well as losing fewer. At this point, it just depends upon the weight of assumptions in the model. But if this is true for diatoms, then what about other groups. Also allowing higher N concentrations as well as P concentrations could impair biological integrity of these minimally disturbed, near-natural systems. For example, releasing N limited systems from severe N limitation could cause loss of diatoms with N-fixing cyanobacterial endosymbionts (e.g. *Epithemia*) or allow invasion of potentially nuisance taxa. Also, the relaxed P and N criteria are close to thresholds for releasing systems from severe nutrient constraint, so nuisance growths of algae could occur more frequently than if nutrient criteria were constrained to reference condition. Quantifying acceptable risk of nuisance growths should guide considerations.

Setting stressor criteria at a stressor level predicted to cause a target responses (i.e. nutrient criteria at nutrient concentrations at which a model predicts a target 125 mg chl a m^{-2}) means that when the stressor is at that level, the response with be greater than the target 50% of the time and less than the target by 50% of the time and by a magnitude that is related to the mean square error of the predicted values. Should quantile regression or conditional probabilities (Paul and McDonald 2005) be used to determine the stressor level that will manage the response with an acceptable frequency and intensity of exceedance?

Thresholds, relatively abrupt changes in rates of response along stressor gradients, are valuable for deriving environmental criteria (Muradian 2001). They identify benchmarks for possible nutrient criteria and help determine which benchmarks should be used as criteria. Some threshold responses are more valuable than others (Stevenson et al. 2008). Information for different threshold responses should be interpreted differently. For example, a response showing assimilative capacity and then a threshold response as stressors increase (Stevenson et al. 2008, Figures 2A & B, , sometimes called a type III response (http://en.wikipedia.org/wiki/Functional_response)) is particularly valuable for deriving criteria because stressor levels just below the threshold are clearly protective of reference condition and provide a margin of safety. Thresholds in responses showing high rates of change at low stressor levels and little change at high stressor levels, sometimes called a saturation curve or type II response (Stevenson et al. 2008, Figure 2C, (http://en.wikipedia.org/wiki/Functional_response), is more difficult to apply in criteria development. Nutrient uptake and algal growth have this type response along nutrient gradients with highly sensitive responses to nutrients at low concentrations and little response to nutrients at high concentrations.

The question then becomes, "How low should you set criteria to constrain growth in a

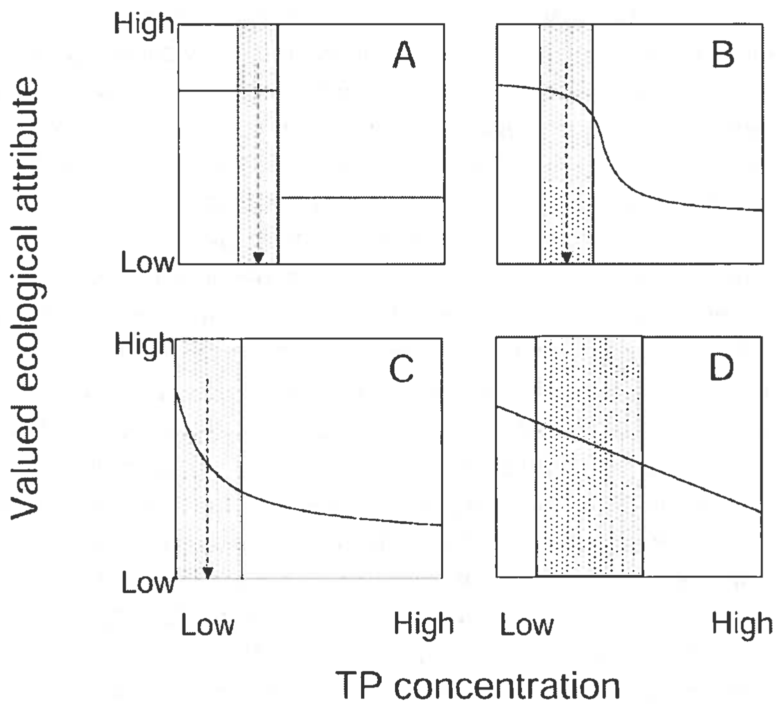

FIG. 2. Approaches to development of stressor criteria when potential responses of valued ecological attributes to stressors (e.g., total P [TP]) are nonlinear with assimilative capacity for increases at low levels the stressor (A, B), nonlinear with strong sensitivity to changes at low levels of the stressor (C), and linear (D). A stressor criterion is established at a level of a stressor that protects the valued ecological attribute. Arrows indicate TP criteria justified on the basis of the form of the stressor–response relationship. Shaded areas indicate the range of TP criteria that could be acceptable. Acceptable ranges vary as a function of the linearity of the stressor–response relationship and the type of nonlinear relationship.

part of the curve that is relatively linear?" Setting criteria just below the threshold (or breakpoint as described in Dodds et al. 2002, 2006 and as applied in this MDEQ document) provides little protection from adverse effects. Algal growth and accrual are largely at their greatest levels at nutrient concentrations just below those breakpoints. So in Transitional Level IV Ecoregions of the Northwestern Glaciated Plains, a justification for TN criteria of 560 µg/L was 560 was lower than the maximum saturation threshold (with saturation thresholds of 367 and 602), this criterion would not constrain algal accrual if these models are correct (which is the assumption of using them). The real explanation for choosing that level would seem to be that 560 µg TN/L is close to the 75th percentile of reference condition and you can't expect to do much better than that, even though biomass-nutrient models indicate biomass accrual could be near maximum levels at that TN concentration.

Implementation. I like the concept of Tier I and II assessments for determining whether sites meet nutrient criteria. This does relate to an issue about risk of use support and the way we use statistics to define reference condition and determine whether a site meets its water quality criteria. I'll discuss this under a later question about exceedance frequencies. In particular, I like the use of biological condition assessment in the Tier II assessments with the diatom and macroinvertebrate metrics. These metrics should provide a temporally integrated signal that should complement the temporally variable assessments of nutrient concentrations. I would, however, recommend that MDEQ include metrics that evaluate decrease in sensitive native taxa as well as the increase nutrient pollution tolerant taxa (e.g. diatom increases and HBI taxa), because these taxa are key elements of biological condition for which we manage waters (Davies and Jackson 2006).

MDEQ chose to implement criteria during the growing season only, which assumes that the mechanism by which nutrients affect designated uses is by stimulating algal growth and that recreational use exposure is during the growing season. This is likely true, that algae do not bloom to nuisance levels or threaten low DO or high pH during non-growing seasons. But there are other potential ways that nutrient pollution can affect aquatic life use, which are poorly understood and poorly documented (i.e. shifts in competitive hierarchies and disease), and some nuisance growths of diatoms that alter habitat structure can occur during cooler seasons of the year.

I'm surprised there is little difference in when the growing season occurs. Why not use water temperature (for algal endpoints) and degree days (for invertebrate endpoints)? Do these time periods allow for interannual variation?

Tiered aquatic life uses should be considered (Davies and Jackson 2006). The problem with potential management challenges in the Rockies (as well as elsewhere), where reference nutrient conditions seem really low and well below most targets for designated use, plus the desire for P enrichment to support fisheries and limit *Didymosphaenia* blooms, is that we could lose attributes of natural systems that now exist. Tiered aquatic life use policies could allow protection of some systems within those ecoregions for near natural structure and function (which now exist) and allow other systems to be managed for fisheries and *Didymo* control.

I sense several issues touch on the policy doctrine of "independent applicability" of stressor and response criteria. Conceptually, one reason to set criteria within reference conditions (with a margin of safety) is because there may be negative responses that we don't know about if stressors are higher. In a perfect world, we know all the possible responses to stressors, so we could relax stressor criteria to levels that protect desired responses with acceptable risk. BUT, do we know enough about nutrient effects on designated uses to make relax criteria to ranges outside the reference condition (i.e. greater than the 75th percentile of reference condition as argued below)? MDEQ does use elements of independent applicability in their assessments. For example, level I assessments only involve comparisons of nutrients and not biological endpoints to nutrient criteria. In addition, if either N or P fail, then the system is not in compliance (MDEQ 2011, pp. 3 and 4), both do not need to fail to be in noncompliance. I did not find and review the assessment methodologies for level II decisions with sufficient detail to evaluate issues related to independent applicability.

MDEQ is as knowledgeable about potential nutrient impacts as any other state or tribal agency. They have chosen a level of risk with which they are comfortable for protecting their waters. Rivers are relatively resistant ecosystems. If errors are made, given the MDEQ "good-faith" effort, designated uses of the rivers should be able to be restored, unless there is regional extirpation of taxa which is unlikely in the short term.

2. **In Section 3.6.1., Montana suggests that no nutrient criteria are needed for streams in the 1 Level IV Ecoregion within the Northwestern Great Plains: River Breaks (43c). The MDEQ rationale for this decision is: "This level IV ecoregion has highly turbid, flashy streams with naturally elevated TP and TN levels. Concentrations observed in the region's reference sites indicate that nutrient concentrations here are already naturally elevated above the harm-to-use thresholds identified for the plains region as a whole. As such, no nutrient criteria are recommended for streams within this level IV ecoregion." Please comment on whether the state has provided a sufficient scientific basis that 1) these levels are <u>naturally</u> elevated, 2) additional increase in nutrients would not cause harm to aquatic life, and 3) that, therefore, criteria are not needed. Is the reviewer aware of any additional information that could be provided to either support the State's assessment of natural background or that could be used to derive site specific criteria?**

I don't like the idea that there are no nutrient criteria set for waters, even given the rationale that natural concentrations are naturally high and no instream or downstream effects are expected to occur. It makes me nervous that we know enough about nutrient-stream relationships to make that call. Will antidegradation policy prevent this system from getting worse? Why not have criteria be existing condition, i.e. the reference condition, as the criterion? Independent applicability would call for using reference condition of a contaminant in this case. Addressing questions 1-3. 1) I am not convinced that MDEQ has provided a sufficient scientific basis that nutrient concentrations are naturally elevated because: the quality of reference sites relative to land use in this region is not described, so how minimally disturbed is the reference condition; how is minimally disturbed and meeting designated used defined in this ecoregion if the systems are so naturally stressed; the number of reference streams sampled is low (n=8), even though the number of samples in relatively high (n=29), but note the 3 outlying samples with TP > 3.6^{10} (i.e. >3981 µg TP/L) that are likely from the same stream and indicating a site-specific dependence; and modeling reference condition with nutrient and land use data from all sites in the region may help better evaluate minimally disturbed conditions. 2) It does not seem likely that there are no instream or downstream effects of elevated nutrients because: phytoplankton blooms can occur during storm-free periods when waters slow and clear ; and downstream effects seem likely because patches of this ecoregion are so small and waters having to flow somewhere. 3) Criteria should be established to prevent dumping in this region, prevent degradation, and prevent surprises.

4. **MDEQ's criteria approach includes a Chl-a value of 125 mg/m^2 to be used as part of the related assessment information. Please comment on the selection of chlorophyll as the**

primary response variable, the derivation of the chlorophyll threshold, and its application as a statewide assessment indicator.

Using chlorophyll a or any indicator of valued ecological attributes, such as the diatom decreasers and Hilsenhoff biotic index, is an important check an assessments based on stressors because they directly address whether uses are being met. Chlorophyll a is a particularly important variable to use in determination of nutrient criteria and assessments of site compliance because it is probably the best indicator of algal biomass that we have and most effects of nutrients on designated uses of rivers and streams are caused by stimulation of either benthic or planktonic algal growth. MDEQ's derivation of 125 mg chl a m^{-2} as a management target to protect recreational use of rivers and aquatic life from DO stress is a model for what should be done by other states and tribes. In general, the chlorophyll standard was appropriately varied from region to region when reference condition nutrients were in the 30 µg TP/L and 300 µg TN/L range, but I do have concerns about using chlorophyll as an endpoint in the Rockies ecoregions where reference nutrient concentrations are low and nuisance levels of chlorophyll causing impairment of aesthetics and DO are not the only likely cause of changes in biological condition. Protecting biological condition at near natural levels may, however, be above the level of protection that stakeholders support in Montana. Although, tiered uses or an outstanding resource waters protection could be used to protect at least some low nutrient systems from increased productivity and resulting changes in biological condition. Other than these overall comments, details supporting the comment for question 4 are covered under question 1.

5. **Section 4.0 outlines a process for determining reach-specific nutrient criteria. Please comment on MDEQ's proposed approach for deriving reach-specific values.**

There are special situations when establishing nutrient criteria based on regional reference condition may be too high or too low. In the case of the Georgetown Lake Dam, the state statues call for a recalibration because they won't alter the location of the intact. The flow weighted approach in Bozeman Creek, Hyalite Creek, and East Gallatin also seems sound. I do question the relaxation of TN criteria above the very low 100 µg/L reference condition to around 250 µg TN/L, again for protecting high quality waters in Bozeman Creek, Hyalite Creek, and East Gallatin. This is the same issue as discussed for very low TP conditions in many of the Rockies ecoregions, but protecting high levels of biological conditions is a different issue than whether reach-specific criteria were determined appropriately based on management endpoints related to algal biomass, DO stress, and aesthetics.

3. **MDEQ is proposing to allow TN and TP criteria to be exceeded 20% of the time and be considered supporting aquatic life uses. This frequency was derived based on analysis of the Clark Fork River chl-a data. Please comment on the proposed exceedance frequency and whether allowing the stated magnitudes to be exceeded 20% of the time would not result in adverse effects on aquatic life. This information is discussed in the State's Assessment Methodology.**

6. **Montana is proposing to interpret the numeric criteria using the Students t-test and binomial test to determine whether a stream segment is impaired. Please comment on the State's rationale for this approach.**

I want to address questions 3 and 6 together. I think they are related. They are kind-of statistical issues.

First, I'd expect that 20% or more of observed TP and TN conditions would exceed criteria levels at sites maintaining an average target condition of, for example 150 mg chl a m^{-2}. The way that the nutrient criteria have been developed is based on the relationships between nutrient concentrations and chlorophyll observed at a site (Dodds et al. 1997, 2002, 2006). I'm going to use one of my Figures from Stevenson et al. (2006, redrawn and rescaled) to illustrate this because the non-linear relationships illustrated in the Dodds et al. (1997) paper are too complex to illustrate these principles and the plotted data looks a bit off in Dodds et al. (2002), which may be related to the erratum of Dodds et al. (2006).

In the statistical models that we use, there is both variation in the measurements of the independent and dependent variables (see Figure 1, blue and red lines respectively could be standard error bars of predicted and measured values). For example, when the average nutrient concentration is 30 µg TP/L in Michigan streams, chl a is approximately 20 mg/m^2. Michigan streams are grazer dominated and little periphyton accumulates with increases in nutrients, unless *Cladophora* can escape grazer control. Note values in Michigan are much lower than Kentucky by almost an order of magnitude. MDEQ based nutrient criteria on algal-nutrient model predictions to maintain biomass at a specific level or lower – usually 125 mg chl a m^{-2} target. So we should expect that nutrient concentrations will sometimes exceed the 125 mg chl a m^{-2} concentrations of the model, because there is variation around the predicted value. Actually, I'd expect the exceedance frequency to approach 50% as observed average nutrient concentrations at a site approach the criterion.

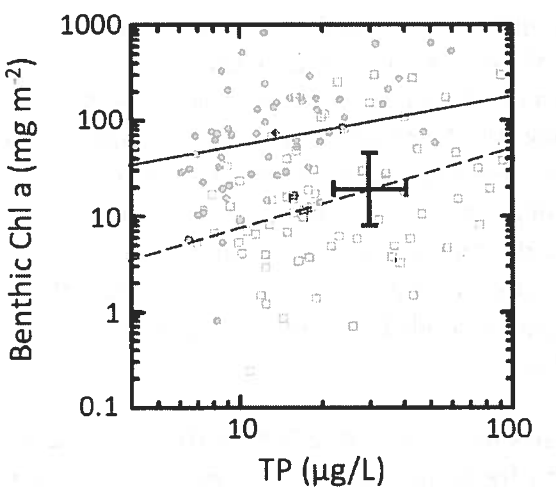

Figure 1. Figure 5 from Stevenson et al. (2006) redrawn and rescale. Open squares are values for streams in Michigan and shaded circles represent streams in Kentucky. Lines, dashed and solid respectively, are the linear relationships among the points.

This increase in exceedance frequency with nutrient concentration and algal biomass is illustrated in Suplee et al. (2011) Figure A4-1. Even for sites with the three lowest nutrient exceedance frequencies (and three lowest nutrient concentrations), maximum summer chl a is frequently greater than the 150 mg/m^2 expectation for the Clark Fork. The exceedance frequency actually provides a measure of risk of losing an attribute, in this case it's an aesthetically pleasing recreational venue and the potential for a DO event, which depending upon severity and extent, could have long-term repercussions for some biota. MDEQ have provided a margin of safety with lower biomass targets than impair aesthetics or cause DO stress and often lower nutrient creiteria than predicted to generate the target chlorophyll concentrations. This margin of safety may be the reason for average exceedance frequency at sites meeting uses being less than 50% even with nutrients concentrations near criteria.

Finally, I address the t-test and binomial test issues and issues with frequency distributions based on observations versus means. These issues are related to the risk of use support being affected by the way we use statistics to define reference condition and determine whether a site complies with water quality criteria. I want to start this discussion by reviewing a rationale for using frequency distributions of observations from reference conditions and a mean from a test sites to assess compliance at the test site. Then I'll transfer those concepts to evaluate how MDEQ's approach affects risk of supporting designated uses based on using observations, means, regression, binomial tests, and t-tests.

Consider the following scenario. Reference sites are selected because they are minimally disturbed based on land use and they support a specific level of aquatic life (and/or other uses) with an acceptable risk (let's say a management endpoint like chlorophyll a exceeds criteria 10% of the time). A frequency distribution is used to characterize central tendency and variation in a stressor (e.g. nutrients) that affect designated use within the range of conditions at reference sites. The frequency distribution is based on single samples from reference sites within an ecoregion (single independent observations in a statistical sense, Y_i, Figure 2A). We then use the 75th percentile of the frequency distribution of observed stressor conditions at references sites (Y_i^{75}) as a criterion, because we recognize that conditions vary around the average or median condition for reference sites due to spatial and temporal variation related to weather, flow, time of day, etc. and measurement error. Conceptually we use the 75th percentile of the frequency distribution of observed stressor conditions at references sites (Y_i^{75}) as a criterion because we feel that it's protective. Why? Well, one reason may be the statistical rule related to hypothesis testing called the 75% error bound. This is a rule of thumb that you can use to compare two means such that: if the mean of one sample (i.e. group) of observations (test sites) is outside the 75th confidence interval of the other sample of observations (reference sites), you can assume that there is little

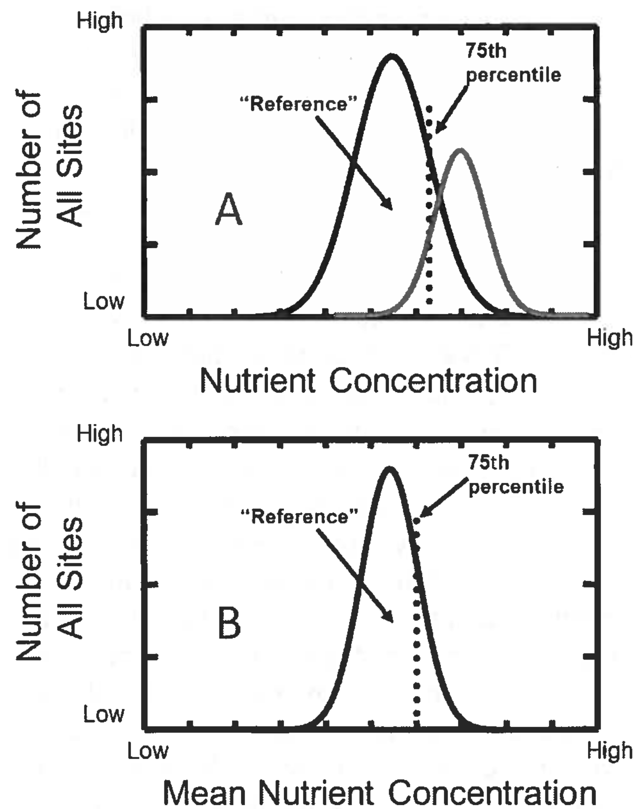

Figure 2. Distribution of nutrient concentrations when concentrations are represented by single samples site (A) and when concentrations are represented by means of samples at a site (B). The black distributions are reference site distributions. The blue distribution is the distribution of observations from a test site that is theoretically greater than the 75th percentile of the reference distribution according to a t-test or a binomial test.

probability that means of the first and second samples (groups of test and reference sites) are equal. Basically, 0.25 (1-0.75) is the attained significance for the difference between two means if the sample size is just 2 (if I remember correctly). So the idea is that conditions in the test set of sites would be different than the reference sites if the mean of test sites was greater than the 75th percentile of reference sites (based on single observations per site, or multiple observations from a smaller set of site that we could assume were independent). If agencies based development of criteria on mean measurements from a site, then the variance of the mean (Figure 2B) is much smaller than the variance of observations (Figure 2A). Comparing mean conditions at a site to the 75th percentile of a frequency distribution of mean observations at a site would be overprotective. Testing that mean of the test sites is significantly greater than the 75th percentile of reference sites (blue distribution in Figure 2A) versus just significantly greater than the mean of the reference site, would be underprotective.

MDEQ has proposed nutrient criteria that are the 75th percentile of reference condition or a higher concentration that is predicted to produce an effect that is undesireable. If my understanding of this process is correct, then using a t-test to determine whether mean conditions are greater than nutrient criteria would be underprotective, i.e. exceedance frequencies would be very high at sites before a site was found to be noncompliant. The mean concentration at the test site would have to be greater than the 75th percentile of the reference condition or the predicted level of nutrients causing a problem by an amount related to the variance in observed nutrient concentrations at the test site, the number of samples from the test site (n), and a t-statistic (which has a value of 2 when n is high). Issues associated with a binomial test are similar – some proportion of observed test site nutrient concentrations greater than 50% have to be greater than the criterion that is set at the maximum concentration that protects designated uses.

To counter these statistical issues with use of a t-stat causing underprotection of test sites, MDEQ does seem to have employed some margin of safety in setting the criteria at 125 mg chl a m^{-2}, which is below the 150 mg maximum okay level. In addition, MDEQ has adjusted acceptable levels of type I and II errors to reduce the problem of not detecting problems when they exist, and MDEQ is using chl a, a diatom indicator, and an HBI response criteria when nutrient concentrations are not obviously high. These additional rules can generate either greater under- or over-protection of waters. If only one the five criteria (N, P, chl, diatoms, HBI) has to fail, then that makes the assessment of compliance more protective than if the two nutrient criteria or all nutrient and biocriteria have to fail. I was not able to find details about compliance rules, which are apparently embedded in the spreadsheet that is referred to, so I can't evaluat Tier II assessments later. Tier II assessments, in which additional samples are collected and information is gathered also improves detection of non-compliance, reducing the error variation around means for the test site, which reduces possible difference between means at the test site and the nutrient criterion.

Overall, if I were a stakeholder concerned about protecting valued attributes within the streams, I'd be more concerned about potentially high risk of frequent loss of valued conditions (exceedance frequencies) when criteria are set at stressor-response model predictions that are too close to unacceptable levels of conditions, not the 20% exceedance problem or the t-test issue. The next frontier in deriving nutrient criteria may be bringing in a stronger risk assessment (e.g. Paul and McDonald 2005).

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Review of: “Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana’s Wadeable Streams and Rivers: Addendum 1”

Peer Review Questions

1. *Approach* - Montana’s approach combines stress-response studies and ecoregional reference distributions to derive numeric nutrient criteria for Montana’s wadeable streams (not really two approaches as identified in review question #1). The approach relies principally on stress-response studies both within the ecoregion and in nearby regions if the reference distributions of TN and TP of the 2 regions are similar. If no stress-response studies were deemed relevant, then MDEQ relied on the reference distribution (e.g., 75th %ile of reference for TP in Absaroka-Gallatin ecoregion 17i). As a further condition on the approach, MDEQ also keeps the N:P ratio in the criteria in a “Redfield range” so that the N:P ratio is unlikely to deviate far from the Redfield ratio, or far from the ecoregional reference if the reference deviates far from Redfield (e.g., Absaroka-Gallatin). The stress-response studies include both experimental nutrient enrichment studies and empirical modeling studies based on monitoring data. Overall, I find the whole approach compelling because it makes effective use of available and relevant information.

   I think the approach would be strengthened by increased use of Montana’s own monitoring data to develop ecoregion-specific empirical models of benthic chl-a response to nutrient enrichment. Such stress-response studies were indeed used if they were available as separate reports or publications, but there is no systematic application of Montana’s data to derive empirical models or confirm the proposed criteria. These models could be used to confirm, refute, or adjust the criteria developed, and would further strengthen the criteria.

   The second part of EPA’s review question, starting with “Please provide documentation on any identified ranges…” is out of line. It is not a reviewer’s task to develop a compendium of alternative methodologies, advantages, disadvantages, data, tools, etc.

2. *River Breaks* – The description of nutrient conditions in the river breaks is plausible, and the river breaks region seems similar (though maybe less extreme) than other badlands ecoregions in the Northwestern Great Plains (including badlands regions of the Dakotas). However, these regions are not familiar to many persons steeped in the Eastern Forest Biome stream paradigms. Accordingly, MDEQ should provide more documenation for the assertions made about the River Breaks. Also, would the same considerations apply to the Little Missouri badlands and the Missouri River Breaks? Evidence could include: published stream studies of badlands-type regions, provided they are similar to the River Breaks; N and P content of soils and geological formations in the River Breaks and similar regions (for example, I have found from EPA’s ecoregion descriptions that the Cretaceous Hell Creek Formation occurs in several of the badlands/breaks ecoregions). Land use/land cover and population density could help show that the breaks and badlands are no different in land use than other Northwestern Great Plains

regions, and perhaps even lower population density and less alteration of land use/land cover than other parts of the Northwestern Great Plains. Early historic descriptions of the regions and their streams are also highly useful, if available.

What is missing from the River Breaks criteria is protection of downstream waters, the large rivers and reservoirs. For example, parts of the region drain into Fort Peck Reservoir. Nutrients in the Breaks streams could contribute to eutrophication of the reservoir. If there are no criteria for the Breaks region, then we could envision the following scenario: In the absence of criteria, the area could become a magnet for large industrial feedlots because no nutrient removal would be required. What happens when hundreds of feedlots drain into the Breaks and on into Ft Peck reservoir? Although criteria may not be required to protect the aquatic life in the streams, they may be required to protect downstream waters.

3. *20% exceedance* – As far as I understand, the 20% exceedance rule means that no more than 20% of single measurements may exceed the nutrient criteria concentrations, or that the site may exceed a criterion up to 20% of the time. Since Montana's proposed criterion is based on single measurements, it is reasonable to expect that some short-term variation above the nutrient criterion concentration will not result in excess chlorophyll. The other alternative is to frame the criteria in terms of an annual average (say, geometric mean) as EPA did for the Florida nutrient criteria. A central tendency measure, geometric or otherwise, also allows for some short-term high concentrations as long as the central tendency is not exceeded. Montana's is basically the same, but based on a percentile of individual measurements.

   The 20% frequency was based on analysis of the Clark Fork River, which shows that for the Clark Fork, the 20% criterion would work well. The problem is that the Clark Fork is a single basin in a restricted set of subecoregions, so we don't have empirical evidence whether 20% would apply to the rest of the state as well. Using Montana's existing monitoring data, I think it may be possible to repeat some of the Clark Fork analysis on other streams throughout the state to confirm or refute the 20% estimate. The Clark Fork data presents another opportunity as well: testing the entire nutrient assessment approach to determine if the actual error rates match with the desired alpha and beta of 0.25 and 0.30. <u>Recommendation</u>: the 20% exceedance rule seems reasonable and has empirical evidence to support it, but would be strengthened by additional analysis from other regions of the state.

4. *125 mg/m^2 chl a* –Benthic chl-a is clearly the most consistent response indicator to nutrients in wadeable streams, as shown by many studies, cited in the MT documents and elsewhere. Benthic macroinvertebrates, while associated with both nutrients and chl-a, have so far proved unsuccessful as a reliable response indicator to nutrient enrichment, as demonstrated by EPA's attempt to develop nutrient criteria for Florida streams. Montana has a rich tradition in monitoring benthic chl-a as well as benthic diatoms, and is making effective use of that tradition for developing nutrient criteria.
   Derivation of the threshold – The threshold was derived from literature values, observations of

streams in the MT ecoregions, an acceptability survey, and a nutrient enrichment study. For example, Welch et al. (1989; cited in MT docs) considered "Nuisance biomass levels" to be in the range 100 – 150 mg/m^2 chl-a. Other values are similar (Biggs 2000: mesotrophy is in the range 60-200 mg/m2; Dodds et al. 2002 [CJFAS 59:865-874]: 125 mg/m^2 is "high end" of chl-*a*). Surveys are context-specific, in that people will identify unacceptable conditions as those that they are not accustomed to seeing. Unaccepatibility thresholds are subject to shifting baselines: if the persons surveyed are accustomed to seeing eutrophic conditions, only hypereutrophy would be identified as unacceptable. Finally, the dose-response study suggested that synoptic reach-average benthic algae in the range 87 – 127 mg/m2 chl a resulted in unacceptable DO at the end of the growing season. These results would suggest that 125 mg/m2 is at or uncomfortably close to a value that could cause fish community degradation due to DO, and for mountain and transitional streams, the chl-a threshold should be lower.

Statewide use - First, it is unclear whether MT plans to use 125 or 150 mg/m^2 as the chl-a standard. Some regions have 125, others 150. As with the nutrient criteria themselves, it may be more appropriate to have chl-a criteria better adjusted to the ecoregions. For example, the expectation for mountain and foothill-transitional ecoregions is that streams are oligotrophic and coldwater, supporting Montana's famous trout fisheries. Given that 125 mg/m2 is in the range of "nuisance", well in "mesotrophy" and has been demonstrated to cause DO problems in Montana, this value is probably too high for mountain and foothill streams. I have no problem with higher values for Plains ecoregions.

5. *Reach-specific criteria* – Two methods for reach-specific criteria are proposed: empirical determination based on pre-defined natural conditions (in this case, dam operations), and ecoregional flow-weighted criteria for streams receiving input from more than a single ecoregion. Both of these approaches appear to be sound.

6. *Tests* – Montana's overall rationale for determining impairment, using both an exact binomial and the t-test, is well thought-out. However, the presentation was a bit confusing; I found I had to jump around between various parts of the 2011 document and its appendixes to understand the approach. The consideration of both significance and power, and the attempt to balance them, is especially encouraging, and shows MDEQ is concerned with both protection of the resource and prevention of unnecessary management. I do have some concerns:

    a. Are the effect size (0.15) and the critical exceedance rate (0.20) really double-counting the same thing? Effect size is a scientific determination that nutrient concentrations within 15% of each other (or within 5% of the criterion) are not meaningfully different in terms of response, so it protects against a statistically significant difference (which may be significant simply due to very large sample size) being declared an impairment when there is actually little chance of impairment for such a small difference. The exceedance rate essentially does the same thing: up to 20% of individual measurements can exceed the criteria, but chl-a will not exceed its criterion value. When both of these are used in the exact binomial, is it testing whether more than 20% of observations exceed the critical nutrient concentration plus 15%? If so, that would be double-counting. The Clark Fork data could be used to test/illustrate this issue empirically. Effect size is typically used in comparisons of central tendency, to protect against scientifically negligible differences being elevated to statistically significant differences simply due to large sample size. It is used most often in equivalence or noninferiority tests. I don't think effect size, as a % of the mean, is appropriate for the exact test, which does not use a mean.

b. Should the effect size be used in the t-test, especially for large or very large sample sizes?

General comments:

In view of several of the questions above, and different ways of calculating status of a streams reach, I was frequently confused whether the document was referring to instantanaeous measures, annual (growing season) maximum, or some measure of central tendency (mean, median, geometric mean, etc.) measure at one time (synoptic) at several sites on a reach, or a "sampling event average". I came to realize that Montana's proposed criteria only make sense in the context of individual measures, i.e., measurements of TN and TP are not to exceed the criterion more than 20%. Similarly, the chl-a criterion of 125 mg/m2 only makes sense as a maximum not-to-be-exceeded. However, it was not clear in the document how exceedance would be calculated. Critically, eventual measures of exceedance should match to the extent possible the way meaningful concentrations were calculated in the considerations to derive the criteria. Recommendation: spell out, with examples, of what is meant by single observations and different central tendencies mentioned in the documents, and which are used for the final criteria and for assessment.

Overall, I found Montana's approach sound and well thought-out. The devil, as always, is in the details: selection of chl-a values, derivation of critical exceedance rates, selection of effect size. Some of the quibbling on these values may never be resolved (including mine), and Montana needs to use best judgment supported by its analysis and other scientific results.

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DEVELOPING NUTRIENT CRITERIA FOR STREAMS: AN EVALUATION OF THE FREQUENCY DISTRIBUTION METHOD[1]

Michael W. Suplee, Arun Varghese, and Joshua Cleland[2]

ABSTRACT: The U.S. Environmental Protection Agency recommends two statistical methods to States and Tribes for developing nutrient criteria. One establishes a criterion as the 75th percentile of a reference-population frequency distribution, the other uses the 25th percentile of a general-population distribution; the U.S. Environmental Protection Agency suggests either method results in similar criteria. To evaluate each method, the Montana Department of Environmental Quality (MT DEQ) assembled data from STORET and other sources to create a nutrient general population. MT DEQ's reference-stream project provided reference population data. Data were partitioned by ecoregions, and by seasons (winter, runoff, and growing) defined for the project. For each ecoregion and season, nutrient concentrations at the 75th percentile of the reference population were matched to their corresponding concentrations in the general population. Additionally, nutrient concentrations from five regional scientific studies were matched to their corresponding reference population concentrations; each study linked nutrients to impacts on water uses. Reference-to-general population matches were highly variable between ecoregions, as nutrients at the 75th percentile of reference corresponded to percentiles ranging from the 4th to the 97th of the general population. In contrast, case studies-to-reference matches were more consistent, matching on average to the 86th percentile of reference, with a coefficient of variation of 13%.

(KEY TERMS: algae; rivers/streams; environmental regulations; nutrients.)

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INTRODUCTION

The over enrichment of rivers and streams by nitrogen and phosphorus (eutrophication) is a serious water quality problem. Eutrophication can, for example, impact recreational and water supply uses (Freeman, 1986; Dodds *et al.*, 1997), result in diel oxygen swings that impact fisheries and aquatic life (Welch, 1992), and increase the levels of organochlorine compounds (PCBs) in localized trout populations (Berglund, 2003). Eutrophication has been recognized as a water quality problem for a long time, well illustrated by the fact that the U.S. Environmental Protection Agency (U.S. EPA) commenced a national eutrophication survey of streams (Omernik, 1977) shortly after the passage of the 1972 Clean Water Act. To address the national eutrophication problem, the U.S. EPA in 1998 announced that it expected all States and Tribes to adopt numeric nutrient standards by 2003. However, recognizing the complexity of developing and implementing such standards, the

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[2]Respectively, Environmental Science Specialist, Montana Department of Environmental Quality, 1520 E. 6th Avenue, Helena, Montana 59620; Senior Associate, ICF International, 33 Hayden Avenue, 3rd Floor, Lexington, Massachusetts 02421; Project Manager, ICF International, 97 Orchard Woods Drive, Saunderstown, Rhode Island 02874 (E-mail/Suplee: msuplee@mt.gov).

U.S. EPA subsequently provided a more flexible approach. This approach allows States and Tribes to submit to the U.S. EPA plans outlining the process and schedule of how they intend to adopt numeric nutrient standards (memorandum to States and Tribes from U.S. EPA, Office of Science and Technology; November 14, 2001). The Montana Department of Environmental Quality (MT DEQ) developed and submitted such a plan in 2002.

It has been widely recognized that numeric nutrient standards would not be the same everywhere, due to natural influences on nitrogen (N) and phosphorus (P) concentrations by landscape-level characteristics such as climate, geology, soils, vegetation, watershed area, etc. (Johnson *et al.*, 1997; U.S. EPA, 1998; Rohm *et al.*, 2002; Snelder and Biggs, 2002; Snelder *et al.*, 2004). Ecoregions integrate into a single mapping system a number of these nutrient-influencing geographic factors (Omernik, 1987). Ecoregions have been used to partition the United States into zones expected to manifest relatively uniform nutrient concentrations (U.S. EPA, 1998, 2000a; Rohm *et al.*, 2002). This partitioning process is a necessary first step towards establishing numeric nutrient standards. However, there remains the need to identify appropriate nutrient criteria for each ecoregional zone.

Two statistically based approaches have been recommended by the U.S. EPA to select a criterion for any particular nutrient (e.g., total N, total P), within any particular ecoregion (U.S. EPA, 2000*b*). The first approach identifies the criterion as the 75th percentile of the frequency distribution of nutrient data from reference stream sites within an ecoregion. Reference stream sites are relatively undisturbed examples (i.e., they have minimal human impacts and support all beneficial water uses) that can represent the natural biological, physical, and chemical integrity of a region (Hughes *et al.*, 1986; Barbour *et al.*, 1996; Kershner *et al.*, 2004). The second approach selects as the criterion the 5th to 25th percentile of the frequency distribution from the general-population of nutrient data (U.S. EPA, 2000b). In practice, however, the 25th percentile is more frequently discussed in the U.S. EPA's nutrient documents than the 5th percentile, and is the basis for the U.S. EPA's national nutrient criteria recommendations (see U.S. EPA, 2000a, 2001; and related Clean Water Act section 304(a) nutrient-criteria documents). The option to select as criterion either the 75th percentile of reference or the 25th percentile of the general population is presumptive, as it assumes that reference and general-population frequency distributions will have a particular relationship to one another (Figure 1), and so nutrient concentrations selected via either approach will be similar.

In accordance with its nutrient criteria plan, the MT DEQ has been examining in detail the two criteria-selection approaches outlined above. MT DEQ identified a number of stream reference sites in the early 1990s (Bahls *et al.*, 1992), and has had a project in place since 2000 to identify and sample reference stream sites around the state (Suplee *et al.*, 2005). The availability of reference stream nutrient data enabled us to examine the relative merits of the reference *vs.* the general-population approach to developing nutrient criteria. Our purpose in writing this paper was to describe our finding that nutrient concentrations at the 25th percentile of general-population frequency distributions may represent overly stringent — or insufficiently protective — criteria. This will be dependent upon the relationship between the nutrient distribution of the general population and that of the corresponding reference population. We also report that nutrient concentrations at the 86th percentile of reference-site frequency distributions appear to be reasonable for establishing criteria. This is because nutrient concentrations at the 86th percentile of reference generally matched nutrient concentrations that begin to cause impacts to beneficial water uses (e.g., recreation and aesthetics, aquatic life) that are published in regional scientific studies.

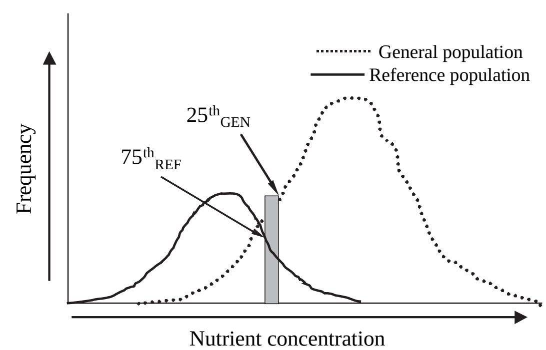

FIGURE 1. Presumptive Relationship Between a Reference and a General Population of Nutrient Data, Redrawn From U.S. EPA (2000b). The range of nutrient concentrations along the *x*-axis falling within the gray-shaded box are recommended by the U.S. EPA as appropriate nutrient criteria.

METHODS

Data Sources for the Development of a River and Stream Nutrient Database

The primary data source for the analyses was from the U.S. EPA's Storage and Retrieval (STORET) database. In March 2001, a request was placed with the then-functioning mainframe STORET database for all ambient surface water-quality data from Montana, excluding data from pipes, wells and springs. The delimited text file received was then transferred to a Microsoft Access® relational database. The STORET data (also referred to as Legacy STORET) contained data collected by 33 agencies or entities (organizations), and held nutrient data from the early 1960s to 1998. A query was run in the "Type" field (a field indicating the waterbody type) to remove lake data. The database was supplemented with all river and stream nutrient data from MT DEQ found in modernized STORET, which were collected from 2000 to 2004. Also added to the relational database were Montana river and stream data collected by the University of Montana, Utah State University, the U.S. EPA's Environmental Monitoring and Assessment Program (EMAP; Lazorchak *et al.*, 1998), and reference-stream nutrient data up through 2005 (reference streams will be discussed further on in Methods.) The database contained 5,300 sampling sites and over 140,000 total records. Readers should note that the data sources we used are comparable to those used by the U.S. EPA in developing its nutrient criteria recommendations. The U.S. EPA used data sources that included Legacy STORET, two United States Geological Survey projects (the National Stream Quality Accounting Network and the National Water-Quality Assessment [NAWQA] Program), and regional U.S. EPA data (see U.S. EPA, 2000a and related documents). Our database contained more records per level III ecoregion than the database the U.S. EPA used to develop its criteria recommendations, because the U.S. EPA restricted its dataset to information collected from 1990 to 1998 (U.S. EPA, 2000a).

Each analytical measurement in Legacy STORET was uniquely identified by a parameter code (e.g., 00665; total P). Other data that were incorporated into the relational database, including those from modernized STORET, did not use these codes. To assure consistency and to facilitate the grouping of data (discussed below), the appropriate parameter code was assigned to each observation lacking a code. The water quality data in the assembled database, which included latitude and longitude coordinates for each observation, were then spatially joined to Geographic Information System (GIS) layers containing information on level III and level IV ecoregions (Figure 2; Woods *et al.*, 2002). Observations were also labeled with the stream order (Strahler, 1964) of the stream reach from which they were collected. Strahler stream orders were derived from the U.S. EPA's reach file 3 (RF3) GIS layer (1:100,000 scale; U.S. EPA, 1994).

The final database was transferred to Stata® (version 7), which was more amenable to statistical analysis programming, and was referred to as the "all-observations" database to distinguish it from a "median" database. The median database was developed from the all-observations database and contained only the medians of the observed values for each nutrient, for each station, and for each season. (Seasonal data stratification will be detailed in a following Methods subsection.) The median database was developed because it was less likely than the all-observations database to be influenced by outliers, and was therefore more amenable to parametric statistical analyses.

Data Quality Control Methodology

Examination of the Legacy STORET dataset confirmed that it did not contain water quality data from pipes, wells or springs. Pipe, well, and spring sampling stations had been included in a Legacy STORET metadata (station-information) file. We linked this metadata file to the water quality database and verified that none of the pipe, well or spring sampling stations could be joined with any water quality data. To eliminate potentially erroneous or highly uncertain data from the analyses, data bearing certain comments codes were excluded (Table 1). Also, observations in the database bearing comment codes indicating the analytical result was below detection were replaced with values equal to 50% of the reported detection limits (DL/2; Table 1). For datasets skewed to the right, which were common in our nutrient database, the DL/2 method is reported to be sufficiently accurate for determining descriptive statistics like the mean and standard deviation (Hornung and Reed, 1990). Further, if less than 15% of the total dataset is below detection, the U.S. EPA (2006) indicated that the nondetect observations may be substituted, preferably with DL/2 values. Less than 15% of total observations in our database were below detection. Finally, nutrient observations with reported values of zero were excluded from use, since they probably represented data entry errors. Most analytical results in the database provided a result value, a detection limit and an indication when the measurement was below detection. True analytical result values of zero are very unlikely; for example, zeros are not reported for low-level organic pesticide analyses using HPLC methods even when no peak is detected (technical memorandum 94-12 from National Water Quality Laboratory to NAWQA study-unit chiefs, July 8, 1994, http://nwql.usgs.gov/Public/tech_memos/nwql.94-12.html).

Water quality data collected from streams and rivers are rarely normally distributed and are

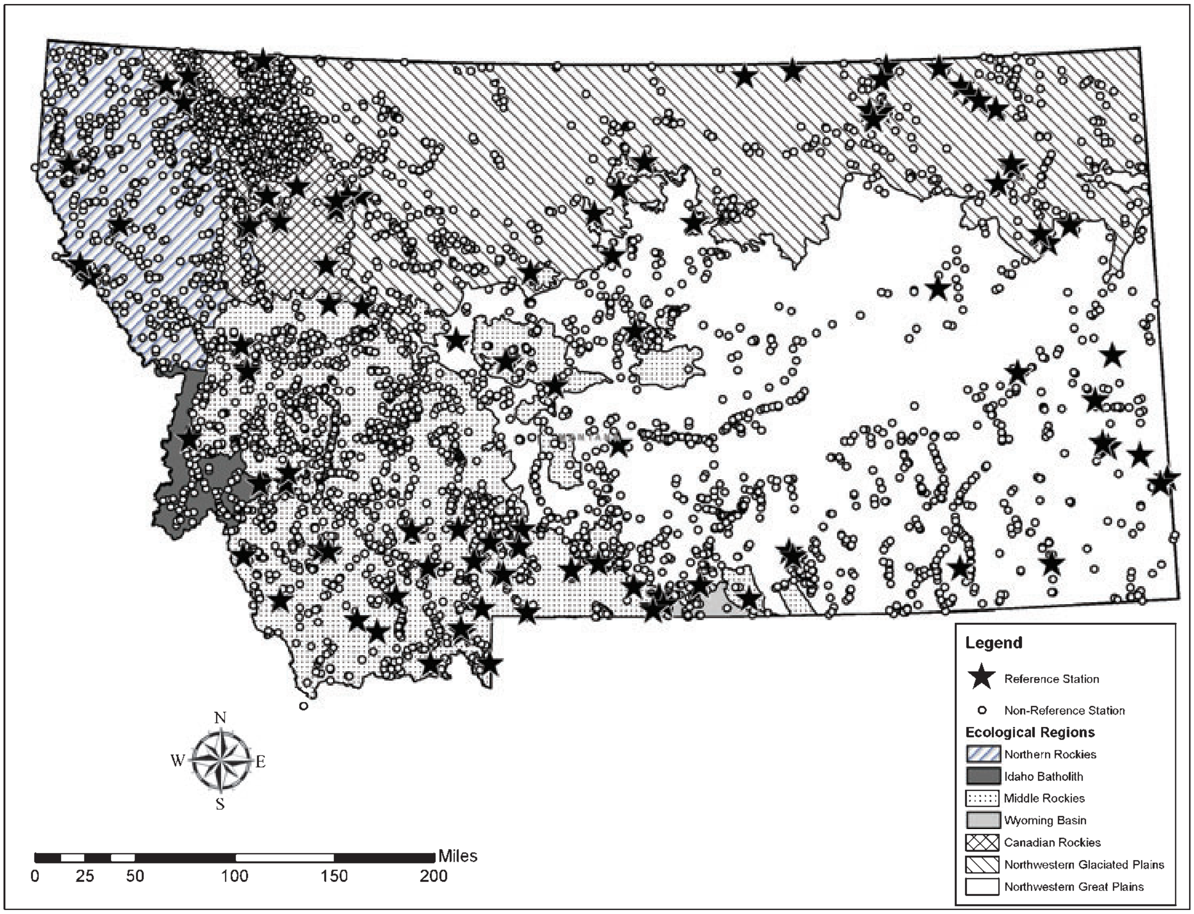

FIGURE 2. Map of Montana Showing the Location of General and Reference Population Sampling Stations. Shaded areas on the map are Omernik Level-III ecoregions.

frequently skewed to the right (i.e., lognormally distributed), and the presence of high outlier values in such datasets is common (Helsel and Hirsch, 1992). We did not have knowledge of the flow conditions or other important factors prevalent at the time the data were collected, and it would have been inappropriate to eliminate outlier data simply because they inconvenienced the statistical analyses (Helsel and Hirsch, 1992). Therefore, beyond the quality control measures described above, we did not further eliminate any data from the database.

Nutrient Data Grouping Methodology

We identified thirty different nutrient analytical measurements of N and P in the database, each bearing its own parameter code. Many appeared to be closely related, and rather than select a single parameter code to represent a given nutrient type (e.g., total P, 00665), we opted to aggregate the analytical measurements into groups. This approach allowed us to retain many nutrient analytical measurements that would have otherwise not been used. The objective was to group nutrient analytical measurements together that were fundamentally equivalent, while at the same time avoiding double-counts in cases where an agency may have reported two or more grouped analytical measurements from the same sample. The approach was undertaken in a series of steps. First, the different analytical measurements were identified by their parameter codes and other identifying information, checked against records (U.S. EPA, 1979; Alexander *et al.*, 1996; Clesceri *et al.*, 1998) to determine what they measured, and then organized into groups. The thirty nutrient measurements in the database were thus aggregated into seven groups (Table 2). Although we developed this grouping methodology independently, it is nearly identical to that used by Mueller *et al.* (1995) to aggregate nutrient data for an analysis of surface and groundwater. Next, a series of exploratory queries were made in the database for each group and for each agency, to ascertain if any analytical measurements within the group were derived from the same sample. In cases where this occurred, only

TABLE 1. Quality Control Actions Taken to Eliminate Potentially Erroneous or Highly Uncertain Data.

Data Remark Code	Remark Code Description	Database Source	QC Action Taken
*	No definition could be found	Various	Eliminated*
<	Less than detection	Various	Kept; used 50% of reported detection limit
E	Estimated value	NWIS	Eliminated
H	Field-kit determination	STORET	Eliminated
J	Estimated value	STORET	Eliminated
K	Value known to be less than reported value	STORET	Kept; used 50% of reported detection limit
L	Value know to be greater than reported value	STORET	Eliminated
M	Presence varified, but at a level too low to quantify	STORET	Kept; used 50% of reported detection limit
ND	Non-detect	Various	Kept; used 50% of reported detection limit
Non-detect	Non-detect	Various	Kept; used 50% of reported detection limit
O	Sampled for, but analysis lost	STORET	Eliminated
Q	Sample held beyond normal holding time	STORET	Eliminated
T	Value reported is less than criteria of detection	STORET	Kept; used 50% of reported detection limit
U	Material was analyzed for, but not detected	STORET	Kept; used 50% of reported detection limit
W	Value observed is less than lowest value reportable under "T"	STORET	Kept; used 50% of reported detection limit
Y	Sample analyzed, but was not properly preserved; may not be accurate	STORET	Eliminated

*Only three observations were found that bore this data remark code.

one of the analytical measurements was retained for that agency (generally the largest sample contributor). Stata® programs were developed to create the nutrient groups, convert all reporting units to "as N" or "as P", and to prevent sample double counts.

Entire analytical measurements were eliminated (those in gray-shaded areas; Table 2) if a clear definition for the measurement could not be located. And although placed in the nitrate & nitrite group, nitrite-only measurements were completely excluded from use. In most ambient waters exposed to oxygen, nitrite is only present in trace quantities and most dissolved inorganic N is nitrate (Horne and Goldman, 1994). A review of the database showed that most nitrite measurements were very low or below the reported detection limit. Therefore, by aggregating analytical measurements that jointly report nitrite + nitrate (e.g., parameter code 00630; Table 2) with measurements that only report nitrate (e.g., 00618), we assumed that the nitrite + nitrate samples were mostly nitrate.

Development of Seasonal Periods to Partition Nutrient Data

Nutrient concentrations in flowing waters can show distinct seasonal patterns (Lohman and Priscu, 1992; Horne and Goldman, 1994). Our objective here was to define seasonal (time) periods for each level III ecoregion, which we assumed would reduce intra-ecoregional variability in nutrient concentrations. Hydrological, biological and climatic data were all used to derive starting and ending dates of each season. Data from United State Geological Survey (USGS) gauge stations were used to address the hydrologic aspect. Two conditions were established to select the USGS gauge stations used to define flow patterns. First, each gauge station had to have at least 5 years of continuous flow records, although the stations did not need to be sampled up to the present (e.g., a continuous record from 1942 to 1963 was acceptable). Second, gauge stations were selected from stream segments having no major hydrologic modifications such as dams. Every effort was made to ensure that the selected stations provided good spatial coverage of each ecoregion, while at the same time meeting the conditions listed above. All together, 63 USGS gauge stations were selected (Appendix A), with from 10 to 12 stations per ecoregion. Two ecoregions (Idaho Batholith and Wyoming Basin; Woods *et al.*, 2002) have very limited geographic extents in Montana, however, and only six and three suitable gauge stations, respectively, could be located.

Flow duration hydrographs based on daily-mean flows were developed for each station in order to derive onset and termination dates for the runoff period. These hydrographs were developed using the complete period of record of gauge-station flow data extracted from the USGS's National Water Information System (NWIS) database. For each hydrograph, the average of all daily flow records was calculated separately for each day of the year. Each of the long-term average daily flows calculated in this manner was then plotted, and the hydrograph curve thus generated represented the average annual flow pattern at the station for the period of record (Figure 3). The two points of greatest inflection on the hydrographs were used to define the runoff onset and termination dates (e.g., day 101 and 205; Figure 3).

TABLE 2. Nutrient Groups Developed for This Study.

STORET Parameter Code	STORET Descriptor 1	STORET Descriptor 2	STORET Descriptor 3	Nutrient Reporting Units	Functional Groups
00610	NH_3 + NH4	N-TOTAL	MG/L	N	Ammonia Group
71845	AMMONIA	TOT-NH_4	MG/L	NH_4	
00608	NH_3 + NH_4	N-DISS	MG/L	N	
71846	AMMONIA	DISS-NH_4	MG/L	NH_4	
00625	TOT KJEL	N	MG/L	N	Total Kjeldahl Nitrogen Group
00605	ORG-N	N	MG/L	N	
00607	ORG-N	DISS-N	MG/L	N	
00623	KJELDL N	DISS	MG/L	N	
00624	KJELDL N	SUSP	MG/L	N	
00600	TOTAL N	N	MG/L	N	Total N Group
00602	DISS.	NITROGEN	MG/L N	N	
71887	TOTAL N	AS NO_3	MG/L	NO_3	
00630	NO_2 + NO_3	N-TOTAL	MG/L	N	Nitrate & Nitrite Group
00631	NO_2 + NO_3	N-DISS	MG/L	N	
00618	NO_3-N	DISS	MG/L	N	
00620	NO_3-N	TOTAL	MG/L	N	
71850	NITRATE	TOT-NO_3	MG/L	NO_3	
71851	NITRATE	DISS-NO_3	MG/L	NO_3	
00613	NO_2-N	DISS	MG/L	N	
71856	NITRITE	DISS-NO_2	MG/L	NO_2	
00615	NO_2-N	TOTAL	MG/L	N	
00665	PHOS TOT		MG/L P	P	Total P Group
71886	TOTAL P	AS PO_4	MG/L	PO_4	
00669	PHOS TOT	HYDRO	MG/L P	P	
00678	PHOS TOT	HYDRO + ORTH	MG/L P	P	
00671	PHOS-DIS	ORTHO	MG/L P	P	Soluble Reactive Phosphate Group
70507	PHOS-T	ORTHO	MG/L P	P	
00660	ORTHOPO_4	PO_4	MG/L	PO_4	
00650	T PO_4	PO_4	MG/L	PO_4	
00666	PHOS-DIS		MG/L P	P	Total Dissolved P Group

Nutrient analytical measurements shown in gray were not used. All values were converted to elemental reporting units (i.e., as N, as P) prior to data analyses.

Total number of samples found in MT: Ammonia Group, 18,647; TKN Group, 19,462; TN Group, 6226; Nitrate Group, 29,798; TOTAL P Group, 24,453; SRP Group, 15,361; TDP Group, 6071.

After the hydrologically based dates for the onset and termination of runoff were compiled, it became obvious that the runoff termination dates suggested by some of the flow-duration hydrographs located in the mountainous ecoregions (Northern, Middle, and Canadian Rockies) extended longer into the summer than the MT DEQ has generally found there to be discernable scouring effects on aquatic life. Therefore, we turned to biological data to further define the seasons. The MT DEQ uses June 21st as the start date for biological sampling in streams of mountainous regions of the state (MT DEQ, Standard Operating Procedures for Sample Collection, Sorting, and Taxonomic Identification of Benthic Macroinvertebrates, Water Quality Planning Bureau, WQPBWQM-009, April 2005), as runoff effects have typically subsided by that time. A number of hydrographs in the mountainous ecoregions showed that runoff was still occurring on June 21st. Therefore, for ecoregions in which this occurred, we selected the runoff termination date as the earliest day after June 21st on which all flow-duration hydrographs in the ecoregion were at least on the declining limb of the peak flow.

The selection of the start-of-winter dates could not be readily determined using hydrograph characteristics. After runoff ends, base flow begins and can be fairly uniform into November and December (day 235-365; Figure 3). However, regional climatic influences such as lowered temperatures and light intensity typically cause by the end of September major reductions in aquatic plant life growth, as well as reductions in aquatic macroinvertebrate productivity (Richards, 1996). In general, the MT DEQ uses September 21st as the termination date for biological sampling (Standard Operation Procedures, cited

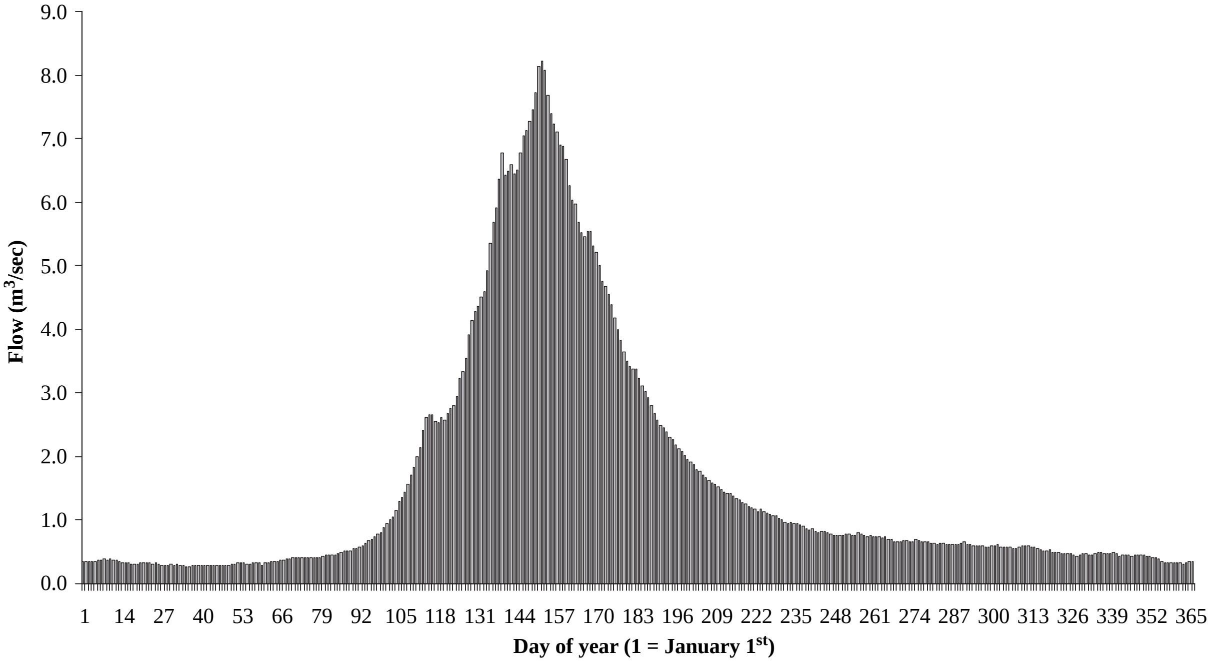

FIGURE 3. Example Daily-Means Flow Duration Hydrograph. Data are for the entire period of record (1982-2003) for USGS Gauge Station 12381400.

above), and only rarely collects biological samples after October 1st. After having examined the hydrological, biological and climatic factors discussed, the onset and termination dates of the seasons were finalized for each ecoregion. The onset and termination dates were then rounded to the nearest end-of-month or mid-month date (Table 3).

Nutrient data in the databases were associated with the appropriate season by their dates of collection. Significant differences (95% confidence) between seasons were tested using the Kruskal-Wallis test (Conover, 1999). Kruskal-Wallis tests were performed on the nutrient general population separately for each level III ecoregion; that is, the data were first stratified by ecoregion before the significance of seasonal groups was tested. (The tests for the reference population had very low power because of the reference population's small size, and the results are not presented here.) Kruskal-Wallis tests were performed for the general population in the all-observations database and the median database.

Selection of Reference Sites

The identification and assessment of Montana reference stream sites is discussed in detail in Suplee *et al.* (2005), and will be only briefly summarized here. A group of candidate reference stream sites was assembled and then assessed using a consistent set of criteria that included both quantitative and qualitative evaluations. Data were examined at two scales: site specific, and watershed (5th or 4th hydrologic unit codes; Seaber *et al.*, 1987). The qualitative component was undertaken by using best professional judgment (BPJ) to assess criteria such as "presence of point sources," "grazing use," "aesthetics," "condition of stream bank vegetation," and "mining

TABLE 3. Starting and Ending Dates for Three Seasons (Winter, Runoff, and Growing), by Level III Ecoregion.

Ecoregion Name	Start of Winter	End of Winter	Start of Runoff	End of Runoff	Start of Growing Season	End of Growing Season
Canadian Rockies	October 1	April 14	April 15	June 30	July 1	September 30
Northern Rockies	October 1	March 31	April 1	June 30	July 1	September 30
Idaho Batholith	October 1	April 14	April 15	July 15	July 16	September 30
Middle Rockies	October 1	April 14	April 15	July 15	July 16	September 30
Northwestern Glaciated Plains	October 1	March 14	March 15	June 15	June 16	September 30
Northwestern Great Plains	October 1	February 29	March 1	June 30	July 1	September 30
Wyoming Basin	October 1	April 14	April 15	June 30	July 1	September 30

impacts." Quantitative analyses consisted of watershed-level assessments and a site-specific analysis. At the watershed level, the proportion of agricultural land use and the total density of roads (km/km^2) was determined for the watershed upstream of each candidate reference site using a GIS. Criteria were then located in the literature (Kershner *et al.*, 2004; Sheeder and Evans, 2004) to estimate thresholds for impacts to aquatic life and other beneficial water uses. At the site-specific level, water quality data for each site were reviewed to determine if they exceeded state water quality standards (MT DEQ (Montana Department of Environmental Quality), 2006) for a suite of metals contaminants commonly released from mining areas (Cd, Cu, Pb, Zn, Hg, and dissolved Al).

Some candidate reference sites were in a reference condition for certain characteristics (e.g., riparian condition), but failed in another category, for example having high density of abandoned mines in the watershed or metals concentrations that exceeded the state standards. Sites of this nature were not retained as reference sites. That is, none of the reference sites that passed to the final list contained any "fatal" flaws, and only sites passing all criteria were included. The final reference site list contained streams ranging in stream-order size (Strahler, 1964) from 1st to 6th, which generally comprised wadeable streams and small rivers. All data associated with reference sites were flagged in the Stata® database to distinguish them from nonreference population data. The locations of reference sites are shown in Figure 2.

Percentile Mapping: Reference-to-General Population

Percentile mapping is the identification of corresponding percentile values of equivalent nutrient concentrations in two different data distributions. Percentile mapping for the reference-to-general population was carried out in two major steps. In the first step, summary statistics were computed for nutrient groups in the reference, nonreference, and general (reference plus nonreference) populations by each alternative stratification methodology (i.e., combinations of ecoregions and seasons). Specific summary statistics included the total number of observations, minimum, maximum, mean, standard deviation, and skewness. The summary statistics also included concentrations at the 25th, 50th, and 75th percentiles for reference, nonreference and general population observations. Percentile mapping was only undertaken when four or more nutrient observations were available at nonreference and reference locations.

In the second step, the reference and general population frequency distributions were matched within each stratification combination. Stata® programs were developed to compute the nutrient concentrations corresponding to the 75th and 90th percentiles of the reference population. Next, an empirical cumulative distribution function was generated to assign a percentile rank to each nutrient concentration observation in the general population. The percentiles in the general population corresponding to the nutrient concentrations at the 75th and 90th percentiles in the reference distribution were then determined using a linear interpolation method. A cubic interpolation method was also tested. However, in most cases, the cubic interpolation method did not differ from the linear method and it resulted in missing values in a few boundary cases. Therefore, the linear interpolation method was exclusively applied for this analysis.

Percentile Mapping: Case Studies-to-Reference Population

Four conditions were used to select stressor-response case studies that were used to make comparisons against the reference-population frequency distributions. These were: (1) the case study reported a scientifically defensible linkage between nutrient concentrations and an impact to a beneficial water use (e.g., recreation & aesthetics, aquatic life, fisheries); (2) each case study's geographic extent was within a level III ecoregion found in Montana; (3) the stream or river in the case study generally fell within the scope of the present work (i.e., similar Strahler stream order); and (4) the case study was documented in some kind of publication. The nutrient concentrations recommended in or derived from these case studies were then mapped to their corresponding concentrations in the reference-population frequency distributions from the same ecoregion and season. In cases where more than one percentile in the reference distribution had the same concentrations (e.g., both the 50th and 75th percentile were equal to 0.05 mg total P/L), the higher percentile was selected.

Five scientific case studies that met the conditions for use were located for four different level III ecoregions. Welch *et al.* (1989) modeled the influence of SRP concentrations on periphyton biomass in the Spokane River of Idaho and Washington. The Spokane River is a sixth-order river in the Northern Rockies ecoregion, which extends into Montana. Watson *et al.* (1990) used artificial stream channels utilizing water from the Clark Fork River in Montana (4th-7th-order) and control nutrient inputs (N and P) to determine the peak biomass of diatom algae and the filamentous algae *Cladophora*. Dodds *et al.* (1997) used a river and stream database comprised of sites from North America, Europe, and New Zealand to develop regression equations between nutrients and algal standing crop, and then recommend criteria for Montana's reach of the Clark Fork River. Based on a 16-year study, Sosiak (2002) recommended P concentrations intended to maintain algae density below nuisance levels in the Bow River (5th order; Alberta, Canada). Lastly, Suplee (2004) presented a regression equation between standing crop of algae and nitrate concentrations in Montana prairie streams (3-4th order), and recommended maximum concentrations for total N, total P and algal standing crop.

Other Descriptive Statistics and Statistical Analyses

As described earlier, we generated summary statistics for nutrient concentrations in the all-observations database for each alternative stratification methodology (i.e., combinations of ecoregions and seasons). This was also carried out for the median database. In addition, we were concerned that nutrient data from large rivers (Strahler order 7 and 8), for example the Missouri and Yellowstone rivers, might bias comparisons between the general and reference-population frequency distributions. (Recall that the reference sites came from first through sixth-order streams and small rivers). Therefore, we also generated summary statistics from an all-observations dataset that excluded data from seventh and eighth-order rivers. Statistically significant differences (95% confidence level) between nutrient concentrations of the reference and general populations were determined using the Wilcoxson ranksum test (Conover, 1999).

RESULTS

Seasonal Differences in Nutrient Concentrations

The results of the Kruskal-Wallis tests for ecoregionally stratified seasonal differences in nutrient concentrations are presented in Tables 4a and b. For nutrient zones based on level III ecoregions, there were significant seasonal differences in median nutrient concentrations in the general population. This was true for the majority of cases in the all-observation database, and for many cases in the median database. In the all-observations database, the majority of the nutrient groups showed significant seasonal differences for each level III ecoregions, except for the Wyoming Basin (Table 4a). The Wyoming Basin has a very limited geographic extent in Montana, which resulted in low power of the tests. For other nutrient groupings for which the trends are not significant, mainly in the median database, the results may reflect the low power of the tests because of the relatively small sample sizes associated with those nutrients.

Percentile Mapping, Descriptive Statistics and Statistical Test Results

Based on the all-observations database, Tables 5a through 5d present the 75th and 90th reference percentile equivalents in the general population for all seven nutrient groups in each level III ecoregion,

TABLE 4. Significance of Seasonal Stratification by Level III Ecoregions

	Nutrient Group*							
Level III Ecoregion	**Ammonia**	**$NO_3 + NO_2$**	**TKN**	**Total N**	**SRP**	**TDP**	**Total P**	**Proportion of Nutrient Groups Showing Significant Differences in Seasons (%)**
(**a**) All-observations database, general population								
Northern Rockies	Y	Y	Y	Y	Y	N	Y	86
Idaho Batholith	Y	Y	Y	N	Y	N	Y	71
Middle Rockies	Y	Y	Y	Y	Y	N	Y	86
Wyoming Basin	N	Y	N	Y	N	N	Y	43
Canadian Rockies	Y	Y	N	Y	Y	N	Y	71
Northwestern Glaciated Plains	Y	Y	Y	Y	Y	Y	Y	100
Northwestern Great Plains	Y	Y	Y	Y	N	Y	Y	86
(**b**) Median database, general population								
Northern Rockies	N	Y	Y	N	N	N	Y	43
Idaho Batholith	N	N	Y	N	N	N	Y	29
Middle Rockies	Y	Y	Y	N	Y	N	Y	71
Wyoming Basin	N	N	N	N	N	N	Y	14
Canadian Rockies	N	Y	N	N	N	N	N	14
Northwestern Glaciated Plains	N	Y	Y	N	Y	N	Y	57
Northwestern Great Plains	Y	Y	Y	Y	N	Y	Y	86

*"Y" means the median concentrations are different between seasons (Kruskel-Wallis test, 95% confidence interval).
"N" means the median concentrations are not significantly different between seasons.

TABLE 5. Cross-Nutrient Percentile Mapping for Level III Ecoregions.

Nutrient Group	Level III Ecoregion* and Reference Percentiles									
	Northern Rockies		Middle Rockies		Canadian Rockies		Northwestern Glaciated Plains		Northwestern Great Plains	
	75th	90th	75th	90th	75th	90th	75th	90th	75th	90th
(**a**) All seasons										
Ammonia	–	–	18	49	13	13	81	91	44	67
$NO_3 + NO_2$	32	47	38	59	19	26	49	74	74	82
SRP	15	16	23	23	46	54	64	80	73	81
TKN	27	49	44	60	52	66	84	91	77	93
Total N	–	–	66	79	62	77	66	85	74	95
Total P	4	4	29	54	27	31	83	92	86	96
TDP	–	–	38	56	–	–	84	95	96	98
Mean:	19	29	36	54	36	44	73	87	75	87
1 SD of mean:	13	23	16	17	20	25	14	8	16	11
CV (%):	65	78	44	31	54	56	19	9	21	13
(**b**) Winter season										
Ammonia	–	–	17	45	–	–	83	85	47	61
$NO_3 + NO_2$	–	–	28	38	9	15	75	79	72	77
SRP	–	–	9	25	15	18	52	78	81	89
TKN	–	–	39	54	–	–	81	88	86	94
Total N	–	–	–	–	–	–	88	97	81	91
Total P	–	–	16	45	6	7	76	85	90	94
TDP	–	–	–	–	–	–	74	88	97	98
Mean:	–	–	21	41	10	14	75	86	79	86
1 SD of mean:	–	–	12	11	5	6	12	6	16	13
CV (%):	–	–	54	26	49	43	15	7	20	15
(**c**) Runoff season										
Ammonia	–	–	19	51	–	–	87	92	16	45
$NO_3 + NO_2$	30	47	39	64	27	35	65	76	54	77
SRP	19	21	19	39	59	70	76	88	71	80
TKN	–	–	47	62	–	–	75	90	89	97
Total N	–	–	–	–	–	–	57	73	91	97
Total P	6	6	33	55	35	42	80	90	91	98
TDP	–	–	40	40	–	–	90	97	89	96
Mean:	18	25	33	52	40	49	76	87	72	84
1 SD of mean:	12	21	12	11	17	19	12	9	28	20
CV (%):	66	84	35	20	41	38	15	10	40	23
(**d**) Growing season										
Ammonia	–	–	34	51	23	23	77	85	57	79
$NO_3 + NO_2$	28	28	33	58	19	27	34	56	80	89
SRP	11	12	27	27	44	51	43	82	74	82
TKN	24	44	41	71	60	70	86	91	69	94
Total N	–	–	68	80	85	91	61	89	81	96
Total P	5	15	19	38	27	30	81	91	87	96
TDP	–	–	–	–	–	–	80	95	96	98
Mean:	17	25	37	54	43	49	66	84	78	90
1 SD of mean:	11	15	17	20	26	27	20	13	13	7
CV (%):	65	60	45	37	60	56	31	16	16	8

For each ecoregion and nutrient, value shown is the percentile in the general data population matching the 75th or 90th percentile of the reference population, respectively.

*Results for the Wyoming Basin and the Idaho Batholyth ecoregions not shown, as there were too few reference observations ($n < 4$) to undertake the matching process. Dashes in the table indicate too few observations ($n < 4$) to undertake analysis.

for all seasons combined (Table 5a) and for each season (Tables 5b through 5d). Reference-to-general population matches for specific nutrients were highly variable between ecoregions, as nutrient concentrations at the 75th percentile of reference corresponded to general-population percentiles ranging from the 4th percentile to the 97th percentile. In general, the mountainous ecoregions (Northern, Middle and Canadian Rockies) showed greater separation between reference and general-population data than did the two prairie ecoregions (Northwestern Glaciated and Great plains). That is, general population streams in mountainous ecoregions had elevated nutrient concentrations relative to their corresponding reference streams whereas, in the prairie ecoregions, nutrients in reference and general-population streams were much more similar. Furthermore, the cross-nutrient standard deviations (and coefficient of variation, CV) around the mean of the mapped percentiles were fairly low in the two prairie ecoregions (see bottoms of Tables 5a to 5d). It is also apparent from Tables 5a through 5d that seasonal trends were not very pronounced in the percentile mappings. The only exceptions to this finding were for the Middle Rockies and the Canadian Rockies, where general-population percentiles corresponding to the 75th and 90th percentiles in the reference population were lower in the winter season than for other seasons. In another analysis not presented here, cross-ecoregional percentile mapping (e.g., grouping all total P percentile matches together across ecoregions) showed that, for a given nutrient, the cross-nutrient standard deviation around the mean in a given ecoregion was generally lower than the cross-ecoregional standard deviation around the mean.

There were only a limited number of cases (11%) for which the 75th percentile of the reference population mapped closely (±5 percentiles) to the 25th percentile of the general population (Tables 5a through 5d). Similarly, of 19 aggregate cross-nutrient results (see "Mean" rows, Tables 5a through 5d) there was only one case (Middle Rockies, winter season) where the 75th percentile of reference population closely mapped (±5 percentiles) to the 25th percentile of the general population.

Tables 6a through 6c show nutrient concentrations (all seasons) at the 25th, 50th, and 75th percentiles of the reference and nonreference populations, for each ecoregion. Table 6a was generated from the all-observations database, Table 6b from the same but excluding stream order 7 & 8 data, and Table 6c was generated from the median database. Overall, all three databases produce very comparable results. One anomaly in the datasets is the fact that TKN concentrations are often higher than TN in equivalent ecoregions and seasons. This resulted because TN data have generally been collected more recently, and have relatively low detection limits, whereas TKN was part of many older datasets, and TKN detection limits where commonly higher in the past. Table 7 shows the results of significance comparisons between reference and nonreference populations (all seasons), by ecoregion, for the all-observations and median databases. (Significance tests were performed for the all-observations database excluding stream order 7 & 8 data, but the results were virtually identical to the all-observations database and are not shown.) Although there was 100% agreement in significance-test results between the all-observations and median databases for the Canadian Rockies, in the remaining ecoregions there was disagreement between database results in about 35% of cases. For the great majority of nutrients in the mountainous ecoregions (Northern, Middle, and Canadian Rockies), there were significant differences between the reference and nonreference nutrient concentrations (Table 7). However, in the two prairie ecoregions (Northwestern Glaciated and Great plains), nutrient concentrations in the reference and nonreference populations were significantly different in only half of the cases or less.

The results of the case studies-to-reference population mapping are shown in Table 8. As for the reference-to-general population mapping, these results are based on the all-observations database. Case studies were located for four of Montana's seven level III ecoregions (Northern Rockies, Middle Rockies, Canadian Rockies, and the Northwestern Glaciated Plains). Overall, nutrient concentrations from case studies mapped to nutrient concentrations in reference-population distributions across a much smaller range than was observed for the reference-to-general population mappings. The case studies-to-reference population mappings ranged from the 68th to the 99th percentiles (Table 8). Overall, nutrient concentrations from the case studies mapped to the 86th (mean) and 86th (median) percentile of the reference populations, with a CV of 13% (Table 8, bottom row).

DISCUSSION

The databases used in the present study comprised data from longitudinal samplings of the same streams, most data were not sampled probabilistically and therefore a number of samples are not truly independent. Our goal, however, was to create a nutrient database having the greatest possible spatial and temporal coverage of the state. To achieve this, we assembled data from as many organizations as possible, over the greatest possible period of time, knowing that each organization had its own sampling goals, objectives and timeframes. We assumed that compiling data from many organizations would minimize bias associated with any one organization's dataset. Even probabilistically collected datasets may contain some type of bias. For example, the one truly probabilistic dataset we incorporated (EMAP; 2000-2004) was entirely collected during a statewide dry cycle when moderate to extreme drought was common (hydrological drought index; Palmer, 1965; NCDC (National Climate Data Center), 2006). In contrast, our database contained data collected during

TABLE 6. Nutrient Concentrations (mg/L) in Reference and Non-reference Populations (All Seasons) for Selected Percentiles of the Frequency Distributions. Data Are Organized by Level III Ecoregion*

Nutrient Group	Northern Rockies						Middle Rockies						Canadian Rockies						Northwestern Glaciated Plains						Northwestern Great Plains					
	Reference			Non-Reference			Reference			Non-Reference			Reference			Non-Reference			Reference			Non-Reference			Reference			Non-Reference		
	25th	50th	75th	25th	50th	75th	25th	50th	75th	25th	50th	75th	25th	50th	75th	25th	50th	75th	25th	50th	75th	25th	50th	75th	25th	50th	75th	25th	50th	75th
(**a**) All-observations database																														
Ammonia	–	–	–	0.005	0.010	0.020	0.002	0.002	0.005	0.005	0.020	0.050	0.004	0.005	0.005	0.010	0.010	0.030	0.020	0.050	0.120	0.010	0.035	0.090	0.005	0.010	0.030	0.010	0.040	0.100
$NO_3 + NO_2$	0.008	0.010	0.020	0.020	0.050	0.100	0.005	0.020	0.040	0.020	0.080	0.230	0.004	0.007	0.010	0.040	0.050	0.100	0.001	0.005	0.050	0.020	0.050	0.200	0.020	0.090	0.250	0.030	0.070	0.260
SRP	0.006	0.007	0.007	0.010	0.010	0.020	0.002	0.005	0.010	0.010	0.020	0.060	0.001	0.002	0.002	0.005	0.010	0.010	0.005	0.007	0.020	0.005	0.010	0.030	0.010	0.020	0.030	0.010	0.010	0.030
TKN	0.050	0.050	0.100	0.100	0.200	0.400	0.050	0.200	0.210	0.130	0.300	0.500	0.050	0.050	0.200	0.080	0.200	0.600	0.700	1.000	1.500	0.480	0.800	1.200	0.270	0.575	1.100	0.400	0.680	1.100
Total N	–	–	–	0.130	0.220	0.370	0.065	0.085	0.175	0.050	0.090	0.280	0.050	0.060	0.090	0.060	0.080	0.240	0.615	0.760	1.120	0.600	0.900	1.400	0.520	0.780	1.300	0.610	0.890	1.300
Total P	0.003	0.003	0.003	0.010	0.020	0.040	0.008	0.010	0.020	0.020	0.040	0.080	0.001	0.002	0.003	0.010	0.010	0.020	0.040	0.060	0.140	0.020	0.050	0.100	0.030	0.060	0.170	0.020	0.040	0.090
TDP	–	–	–	0.010	0.020	0.030	0.010	0.020	0.020	0.010	0.030	0.040	–	–	–	0.020	0.020	0.020	0.010	0.030	0.050	0.010	0.020	0.030	0.030	0.050	0.140	0.010	0.020	0.030
(**b**) All-observations database excluding data collected from streams/rivers of strahler stream order 7 & 8																														
Ammonia	–	–	–	0.005	0.010	0.030	0.002	0.002	0.005	0.010	0.020	0.050	0.004	0.005	0.005	0.005	0.010	0.030	0.020	0.050	0.120	0.010	0.040	0.100	0.005	0.010	0.030	0.010	0.040	0.100
$NO_3 + NO_2$	0.008	0.010	0.020	0.020	0.050	0.110	0.005	0.020	0.040	0.020	0.070	0.200	0.004	0.007	0.010	0.030	0.050	0.100	0.001	0.005	0.050	0.020	0.050	0.200	0.020	0.090	0.250	0.020	0.050	0.240
SRP	0.006	0.007	0.007	0.010	0.010	0.030	0.002	0.005	0.010	0.010	0.030	0.060	0.001	0.002	0.002	0.005	0.005	0.010	0.005	0.007	0.020	0.005	0.010	0.030	0.010	0.020	0.030	0.010	0.010	0.030
TKN	0.050	0.050	0.100	0.100	0.200	0.500	0.050	0.200	0.210	0.050	0.200	0.450	0.050	0.050	0.200	0.060	0.120	0.400	0.700	1.000	1.500	0.500	0.810	1.300	0.270	0.575	1.100	0.430	0.700	1.700
Total N	–	–	–	0.170	0.270	0.410	0.065	0.085	0.175	0.050	0.090	0.200	0.050	0.060	0.090	0.060	0.080	0.240	0.615	0.760	1.120	0.640	0.930	1.400	0.520	0.780	1.300	0.630	0.910	1.300
Total P	0.003	0.003	0.003	0.010	0.010	0.050	0.008	0.010	0.020	0.010	0.030	0.070	0.001	0.002	0.003	0.010	0.010	0.020	0.040	0.060	0.140	0.020	0.050	0.100	0.030	0.060	0.170	0.020	0.040	0.080
TDP	–	–	–	0.010	0.010	0.030	0.010	0.020	0.000	0.020	0.030	0.050	–	–	–	0.005	0.010	0.015	0.010	0.030	0.050	0.010	0.020	0.030	0.030	0.050	0.090	0.010	0.020	0.030
(**c**) Median database																														
Ammonia	–	–	–	0.005	0.010	0.020	0.005	0.005	0.007	0.005	0.020	0.050	0.004	0.004	0.005	0.005	0.010	0.020	0.005	0.015	0.055	0.010	0.030	0.060	0.005	0.010	0.010	0.010	0.020	0.050
$NO_3 + NO_2$	0.007	0.050	0.115	0.015	0.045	0.130	0.010	0.021	0.055	0.020	0.050	0.130	0.007	0.010	0.040	0.035	0.050	0.105	0.003	0.005	0.020	0.010	0.050	0.185	0.005	0.040	0.100	0.020	0.057	0.200
SRP	–	–	–	0.010	0.010	0.020	0.003	0.009	0.014	0.010	0.020	0.050	0.002	0.003	0.005	0.005	0.010	0.010	0.003	0.009	0.016	0.010	0.030	0.070	0.004	0.010	0.030	0.010	0.020	0.035
TKN	0.050	0.050	0.100	0.100	0.200	0.500	0.100	0.200	0.215	0.095	0.240	0.450	0.050	0.100	0.200	0.050	0.110	0.300	0.565	1.000	1.660	0.335	0.700	1.132	0.345	0.530	1.565	0.280	0.522	1.000
Total N	–	–	–	0.100	0.150	0.250	0.070	0.090	0.260	0.065	0.115	0.270	0.050	0.070	0.090	0.060	0.080	0.090	0.590	0.720	0.940	0.680	0.975	1.300	0.290	0.620	1.250	0.500	0.810	1.200
Total P	0.001	0.003	0.010	0.010	0.010	0.030	0.006	0.010	0.022	0.010	0.030	0.060	0.001	0.002	0.004	0.009	0.010	0.020	0.015	0.055	0.110	0.020	0.050	0.100	0.030	0.040	0.150	0.020	0.040	0.090
TDP	–	–	–	0.010	0.020	0.030	–	–	–	0.020	0.040	0.090	–	–	–	0.020	0.020	0.020	0.020	0.025	0.175	0.020	0.020	0.050	–	–	–	0.010	0.020	0.030

*Results for the Wyoming Basin and the Idaho Batholyth ecoregions not shown; too few reference observations ($n < 4$). Dashes in the table indicate too few observations ($n < 4$) to generate distribution.

TABLE 7. Statistically Significant Differences Between Nutrient Concentrations of Reference and Non-reference Populations (All Seasons Combined) in Montana Level-III Ecoregions*.

Nutrient Group	Northern Rockies		Middle Rockies		Canadian Rockies		Northwestern Glaciated Plains		Northwestern Great Plains	
	All-observations Database†	Median Database†	All-observations Database†	Median Database†	All-observations Database†	Median Database†	All-observations Database†	Median Database†	All-observations Database†	Median Database†
Ammonia	–	–	Y	Y	Y	Y	N	N	N	Y
$NO_3 + NO_2$	Y	N	Y	Y	Y	Y	Y	Y	N	N
SRP	Y	–	Y	Y	Y	Y	Y	Y	N	N
TKN	Y	Y	Y	N	N	N	Y	N	N	N
Total N	–	N	N	N	N	N	N	N	N	N
Total P	Y	Y	Y	Y	Y	Y	Y	N	Y	N
TDP	–	–	N	–	–	–	Y	N	Y	–

Results from the all-observations and median databases are shown. "Y" Indicates a significant difference in concentrations, " N" Indicates an insignificant difference (Wilcoxson-Ranksum Test, 95% confidence level).

*Results for the Wyoming Basin and the Idaho Batholyth ecoregions not shown; too few reference observations (n < 4).

†Dashes in the table indicate too few reference observations (n < 4) to generate distribution and conduct test.

numerous wet/dry climatic cycles, including several periods of extreme drought and extreme moisture (Palmer, 1965; NCDC (National Climate Data Center), 2006). Drought, and precipitation patterns in general, can influence water quality (Ojima *et al.*, 1999; Little *et al.*, 2003), and our database is capable of reflecting these influences because of its relatively long period of record.

In its guidance on the development of river and stream nutrient criteria, the U.S. EPA has recommended that for any given physiographic region the 75th percentile of a reference-site frequency distribution be selected or, alternatively, the 5th to 25th percentile of the general-population frequency distribution (U.S. EPA, 2000b). This recommendation assumes that either method "should approach a common reference condition along a continuum of data points" (page 95, U.S. EPA, 2000b). This presumption is based on three case studies — one in Tennessee, one in Minnesota, and one in New York — where it was found that the 75th percentiles of the reference site frequency distributions for nutrients closely matched to the 25th percentile of the general population frequency distributions (U.S. EPA, 2000a,b,c). However, two of these three case studies are from lakes (New York and Minnesota), waterbody types that are different from rivers and streams. Aside from the vast body of scientific literature on the topic of lotic and lentic waters, the fundamental difference between rivers/streams and lakes is illustrated by the fact that the U.S. EPA has developed its nutrient criteria recommendations separately for each of these two waterbody types (e.g., U.S. EPA, 2000a,d). Therefore, it is questionable whether the finding in lakes that nutrient concentrations at the 75th percentile of a reference population are similar to nutrient concentrations at the 25th percentile of the general population can be, unexamined, directly transferred to rivers and streams. The remaining case study (Tennessee) was undertaken in rivers and streams using an approach similar to ours. However, when the reference-to-general population nutrient relationship was examined for Tennessee's level III ecoregions, only three out of four of Tennessee's ecoregions showed a close match between the 75th percentile of reference and the 25th percentile of the general population (Appendix A; U.S. EPA, 2000b). Similarly, an analysis of reference and general-population nutrient data for small streams in parts of North Carolina and Tennessee shows that the 75th percentile of the reference distribution matches to about the 45th and 40th percentile of the general population for TN and TP, respectively (Rohm *et al.*, 2002).

The use of the 5th to 25th percentile of a general population frequency distribution to identify nutrient criteria is a secondary approach, to be used when

TABLE 8. Case-Study Nutrient Concentrations Mapped to Their Corresponding Percentile Value in the Reference Site Population Frequency Distributions.

			Reference Stream Population			
Case Study	Nutrient	Case-Study Nutrient Concentration (mg/L)	Season of Application	Level III Ecoregion	Percentile in Reference Distribution Matching Case-Study Concentration	Notes on Case Studies*
Welch *et al*. (1989)	SRP	0.010	Year Round	Northern Rockies	99	SRP concentration would constrain river distance with algal biomass exceeding 200 mg Chl *a* m^{-2} to under 10 km.
Watson *et al*. (1990)	SRP	0.011	Growing	Middle Rockies	94	Artificial stream study. SRP concentration corresponding to algal standing crop of 150 mg Chl *a* m^{-2}.
Dodds *et al*. (1997)	TN	0.275	Year Round	Middle Rockies	84	Based on regression equation, concentration is intended to maintain algal standing crop<100 mg Chl *a* m^{-2} (max).
	TP	0.035	Year Round	Middle Rockies	88	Based on regression equation, concentration is intended to maintain algal standing crop<100 mg Chl *a* m^{-2} (max).
Sosiak (2002)	TP	0.018	Year Round	Canadian Rockies	99	Based on regression equation, concentration intended to maintain algal standing crop<150 mg Chl *a* m^{-2} on the Bow River near Calgary, Alberta, Canada.†
Suplee (2004). Technical Report	$NO_2 + NO_3$	0.006	Growing	Northwestern Glaciated Plains	68	Based on nitrate-benthic algae regression equation, concentration would maintain maximum algal standing crop<150 mg Chl *a* m^{-2}.‡
	TN	1.04	Growing	Northwestern Glaciated Plains	73	Concentration extrapolated from $NO_2 + NO_3$ concentration.
	TP	0.153	Growing	Northwestern Glaciated Plains	83	Concentration extrapolated from $NO_2 + NO_3$ concentration.
				Mean:	86	
				Median:	86	
				1 SD:	11	
				CV (%):	13	

*Stream benthic algae densities above 100-150 mg Chl *a* m^{-2} are reported to exceed a nuisance threshold (Horner *et al*., 1983). Algae densities above 200 mg Chl *a* m^{-2} are reported to impact trout habitat (Biggs, 2000a).

†The Bow River at Calgary is downstream of the boundary of the level III ecoregion 'Canadian Rockies'. The Sosiak (2002) TP recommendation was assigned to the Canadian Rockies ecoregion since the majority of the river's drainage upstream of Calgary is within the Canadian Rockies ecoregion.

‡Additional data were collected at sites described in the Suplee (2004) report after the report was completed. Subsequent analysis of the larger dataset showed that 90% of the maximum algae densities in the reference sites were < 150 mg Chl *a* m^{-2}. Therefore, 150 mg Chl *a* m^{-2} was used to derive the $NO_2 + NO_3$ concentration shown here.

reference data are unavailable (U.S. EPA, 2000a). Our results and those of Rohm *et al.* (2002) demonstrate that caution should be taken when using this general-population approach to selecting criteria because, in effect, it creates a "moving target" because of its complete reliance upon the degree of eutrophication prevalent when the data were collected (Dodds and Oakes, 2004). If the ecoregion in question has not had a substantial degree of eutrophication, then the 25th percentile of the general population will result in overly restrictive criteria; Figure 4 demonstrates this point. In Figure 4, the reference and general population distributions for TN in the Northwestern Glaciated Plains of Montana overlap a great deal. The 75th percentile of the reference population maps to about the 63rd percentile of the general population, and so the general population 25th percentile represents an unduly restrictive criterion. The corollary to this is that in highly eutrophied regions, the general-population 25th percentile is probably not sufficiently protective of water beneficial uses. How one would go about systematically selecting more restrictive criteria (e.g., the 5th percentile) in the absence of reference sites, at least using these statistically based approaches, is not entirely clear in the U.S. EPA's guidance (U.S. EPA, 2000*b*).

Results from the present study also illustrate that it is not always easy to predict upfront, for any particular ecoregion, what the reference-to-general population relationship for any given nutrient will be. Prior to the analysis of Montana's data, we would have predicted — based on our general understanding of land use in Montana — that the prairie region east of the Rocky Mountain Front would have demonstrated a greater degree of elevated nutrients than the western, mountainous region of the state. The two prairie ecoregions comprising most of eastern Montana's land area (Northwestern Glaciated Plans and Northwestern Great Plains) are almost entirely used for grazing, dry-land agriculture (cereal crops such as wheat and barley) and, to a lesser degree, irrigated agriculture, and we assumed that nutrients in those ecoregions' streams would be highly elevated relative to their corresponding reference streams. However, we found that in these two ecoregions the reference and general-population nutrient concentrations were significantly different in only about a third of the cases (Table 7), much less often than was observed in the mountainous ecoregions of the state. There are four likely explanations for this: (1) the reference sites were poorly selected and actually represent eutrophied conditions; (2) most of the nutrients were sequestered by heavy growth of algae and aquatic plants and nutrient concentrations were, consequently, low; (3) not all nutrients are good indicators of regional eutrophication, and special attention should be paid to certain nutrient groups; or (4) the region — as a whole — is not as heavily eutrophied as initially thought.

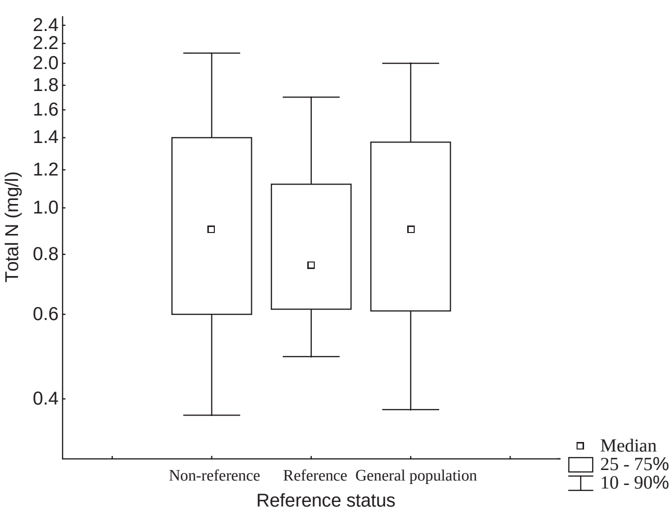

FIGURE 4. Comparison of Total N Concentrations in Reference and Nonreference Stream Sites of the Northwestern Glaciated Plains Ecoregion in Montana. Data shown are for all seasons.

Of these four possibilities, the latter two are probably closest to the truth, and can be exemplified using the Northwestern Glaciated Plains ecoregion. To address the first possibility, two specific reference sites demonstrate the overall quality of the reference sites. The reference site "Rock Creek below Horse Creek, Near Int. Boundary" (USGS gauge station 06169500) is a USGS Hydrologic Benchmark Network (HBN) site located on the U.S.-Canadian border in the Northwestern Glaciated Plains ecoregion. The HBN network comprises stream sites located in relatively undeveloped basins which serve as controls for separating natural from human-caused changes in stream water quality (Alexander *et al.*, 1996; Clark *et al.*, 2000). Much of Rock Creek's watershed upstream of the site is contained within the Grasslands National Park of Canada (Parks Canada – Grasslands National Park of Canada, website, http://www.pc.gc.ca/pn-np/sk/grasslands, accessed October 21, 2005), and only about 7% is used for crop agriculture (U.S. and Canada combined). The reference site "Bitter Creek" (same ecoregion) has as its immediate upstream drainage a land area that has been described by the Montana Natural Heritage Program (a branch of the Nature Conservancy) as the largest intact grassland in northern Montana, and one of the most extensive naturally functioning glaciated plains grasslands in North America (Cooper *et al.*, 2001). Bitter Creek's drainage is not used for dry-land or irrigated agriculture, and grazing use of the area is highly compatible with natural ecological processes that maintain grasslands of this type (Cooper *et al.*, 2001). These two stream sites are arguably as close to true reference as one is likely to find today in the Northern Great Plains. Available nutrient concentration data (all seasons) from these two sites were combined, and the 75th percentile of four nutrient groups — TN, TP, SRP and $NO_2 + NO_3$ — were matched to their corresponding general-population data in the Northwestern Glaciated Plains ecoregion. The four nutrient groups matched to the 84th, 78th, 58th, and 39th percentiles, respectively. As an aggregate, nutrient concentrations in Rock and Bitter creeks matched to the 65th percentile of the general population, lower than the percentile for the aggregate of all reference sites in the ecoregion (73rd; Table 5a) but clearly not at the 25th percentile. So even when nutrient data from the very best reference sites of the Northwestern Glaciated Plains ecoregion in Montana are examined, their frequency distributions overlap a great deal with the general population, which suggests that the general-population 25th percentile would represent too stringent criteria.

Regarding the second possibility, the winter season data do not support the assertion that nutrients were sequestered in dense growths of algae and aquatic plants. The winter season for the Northwestern Glaciated Plains (October 1st to March 14th; Table 3) occurs when aquatic plant growth has greatly slowed due to low light and freezing temperatures, and so the plant's ability to sequester nutrients and diminish water-column concentrations is negligible. Because soluble nutrients are most biologically available, they are probably the most sensitive measure of potential nutrient uptake by aquatic plants. In the winter season, the concentration at the 75th percentile of reference for ammonia, $NO_3 + NO_2$ and SRP matched to the 83rd, 75th, and 52nd percentiles of the general population (Table 5b). If general population streams were highly eutrophied and had heavy algal and aquatic plant growth taking up nutrients in the growing season, the plants' uptake would not be manifested in winter and one might expect soluble nutrient concentrations to become elevated in the winter season. The net result would be that reference site concentrations would match to much lower general-population percentiles (i.e., more like Figure 1) in winter than we observed.

Concerning the third possibility, note that $NO_3 + NO_2$ was significantly different between reference and nonreference sites in the Northwestern Glaciated Plains (Table 7). $NO_3 + NO_2$ is also, among the seven nutrient groups in Tables 5a and 5d, the nutrient showing the greatest separation from the 75th percentile of the reference sites. Suplee (2004) showed that $NO_3 + NO_2$ is significantly correlated to algae density in the region's streams, and another study in the ecoregion found that dryland crop-fallow practices elevate nitrate concentrations in soil porewater and groundwater (Nimick and Thamke, 1998). These facts suggest that special attention should be paid to this particular nutrient, as it is the one most likely to be linked to eutrophication problems in the region.

Finally, the fourth possibility can best be gauged relative to other parts of the state. In the mountainous ecoregions, which have forestry activities and also comprise intermountain valleys that have substantial agricultural activity, reference and nonreference streams were significantly different for many more nutrient groups than was found to be the case for the Northwestern Glaciated Plains. Furthermore, the reference 75th percentiles of the mountainous ecoregions mapped to much lower percentiles in their corresponding general populations than was observed in the Northwestern Glaciated Plains. So, relative to the mountainous region of the state, Northwestern Glaciated Plains nutrients are not as elevated, and there are fewer nutrient groups that are elevated. One is left to conclude that, in this prairie ecoregion of Montana, eutrophication is not as severe and is more nutrient-specific than in the western, mountainous part of the state.

The idea that the water quality of reference sites should be acceptable and support all beneficial water uses is fairly intuitive. This idea is intrinsic in the U.S. EPA's recommendation that the 75th percentile of a nutrient-concentration reference distribution be used to set criteria, because the 75th percentile will assure that the majority of the nutrient data from reference sites will not exceed the criteria thresholds. Nevertheless, the 75th percentile is still a cautious (i.e., protective) approach, as 25% of nutrient data collected from reference sites could exceed the criteria. Our results indicated that a somewhat higher percentile (about the 86th) from nutrient-concentration reference distributions is more appropriate for Montana streams, as this percentile has been ground truthed to regional case studies that demonstrate nutrient impacts to beneficial water-uses.

Impact-to-use nutrient concentrations (i.e., those at or above the 86th percentile of reference in the present study) are altogether different from "pristine" nutrient concentrations. Estimates of pristine nutrients concentrations in streams are reported in the literature, however (Kemp and Dodds, 2001; Smith *et al.*, 2003; Dodds and Oakes, 2004), and some of these concentrations can be compared with the present study. The best estimate of "pristine" from our study would be approximately the 50th percentile of reference, as it represents the central tendency for groups of reference sites. In the Central Cultivated Great Plains of the United States, pristine TN concentrations are estimated to range from 200 to 566 μg/l (Kemp and Dodds, 2001; Smith *et al.*, 2003; Dodds and Oakes, 2004), whereas this study suggests 760 μg/l (Northwestern Glaciated Plains; 50th percentile; Table 6a). Pristine TP concentrations for the same region range from 23 to 58 μg/l (Smith *et al.*, 2003; Dodds and Oakes, 2004), while this study suggests 60 μg/l (Table 6a). In the Western Forested Mountains of the United States, the results of this study are lower than other literature values. For example, in the Western Forested Mountains pristine concentrations range from 19 to 45 μg/l (Smith *et al.*, 2003; Dodds and Oakes, 2004), and this study suggests 3-10 μg/l (Northern, Middle and Canadian Rockies; Table 6a).

We acknowledge that in some regions of the United States (like Montana) the possibility of locating reference sites is much greater than in areas having widespread intensive agriculture (e.g., the U.S. corn belt). The process of identifying appropriate nutrient criteria in areas of intensive agriculture is clearly challenging, and although difficult to accomplish it would be prudent in such regions to try to locate at least a few reference sites, so that some sense of the reference-to-general population relationship can be developed. If this cannot be done, another approach would be to model the factors controlling a region's water quality and then factor out the affects of land use (e.g., Robertson *et al.*, 2001; Dodds and Oakes, 2004) or, alternatively, develop stressor-response models (e.g., Biggs, 2000b; Dodds *et al.*, 2002) between nutrients and demonstrable impacts to beneficial water uses.

In conclusion, our findings indicated that the relationship between nutrient concentrations in reference populations and nutrient concentrations in their corresponding general populations can vary a great deal from ecoregion to ecoregion. We found in this study that the 75th percentile of reference corresponded to the 4th to 97th percentile of the general population. Further, an expected relationship between reference and general population nutrient data — based on an *a priori* understanding of land use in a region — may not always manifest itself as anticipated. As a result, if the 25th percentile of a general-population frequency distribution is used to establish nutrient criteria, then the resulting nutrient standards could be overly stringent or insufficiently protective, depending upon what the actual relationship between the reference and the general population looks like. On the other hand, nutrient concentrations derived from five regionally applicable scientific studies (nutrient as stressor, impact to a beneficial water use as response) fell within a relatively narrow band around the 86th percentile of the reference-site nutrient frequency distributions. The latter result indicated that nutrient concentrations at high percentiles of reference-site frequency distributions (this study suggests the 86th) represent, fairly consistently, the threshold where impacts to beneficial water uses begin to occur.

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APPENDIX A. List of USGS Gauge Stations Used to Define Flow Patterns of Level III Ecoregions in Montana.

USGS Station	Station Name	Flow Data Range	Years of Data	Ecoregion (III)
12354000	St Regis River near St. Regis, MT	1910-2003	94	15
12390700	Prospect Creek at Thompson Falls, MT	1956-2003	48	15
12389500	Thompson River near Thompson Falls, MT	1911-2003	93	15
12301999	Wolf Creek near Libby, MT	1967-77	11	15
12304500	Yaak River near Troy, MT	1956-2003	48	15
12366000	Whitefish River near Kalispell, MT	1928-2003	76	15
12301300	Tobacco River near Eureka, MT	1959-2003	45	15
12391550	Bull River near Noxon, MT	1973-82	10	15
12374250	Mill Creek above Bassoo Creek, near Niarada, MT	1982-2003	22	15
12300500	Fortine Creek near Trego, Mt	1947-53	7	15
12351500	Lolo Creek near Lolo, MT	1911-15	5	15
12388400	Revais Creek below West Fork near Dixon, MT	1983-2003	21	15
12343400	East Fork Bitterroot near Conner, MT	1956-2003	48	16
06024500	Trail Creek Near Wisdom, MT	1948-72	25	16
12345000	Rock Creek near Darby, MT	1946-59	14	16
12347500	Blodgett Creek near Corvallis, MT	1947-69	23	16
12349500	Fred Burr Creek near Victor, MT	1947-51	5	16
12350500	Kootenai Creek near Stevensville, MT	1949-63	15	16
12381400	South Fork Jocko River near Arlee, MT	1982-2003	22	17
12332000	Middle Fork Rock Creek near Philipsburg, MT	1938-2003	66	17
06015500	Grasshopper Creek near Dillon, MT	1921-61	41	17
06013500	Big Sheep Creek below Muddy Creek near Dell, MT	1936-79	44	17
06037500	Madison River near West Yellowstone, MT	1913-2001	89	17
06209500	Rock Creek near Red Lodge, MT	1932-2003	72	17
06035000	Willow Creek near Harrison, MT	1938-2002	65	17
06055500	Crow Creek near Radersburg, MT	1901-90	90	17
06071000	Little Prickly Pear Creek near Canyon Creek, MT	1909-24	16	17
06077000	Sheep Creek near White Sulphur Springs, MT	1941-72	32	17
06154410	Little Peoples Creek near Hays, MT	1972-89	18	17
06207540	Silver Tip Creek near Belfry, MT	1967-75	9	18
06207500	Clarks Fork Yellowstone River near Belfry, MT	1921-2003	83	18
06207510	Big Sand Cl at WY-MONT State line	1973-81	9	18
06078500	North Fork Sun River near Augusta, MT	1911-93	83	41
05011500	Waterton River near International Boundary	1947-64	18	41
12359000	South Fork Flathead River at SBRS, near Hungry Horse, MT	1948-67	20	41
12361000	Sullivan Creek near Hungry Horse, MT	1948-76	29	41
12357000	Middle Fork Flathead at Essex, MT	1940-64	25	41
12355500	North Fork Flathead near Columbia Falls, MT	1910-2003	94	41
05010000	Belly River at International Boundary	1947-64	18	41
12382000	Middle Fork Jocko River near Jocko, MT	1912-16	5	41
05014500	Swiftcurrent Creek at Many Glacier, MT	1912-2002	91	41
06072000	Dearborn River AB Falls Creek, near Clemons, MT	1908-12	5	41
06180000	West Fork Poplar River near Richland	1935-49	15	42
06168500	Rock Creek at International Boundary	1914-61	48	42
06142400	Clear Creek near Chinook, MT	1984-2002	19	42
06154400	Peoples Creek near Hays, MT	1966-2003	38	42
06176500	Wolf Creek near Wolf Point, MT	1908-92	85	42
06185110	Big Muddy Creek near mouth near Culbertson, MT	1981-92	12	42
06183800	Cottonwood Creek near Dagmar, Mt	1985-2003	19	42
06170200	Willow Creek near Hisdale, MT	1965-73	9	42
06099000	Cut Bank Creek at Cut Bank, MT	1905-2003	99	42
06133500	North Fork Milk River AB St. Mary Ca near Browning, MT	1911-2002	92	42
06107000	North Fork Muddy Creek near Bynum, MT	1912-24	13	42
06129500	McDonald Creek at Winnett, MT	1930-56	27	43
06336500	Beaver Creek at Wibaux, MT	1938-84	47	43
06307600	Hanging Woman Creek near Birney, MT	1973-95	23	43
06126470	Halfbreed Creek near Klein, MT	1978-91	14	43
06121000	American Fork near Harlowton, MT	1907-32	26	43
06111000	Ross Fork Creek near Hobson, MT	1946-62	17	43
06294995	Armells Creek near Forsyth, MT	1974-95	22	43
06287500	Soap Creek near St. Xavier, MT	1911-72	62	43
06324500	Powder River at Moorhead, MT	1929-2003	75	43
06334000	Little Missouri River near Alzada, MT	1911-69	59	43

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HOW GREEN IS TOO GREEN? PUBLIC OPINION OF WHAT CONSTITUTES UNDESIRABLE ALGAE LEVELS IN STREAMS[1]

Michael W. Suplee, Vicki Watson, Mark Teply, and Heather McKee[2]

ABSTRACT: A public opinion survey was carried out in Montana to ascertain if the public identifies a level of benthic (bottom-attached) river and stream algae that is undesirable for recreation. The survey had two parts; an On-River survey and a By-Mail survey. The On-River survey was conducted via 44 trips randomly scheduled throughout the state during which recreators were interviewed in-person at the stream. Selection of stream segments and survey dates/times was based on known, statewide recreational use patterns. By-Mail survey forms were sent to 2,000 individuals randomly selected from Montana's Centralized Voter File (CVF) available from the Montana Secretary of State. The CVF was current through 2004 and represented over 85% of the state's eligible voting population. In both surveys, eight randomly ordered photographs depicting varying levels of stream benthic algae were presented, and participants were asked if the algae level shown was desirable or undesirable for recreation. Survey form design, selection of photographs, and pretesting followed acceptable protocols that limited unintentional bias through survey execution. There were 433 returned forms (389 complete) for the By-Mail survey, while the On-River survey documented 563 interviews. In both surveys, as benthic algal chlorophyll *a* (Chl *a*) levels increased, desirability for recreation decreased. (Other measures of benthic algae biomass are presented as well.) For the public majority, mean benthic Chl *a* levels ≥200 mg/m^2 were determined to be undesirable for recreation, whereas mean levels ≤150 mg Chl *a*/m^2 were found to be desirable. Error rates were within the survey's statistical design criteria (≤5%). The largest potential error source was nonresponse in the By-Mail survey; however, the population represented by nonrespondents would have to exhibit profoundly different perceptions of river and stream algae to meaningfully alter the results. Results support earlier work in the literature suggesting 150 mg Chl *a*/m^2 represents a benthic algae nuisance threshold.

(KEY TERMS: rivers/stream; algae; environmental regulations; environmental impacts; public participation.)

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INTRODUCTION

One of the most basic components of water quality protection in the United States (U.S.) and abroad is the establishment of water body beneficial uses, which are also referred to as instream values. For example, the U.S. Clean Water Act (1972) requires that water bodies be classified for the type of beneficial water uses they are to support (e.g., fisheries, aquatic life, recreation and aesthetics, and drinking). Language of a similar nature is provided in New Zealand's 1991 Resource

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[2]Respectively, Environmental Science Specialist, Montana Department of Environmental Quality, 1520 E. 6th Ave., Helena, Montana 59620; Professor, Environmental Studies, University of Montana, 32 Campus Dr., Missoula, Montana 59812; Statistician, 5131 Chambers Cr Lp SE, Olympia, Washington 98501; and Graduate Research Assistant, Environmental Studies, University of Montana, 32 Campus Dr., Missoula, Montana 59812 (E-Mail/Suplee: msuplee@mt.gov).

Management Act. Within the more detail-oriented text of the administrative rules, regulations, and guidance documents that support these laws are found water-quality criteria. Water-quality criteria are numeric or narrative expressions that, if met, assure protection of the beneficial water uses. The 1972 Clean Water Act is overseen by the U.S. Environmental Protection Agency (USEPA), and many water-quality criteria are provided in USEPA's so-called blue, red, and gold books (USEPA, 1973, 1976, 1986). These three documents indicate that the protection of recreation and aesthetics in U.S. waters requires the prohibition of undesirable or nuisance aquatic life (e.g., algae blooms). Similarly, New Zealand's guidance on interpreting the 1991 Resource Management Act (Biggs, 2000) addresses nuisance proliferations of periphtyon (i.e., stream bottom-attached algae) and recommends appropriate criteria to protect stream aesthetic, recreational, and landscape values. In Australia, guidelines for fresh and marine water quality indicate that nuisance organisms (including filamentous algal mats) should not be present in excessive amounts (Australian Department of the Environment, Water, Heritage and the Arts, available at http://www.mincos.gov.au/__data/assets/pdf_file/0003/316128/wqg-ch5.pdf, *accessed* September 30, 2008). In Montana, where the work to be presented took place, regulations that support the Montana Water Quality Act prohibit human caused conditions that result in undesirable aquatic life (Administrative Rules of Montana 17.30.637[1][e], available at http://www.deq.state.mt.us/dir/Legal/Chapters/CH30-06.pdf, *accessed* October 29, 2007).

What exactly constitutes an undesirable or nuisance level of aquatic life in a water body can be a subjective matter, especially when it comes to protecting beneficial water uses such as recreation and aesthetics. Nevertheless, work has been published in the scientific literature describing levels of benthic (i.e., bottom attached) algae in rivers and streams that may constitute a nuisance (Horner *et al.*, 1983; Welch *et al.*, 1988). These two papers suggest that benthic algae levels in excess of 100-150 mg chlorophyll *a* (Chl *a*)/m^2 are a nuisance. Horner *et al.* (1983) reviews 26 citations describing benthic algae growth in natural and artificial streams and finds that benthic algae levels greater than 150 mg Chl *a*/m^2 are only reported in cases where nutrient enrichment was above "ordinary natural levels" (page 131). Following up on this work, Welch *et al.* (1988) indicate that biomass greater than 100-150 mg Chl *a*/m^2 corresponds to streambed algae coverage >20%, which they suggest may present an aesthetic nuisance. Although algae of 100-150 mg Chl *a*/m^2 may impact recreation and aesthetics, impacts by such algae levels on aquatic life is unclear (Nordin, 1985; Welch *et al.*, 1988; Quinn and Hickey, 1990).

Horner *et al.* (1983) and Welch *et al.* (1988) – and the 100-150 mg Chl *a*/m^2 level they suggest to prevent nuisance growth – have been widely cited in the scientific literature and in government documents (e.g., Welch *et al.*, 1989; New Zealand Ministry for the Environment, 1992; Watson and Gestring, 1996; Dodds *et al.*, 1997; Biggs, 2000; Dodds and Welch, 2000; USEPA, 2000; Sosiak, 2002; Dodds, 2006; Carey *et al.*, 2007; Suplee *et al.*, 2007; Wang *et al.*, 2007). But Horner *et al.* (1983) recognized that establishing a nuisance algae level required further field verification, and government documents suggesting the use of the 100-150 mg Chl *a*/m^2 range qualify the recommendations by noting that what constitutes too much algae to the general public has not been firmly established, or stating that these levels will probably protect an aesthetic beneficial use (Biggs, 2000; USEPA, 2000). Thus, some type of assessment of the public's opinion on the matter is clearly warranted.

In our study, photographs of benthic stream algae at different levels were used to assess public opinion of what might constitute a nuisance. Independent studies show a high degree of consistency between perceptual judgment of photographs of environmental scenes and perceptual judgment of the same scenes experienced directly (Zube, 1974; Shuttleworth, 1980; Kellomäki and Savolainen, 1984; Stewart *et al.*, 1984; Stamps, 1990). Photographs preclude the need to transport large numbers of study participants to the environmental sites in question (Shuttleworth, 1980; Daniel and Meitner, 2001), and can be used to show conditions that may not currently exist (Manning and Freimund, 2004). The latter point was particularly relevant to our study, as some benthic algae (e.g., the filamentous algae *Cladophora* sp.) can demonstrate peak levels that develop rapidly in early summer and then again in early fall (Whitton, 1970), the timing of which is highly variable. It would have been very difficult to coordinate the study with such time-variable events.

The use of photographs to represent environmental scenes has the advantages outlined above, but is not without shortcomings. Photographs cannot invoke dynamic elements like sound, motion, or other factors which can be significant components especially in environments involving streams and rivers (Hetherington *et al.*, 1993; House, 1996). In general, the validity of using a particular presentation medium (like photographs) depends on how well that medium can convey the key components of an environmental scene to the participants who are judging specific aspects of the scene (Hetherington *et al.*, 1993; Manning and Freimund, 2004). We concluded that photographs would convey the "key components" needed for study participants to judge what were (or were not) undesirable algae levels, as previous work consistently discuss/present nuisance algae in contexts that can readily be assessed by eye. For example, large benthic algal growths interfere with swimmers and boats (physical entanglement of both), are unpopular with fisherman because of the danger of slipping and the snagging of lines, and are very conspicuous (unaesthetic) from the bank (e.g., Whitton, 1970; Horner *et al.*, 1983; Biggs and Price, 1987; Welch *et al.*, 1988; Biggs, 2000). All these factors can be assessed visually in a quality photograph. Furthermore, the New Zealand Ministry for the Environment uses photographs of varying benthic algae levels to convey to the public the appearance of different algae quantifications (Chl a/m^2, % bottom cover, etc.) (New Zealand Ministry for the Environment, 1992; Biggs, 2000).

Herein, we present results from a survey that assessed the public's opinion concerning river and stream benthic algae levels. The objective was to determine if the general public identifies a particular level of benthic algae that is not desirable for recreation. In summer 2006, the Montana Department of Environmental Quality (MT DEQ) and the University of Montana surveyed the public on their perceptions of benthic algae in rivers and streams as it affected water recreational activities, whatever those activities might be. To our knowledge, this is the only large-scale research that has explored the relationship between public perceptions of benthic algae levels in rivers and streams and recreation water uses. Study surveys were carried out on two groups that were not mutually exclusive: river and stream users throughout Montana, and registered Montana voters. We found that the public majority showed a clear preference for benthic algae levels at or below 150 mg Chl a/m^2. These findings provide strong support for the more qualitatively derived recommendations of Horner *et al.* (1983) and Welch *et al.* (1988).

METHODS

Overview of Survey Goals and Design

The survey was carried out in Montana (Figure 1) on two public groups that were not mutually exclusive. An On-River survey was carried out on wadeable rivers and streams throughout the state, and a By-Mail survey was sent to randomly selected registered Montana voters. The first was undertaken because the opinion of active river and stream users was considered particularly relevant. This group included Montana residents and visitors. The second group (registered Montana voters) was chosen because the outcome of the survey had the potential to impact Montana water quality regulations, and therefore the opinion of a representative sample of Montanans was important. We also tracked in the On-River survey the opinions of residents *vs*. nonresidents to elucidate if actively recreating Montanans had opinions different from visitors. In both surveys we tracked opinions by region and by watershed, as the location where a public opinion survey is carried out can significantly influence the results (Ross and Taylor, 1998; Brunson and Shindler, 2004), and we wanted to be able to test for this.

Both surveys consisted of the same randomly ordered photographs of Montana rivers and streams, each photograph depicting a different algae level. In the On-River survey, the survey's purpose and instructions were verbally provided to participants in-person by an interviewer; for the By-Mail survey, these were provided on the survey form. In each survey, participants were asked to indicate if the algae level in each photograph was desirable or undesirable in relation to their main form of river and stream recreation. We did not specify which recreation, thus allowing survey participants to respond relative to whatever form of river and stream recreation they enjoyed. The terms desirable and undesirable (as opposed to alternatives like acceptable/unacceptable) were chosen because they have long been used in U.S. national water quality criteria (e.g., "surface waters should be free of substances attributable to discharges or wastes [which] … produce undesirable aquatic life") (Federal Water Pollution Control Administration, 1968; USEPA, 1973, 1976, 1986), and

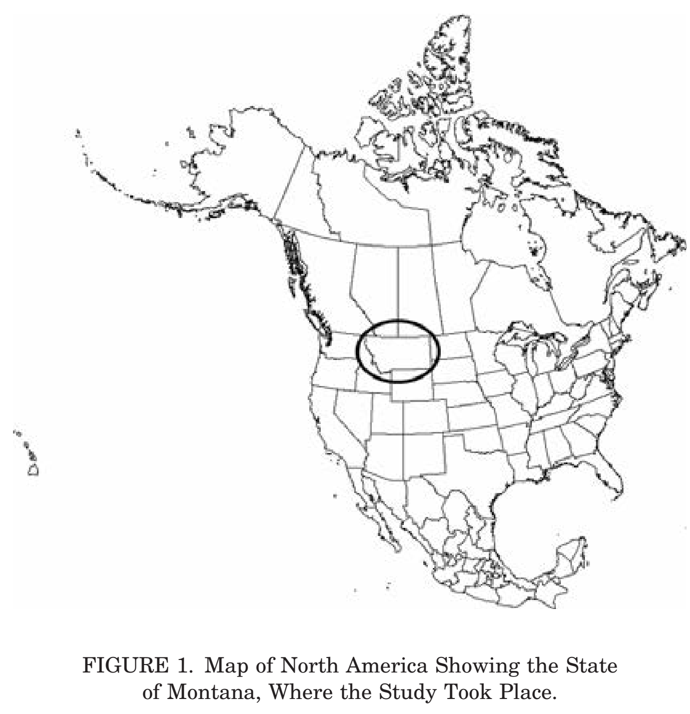

FIGURE 1. Map of North America Showing the State of Montana, Where the Study Took Place.

we believed they would be easily understood for making a choice. Error rates for responses to each photograph were targeted to be ≤5%. Further details about each survey are provided later in Methods.

Selection of Photographs for the Survey

Photographs representing a range of algae levels found in Montana rivers and streams were selected from the collection of one of the authors. At each photographed site, 10-20 benthic algae samples had been collected and analyzed so that the benthic algal Chl *a* (extracted with EtOH and corrected for phaeophytins) (Sartory and Grobbelaar, 1984) and ash-free dry weight (AFDW) (Clesceri *et al.*, 1998) of the stream cross-section seen in each photograph was known. Many different sites had been sampled over a number of years, providing a large collection of photographs. The mean of the repeat measures of algae at each site during any given sampling event provided a benthic Chl *a* density (mg/m^2) and AFDW (g/m^2) for each photograph. Photographs of river and stream sites were available that showed a range of mean benthic Chl *a* from <50 mg/m^2 to 1,276 mg/m^2. This generally covers the maximum range of benthic algae measured in MT rivers and streams. Photographs included streams in which bottom algae was filamentous, diatomaceous or, often, a combination of both.

Photographs were sorted by visual clarity and consistent perspective, and then grouped into benthic Chl *a* "bins" (ca. 50 mg Chl *a*/m^2 bin, ca. 100 mg Chl *a*/m^2 bin, ca. 150 mg Chl *a*/m^2 bin, etc.). Algae were staggered by about 50 mg Chl *a*/m^2 as it is the authors' experience that a visual distinction can best be made between algae levels staggered at this degree of resolution. A "zero" level was not provided because levels below 50 mg Chl *a*/m^2 are, in our experience, difficult to visually discern from 50 mg Chl *a*/m^2. Each algae bin was initially represented by between 5 and 20 photographs. The photographs were then provided to a review committee (MT DEQ Water Quality Standards Section; six individuals, one an author on the present study). Each member was asked to identify a photograph for each bin that best represented the central tendency of the series of photographs in the bin – that is, it was not too "green," and not too "clean." (The author who is a member of the Standards Section did not reveal his choices to the team prior to their selections.)

The survey was developed using eight of the committee-selected photographs. Each was assigned a letter, and are ordered here by reach mean Chl *a* values (lowest to highest): (A) 44 mg/m^2, (G) 112 mg/m^2, (F) 152 mg/m^2, (E) 202 mg/m^2, (B) 235 mg/m^2, (H) 299 mg/m^2, (C) 404 mg/m^2, and (D) 1,276 mg/m^2 (Appendix A). Other algae characterizations (g AFDW/m^2, dominant algae type, % filamentous cover) are shown in Table 1. In general, stream bottom coverage by filamentous algae is higher at higher Chl *a* levels (Welch *et al.*, 1988), and this is reflected in our photograph set. The photographs with the second-highest and highest Chl *a* values (C and D) do not follow the approximate 50 mg Chl *a*/m^2-increment pattern. This was carried out because (1) practical matters of design and simplicity kept the survey to eight photographs, (2) we wanted to have the most resolution among photographs in the mid-range algae levels, as that was where a nuisance threshold (per Welch *et al.*, 1988) was most likely to be identified, and (3) we wanted to show the public the full range of algae levels common in Montana. Variation around the reach-mean Chl *a* value for each of the eight photographs (SEM as a percent of the mean) ranged from 5 to 27% (mean 14%). Given this variation and for simplicity, for the remainder of this paper, each photograph's reach mean Chl *a* level is presented rounded to the nearest 10 mg/m^2.

Pretest of the Survey Form and Testing of Photograph Sequence

Survey form design and refinement followed generally accepted public opinion survey techniques (Dillman, 2000). A pretest of a draft By-Mail survey form was undertaken on 44 individuals in Helena, Montana. The pretest survey form closely resembled the final form in that it had an introduction and instructions, a dichotomous choice for each photograph, and presented the eight photographs in the same randomly derived order. The 44 individuals asked to take the survey were not randomly selected, but most were not directly involved with this water quality issue and so provided information on the survey form's clarity and logic. Two changes to the final survey form resulted from the pretest results. One photograph (D; 1,280 mg Chl *a*/m^2) was replaced with a photograph from the same site but looking downstream (rather than upstream) which also depicted 1,280 mg Chl *a*/m^2. This was performed because color hues of the original photograph were thought to be too bright relative to the other seven photographs and confused decision making. The other change was the addition of, in the survey form's introduction, a statement that if a stream's algae level was naturally high MT DEQ would take no action (Appendix A). This stemmed from individuals' comments that they were concerned that their answers would lead MT DEQ to chemically treat and kill algae in streams that have naturally elevated algae levels, an action they did not want to occur.

TABLE 1. Quantification of Algae Levels, Dominant Algae Type, and Reach Description From Field Notes on the Day of Sampling.

Photograph Letter	Mean Benthic Algae Level (mg Chl a/m^2)	Mean Benthic Algae Level (g AFDW/m^2)	Dominant Algae Type	Reach Description (field notes)
A	44	10	Diatoms	Almost bare to naked eye, no filaments
G	112	30	Diatoms	5-10% of rocks had *Cladophora*; the remaining rocks were bare or lightly coated with diatoms
F	152	36	Diatoms	50-80% *Cladophora* cover, but filaments only 1 cm long; the filaments were very diatom encrusted as were the rocks
E	202	95	Filamentous	20-60% *Cladophora* cover; 3-30 cm long filaments
B	235	117	Filamentous	50% *Cladophora* cover; very diatom-encrusted
H	299	209	Filamentous	30-100% *Cladophora* cover; 50-100 cm long filaments
C	404	136	Filamentous	70% *Cladophora* cover; 10-30 cm long filaments
D	1,276	221	Filamentous	90% *Cladophora* cover; 30-50 cm long filaments

Notes: AFDW, ash-free dry weight; Chl *a*, chlorophyll *a*.

To evaluate potential bias resulting from the presentation order of the survey photographs, in summer 2006, 32 recreators along the Clark Fork River in Missoula, Montana were asked for their opinion (desirable or undesirable) concerning the algae level shown in each of the eight photographs as it would affect their recreation. The eight photographs were shown on laminated 20.3 × 25.4 cm sheets and were presented in a particular random order. Later in the summer, a second randomly selected presentation order of the same eight photographs was prepared and 31 recreators along the same Clark Fork River reach were similarly interviewed. These data were not included in the By-Mail or On-River survey analyses.

By-Mail Survey

By-Mail surveys were sent to individuals randomly selected from Montana's Centralized Voter File (CVF) available from the Secretary of State. This file was current through 2004, contained about 624,000 records, and represented over 85% of the eligible voting population of Montana. The CVF list provided unbiased selection of sampling units because people on the CVF list are certified and there is minimal over representation or under representation of individuals, making it a good sample frame. Simple random sampling procedures were used to select individuals from the CVF. Sample size was determined using very conservative levels (99% confidence level, ±3% sampling error, 50/50 split, very large population >500,000) (Dillman, 2000). This calculates to 1,837 surveys; we mailed out 2,000 surveys as it was within our budget and helped assure we would ultimately achieve our goal of a 5% sampling error rate.

The cover of the By-Mail Survey (Appendix A) explained the purpose of the survey and provided instructions on how to fill it out. Inside the pamphlet, the eight photographs were presented in a random order. Next to each photograph, the respondent was asked to mark the box indicating if the algae level was desirable or undesirable relative to his or her major form of river and stream recreation. Respondents were also provided a few lines with each photograph to explain their answer, if they chose to. A return envelope with a postage stamp was included with each survey.

Survey implementation was intended to maximize response. We used as a guide generally accepted techniques from Dillman's (2000) Tailored Design Method (TDM), but departed from the TDM in some ways. Dillman (2000) calls for a five-contact approach, the last three of which are follow-ups after the survey is mailed ([3] a reminder/thank you postcard, [4] a replacement survey for nonrespondents, and [5] telephone/certified mail contact for nonrespondents). In our study, complete anonymity of respondents was deemed critical given the potential regulatory implications of the work and, in addition, anonymity may increase response rate (Kindra *et al.*, 1985) and accuracy (Kerin and Peterson, 1977). This decision precluded strict adherence to the last two TDM steps. In our study, each potential respondent was first sent a single-page letter introducing the project and notifying them that they would be receiving a survey. A week after the introductory letter, the survey forms were sent out. A week after the survey was sent, follow-up postcards were sent to everyone, encouraging recipients to complete and return their survey and thanking them if they already had. This three-contact process occurred between July 21 and August 4, 2006.

In September 2006, about 60 days after the By-Mail survey forms were mailed out, it was clear that nonresponse was high (ca. 78%). Preliminary analysis showed that response splits were far different from 50/50 for each photograph and, as a result, the response rate was already adequate to meet the study's design criteria. But we wanted to try and characterize nonrespondent opinions due to concerns about potential bias, so results from respondents were then used to estimate the number of follow-ups needed. These were calculated by specifying particular confidence levels around the response to a photograph under different scenarios (e.g., number follow-ups required to maintain 95% confidence level in the response to a photograph when nonrespondents respond X% differently than the original respondents). The calculations provided a range of follow-ups from 50 to 400. In September 2006, 150 randomly selected individuals from the CVF list were contacted by phone and asked if they would fill out and return the survey. Because the study was anonymous, this process included individuals that had already responded.

Selection of River and Stream Segments for the On-River Survey

On-River surveys were conducted via 44 survey trips randomly scheduled throughout the state. Angling is a dominant activity at fishing access sites throughout Montana (Montana Fish, Wildlife and Parks, http://www.fwp.mt.gov/content/getItem.aspx?id=11065, *accessed* April 2005), and therefore provided a good indication of relative river and stream recreation use. Selection of stream segments and survey dates for the survey was based on angling-use patterns summarized by the Montana Department of Fish, Wildlife, and Parks (FWP) (McFarland and Tarum, 2005; an updated, web-available version of the report is at http://www.fwp.mt.gov/content/getItem.aspx?id=29639). Prior to using FWP's list a few large, nonwadable river segments (as judged by the authors) were removed, as our main interest was the opinion of recreators using stream reaches similar to those in the photographs. FWP's list provided the On-River survey's sampling frame and directly informed unbiased selection of sample units (recreational use by time and location) through a two-stage random sampling scheme. Primary sample units were represented by FWP river drainages; probability of selection was proportional to angler use on wadeable streams. Within each primary sampling unit, secondary sample units were represented by streams that had been shown to have fishing pressure according to the FWP survey; probability of selection was proportional to angler use within the drainage. The randomly scheduled order in which stream reaches were to be surveyed was further scheduled (randomly) for interviews to occur either in the morning from 06:00 to 11:00, or in the afternoon and evening, 13:00 to 18:00 (Table 2).

Surveys were undertaken from June 17 to August 27, 2006. At the beginning of the survey (June 17 to June 20), a field interviewer noted that encounters with recreators would likely be more effective if the evening interview period was moved later in the day. The authors concurred and, from June 25 until the end of the survey, PM interviews were carried out from 14:00 to 19:00. Interview site and time scheduling was strictly adhered to, with only a small number of minor changes (e.g., two stream reaches within a primary sampling unit [drainage] to be sampled sequentially were performed in the reverse order from originally planned). Interviewer scheduling conflicts required that the ultimate surveys (August 28/29) be completed a week earlier (August 24/25).

On-River Survey Interview Process

River and stream segments on which survey interviews were carried out varied widely, but were often about 80 km long. Most river and stream segments had roads along them, with designated and undesignated public access points. Some segments, however, were only accessible by foot trails or by intermittent Forest Service roads crossing the water. The survey protocol reflects the diversity of accessibility to the segments.

The interviewer approached each survey segment from the headwaters and moved downstream. On a few occasions where travel time was underestimated, interviewing was begun before reaching the headwaters. The interviewer stopped at each designated or undesignated public fishing access point and interviewed any recreators present at that location. After finishing any available interviews, the interviewer proceeded to the next public access downstream. If the interviewer completed surveying along the length of the survey segment before the five-hour survey allocation, she turned around and repeated the process heading upstream.

In the case of a foot trail on a closed loop (no access from other trails/roads), the interviewer positioned herself at the trailhead to interview stream users both coming and going. If the river or stream was located along an open loop foot trail (access to other trails/roads), and there were vehicles present at the trailhead, the interviewer walked as much of the length of the stream as possible, given time and personal safety considerations.

Upon approaching a potential respondent, the interviewer identified herself as from the University of Montana, working on a project with MT DEQ. She explained to the respondent that the goal of the project was to determine if and when algae in rivers and streams was ever a nuisance to recreators.

TABLE 2. Prearranged, Randomly Derived Visitation Schedule for Stream Reaches in the On-River Survey.

Month	Drainage Trip	Drainage Name	Stream Trip	AM/PM	Stream Reach
June	17/18	Big Hole Drainage	Sat	PM	Big Hole River Sec 02
			Sun	AM	Big Hole River Sec 02
	19/20	Upper Clark Fork Drainage	Mon	PM	Rock Creek Sec 01
			Tues	AM	Little Blackfoot R Sec 02
	24/25	Bitterroot Drainage	Sat	PM	Bitterroot River Sec 02
			Sun	AM	Boulder Creek
	26/27	Madison Drainage	Mon	PM	Madison River Sec 02
			Tues	AM	Madison River Sec 02
July	1/2	Upper Yellowstone Drainage	Sat	PM	Sage Creek
			Sun	AM	Stillwater River Sec 02
	1/2	Upper Clark Fork Drainage	Sat	PM	Warm Springs Creek
			Sun	PM	Clark Fork River Sec 03
	3/4	Beaverhead Drainage	Mon	PM	Bloody Dick Creek
			Tues	AM	Poindexter Slough
	8/9	Upper Clark Fork Drainage	Sat	PM	Flint Creek Sec 01
			Sun	AM	Storm Lake Creek
	10/11	Big Hole Drainage	Mon	AM	Big Hole River Sec 03
			Tue	AM	Big Hole River Sec 02
	15/16	Bitterroot Drainage	Sat	PM	Lolo Creek
			Sun	AM	Bitterroot River Sec 02
	17/18	Beaverhead Drainage	Mon	PM	Beaverhead River
			Tue	PM	Beaverhead River
	24/25	Blackfoot Drainage	Mon	AM	Blackfoot River Sec 02
			Tue	AM	Blackfoot River Sec 02
	29/30	Upper Clark Fork Drainage	Sat	AM	Storm Lake Creek
			Sun	AM	Clark Fork River Sec 03
	31/Aug 1	Mussellshell Drainage	Mon	PM	Checkerboard Creek
			Tue	PM	Spring Creek
August	5/6	Upper Flathead Drainage	Sat	AM	M Fk Flathead River
			Sun	PM	M Fk Flathead River
	12/13	Bitterroot Drainage	Sat	PM	Lost Horse Creek
			Sun	PM	Bitterroot River Sec 02
	14/15	Upper Yellowstone Drainage	Mon	PM	Rock Creek Sec 01
			Tue	PM	Rock Creek Sec 01
	19/20	Upper Missouri Drainage	Sat	PM	Missouri River Sec 09
			Sun	AM	Missouri River Sec 09
	21/22	Upper Missouri Drainage	Mon	AM	Missouri River Sec 09
			Tue	PM	Missouri River Sec 09
	21/22	Upper Clark Fork Drainage	Mon	AM	Rock Creek Sec 02
			Tue	PM	Warm Springs Creek
	26/27	Upper Yellowstone Drainage	Sat	PM	Rock Creek Sec 01
			Sun	PM	Rock Creek Sec 01
	28/29	Upper Yellowstone Drainage	Mon	PM	Bighorn River Sec 01
			Tue	AM	Stillwater River Sec 01

Respondents were asked to examine the eight photographs provided (same photographs and order used in the By-Mail survey), and express whether they found the level of algae shown in each to be desirable or undesirable for their primary river or stream recreational activity, and why. At the end of the interview, the interviewer recorded for each respondent whether they were a Montana resident. She also gave each respondent the opportunity to ask any questions about the project. Most interviews lasted from two to three minutes, but up to 20 minutes in rare cases. Longer interviews resulted when respondents had extensive questions about the project or wanted to share their particular algae experiences. Any river user encountered who was capable of comprehending what the survey asked was encouraged to take the survey. This included some youths and several visitors from foreign countries.

Inferential Statistics

After the On-River surveys were complete and the By-Mail returns had stopped, six comparisons were evaluated using statistical test methods appropriate to binomially distributed data. These were (1) comparison of responses to a standard level of 50%, (2) comparisons of responses among photographs within each survey, (3) comparisons of responses within each survey to Chl *a* levels, (4) comparisons of responses among survey locations, (5) comparisons of responses by residency, and (6) comparisons of responses between surveys (i.e., By-Mail *vs*. On-River). Where applicable, reach-mean Chl *a* values were used in tests. Specifics are provided below.

(1) The proportion of desirable responses for each photograph was compared with a standard value of 50%. Fifty percent was selected because it represents a simple majority, which is a logical and clearly understood threshold. For binomial data, it also represents a level the difference from which represents a meaningful response; i.e., different from a coin flip. For each comparison, a two-sided null hypothesis was stated that the proportion observed was equal to 50%; the alternative hypothesis was that the proportion observed was not equal to 50%. The binomial sign test for a single sample was employed (Sheskin, 1997) using a calculated *z*-statistic, appropriate for large samples, to approximate the test-statistic. All tests used an *a priori* 5% significance level.

(2) Within each survey, the McNemar test (Sheskin, 1997) was used to evaluate the likelihood that preferences differed among photographs presented. This test was conducted for all pairs of photographs, testing the null hypothesis that there was no difference in respondents' preferences between two photograph pairs. All tests were two-sided (*a priori* 5% significance level).

(3) Within each survey, the test of significance for Kendall's tau (Sheskin, 1997) was used to evaluate the relationship (correlation) between percent desirable response by photograph and the Chl *a* levels depicted. This was evaluated for each survey testing the null hypothesis that no correlation existed.

(4) Within each survey, comparisons were conducted to evaluate whether preferences varied significantly by survey location. The *z*-test for two independent proportions was employed (Sheskin, 1997) testing the null hypothesis that there was no difference in percent desirable responses between survey results from two locations. In the case of the By-Mail survey, respondent identities were unknown; however, the main post office of origin of each returned survey had been recorded. Therefore, for By-Mail surveys, comparisons were conducted among locations defined by the post office of origin. For the On-River surveys, comparisons were conducted among responses by the drainage where the survey was conducted. All tests were two-sided (*a priori* 5% significance level).

(5) In the On-River survey, comparisons were conducted to evaluate whether preferences varied significantly by residency. The *z*-test for two independent proportions was employed (Sheskin, 1997) testing the null hypothesis that there was no difference in percent desirable responses due to residency (MT resident *vs.* non-resident). All tests were two-sided (*a priori* 5% significance level).

(6) Comparisons were conducted to evaluate whether preference for a particular photograph varied significantly (*a priori* 5% significance level) between surveys. The *z*-test for two independent proportions was employed (Sheskin, 1997), testing the null hypothesis that there was no difference in percent desirable responses to a given photograph between surveys.

RESULTS

For each photograph, the percent of desirable responses was not significantly different between the two different randomly ordered photograph presentations. Both photograph orders yielded the same relative rank of photographs based on percent desirable responses (Table 3). Therefore, the effect of photograph order was not considered significant or meaningful and was not further considered in interpreting the results.

For the By-Mail survey there were 433 returned surveys, 389 of which were complete (all answers filled out) and could be used in statistical analyses. The 150 telephone follow-ups were unsuccessful, in that only 14 individuals indicated they would fill out a provided survey and, at most, seven of these were returned. For the On-River survey, there were 563 documented interviews. Recreators of all kinds were encountered during these interviews, including wading fisherman, rafters, canoeists, kayakers, swimmers, tubers, and sightseers. Results from each survey are summarized in Tables 4 and 5, indicating for each photograph the number of respondents who considered the Chl *a* level depicted to be desirable or undesirable, the percentage of respondents who considered the level to be desirable, and the 95% confidence level of this proportion expressed as percent error. All error rates are within the statistical design criteria for the survey (i.e., less than 5%). Results from the On-River and By-Mail surveys depict similar patterns of response to the Chl *a* levels represented. For both groups of survey respondents, as algal chlorophyll levels increased the desirability for recreation

TABLE 3. Summary of Random Photograph Order Surveys.

Photograph	Chlorophyll *a* (mg/m^2)	Random Order #1 (n = 32)		Random Order #2 (n = 31)	
		Presentation Order	Percent Desirable	Presentation Order	Percent Desirable
A	40	1	100	1	97
G	110	7	97	3	94
F	150	6	78	8	77
E	200	5	63	7	48
B	240	2	41	5	35
H	300	8	22	2	16
C	400	3	9	4	10
D	1,280	4	19	6	13

TABLE 4. Summary of the By-Mail Survey, Montana Residents (n = 389).

Photograph	Chlorophyll *a* (mg/m^2)	Number Desirable	Number Undesirable	Percent Desirable	Percent Error
A	40	372	17	95.6	2.0
G	110	369	20	94.9	2.2
F	150	271	118	69.7	4.6
E	200	64	325	16.5	3.7
B	240	112	277	28.8	4.5
H	300	49	340	12.6	3.3
C	400	65	324	16.7	3.7
D	1,280	44	345	11.3	3.1

TABLE 5. Summary of the On-River Survey, Recreational River and Stream Users (n = 563).

Photograph	Chlorophyll *a* (mg/m^2)	Number Desirable	Number Undesirable	Percent Desirable	Percent Error
A	40	553	10	98.2	1.1
G	110	527	36	93.6	2.0
F	150	427	136	75.8	3.5
E	200	179	384	31.8	3.8
B	240	164	399	29.1	3.8
H	300	114	449	20.2	3.3
C	400	65	498	11.5	2.6
D	1,280	51	512	9.1	2.4

decreased. Specifically, levels of Chl *a* ≥200 mg/m^2, represented by photographs E, B, H, C, and D (Appendix A) were determined to be undesirable for recreation by both groups of survey respondents. Levels ≤150 mg Chl a/m^2, represented by photographs A, G, and F, were determined to be desirable. Results for each of the six statistical analyses described in Methods are given below.

Comparisons to a Standard Level

In all instances – all photograph results in both surveys – the proportion of desirable responses was significantly different than 50% ($p < 0.05$). Therefore, all responses can be considered meaningful in that they show significant preferences (desirable or undesirable) for each photograph.

Comparisons Among Photographs Within a Survey

Within each survey, significant differences were found between most pairs of photographs within both surveys. Exceptions occurred at the lower extreme of Chl *a* levels – A (40 mg/m^2) *vs.* G (110 mg/m^2) from the By-Mail survey – and among selected photographs considered undesirable by the public majority – E (200 mg/m^2) *vs.* C (400 mg/m^2) and H (300 mg/m^2) *vs.* D (1,280 mg/m^2) from the By-Mail survey and E (200 mg/m^2) *vs.* B (240 mg/m^2) from the On-River survey. Therefore, responses can be considered meaningful in that they show preferences that differ significantly among the photographs presented.

Comparisons of Responses to Algae Chl a *Levels*

In both surveys, the null hypothesis was rejected in favor of the one-sided alternative hypothesis that a negative correlation existed ($p < 0.05$). That is, as the Chl *a* levels depicted increased, the percent of desirable responses was shown to significantly decrease. Therefore, responses can be considered meaningful in that preferences show concordance with algae levels depicted in the photographs.

Comparisons Among Survey Locations

Table 6 summarizes By-Mail survey results showing percent desirable responses by post office of origin; 21 responses had an unidentifiable post mark and are not included in this summary. Table 7 summarizes On-River survey results showing percent desirable responses by drainage basin. Values in each table are shaded if the preference is significantly different than 50% (i.e., meaningful). In several instances – Billings, Butte, Missoula, and Kalispell postmarks and Big Hole, Bitterroot, and Upper Flathead drainages – results for all photographs were significantly different than 50% ($p < 0.05$). Otherwise, among the remaining locations, one or more results for photographs F through H – the midrange of algae levels depicted – exhibited preferences that were not significant. Small sample size is a factor in many of these negative results; however, several locations – Great Falls postmark and Beaverhead, Upper Clark Fork, and Upper Missouri drainages – had larger sample sizes and still exhibited no significant preference for one or more of these photographs.

Where photograph preferences were meaningful (i.e., significantly different from 50%) in Tables 6 and 7, the difference in percent desirable response between locations was evaluated. For the By-Mail survey, the null hypothesis was accepted in most cases; i.e., photograph preference did not vary

TABLE 6. Summary of the By-Mail Survey of Montana Residents by Post Office.

	Post Office of Response Origination								
Photograph	**Billings** (%) (***n*** **= 99**)	**Wolf Point** (%) (***n*** **= 7**)	**Miles City** (%) (***n*** **= 13**)	**Great Falls** (%) (***n*** **= 48**)	**Havre** (%) (***n*** **= 14**)	**Helena** (%) (***n*** **= 23**)	**Butte** (%) (***n*** **= 42**)	**Missoula** (%) (***n*** **= 78**)	**Kalispell** (%) (***n*** **= 44**)
A	94.9	100	100	91.7	85.7	95.7	95.2	98.7	95.5
G	93.9	100	100	95.8	92.9	100	97.6	93.6	90.9
F	70.7	100	69.2	72.9	64.3	65.2	69.0	61.5	72.7
E	20.2	14.3	15.4	18.8	7.1	13.0	23.8	12.8	9.1
B	34.3	71.4	46.2	37.5	28.6	21.7	23.8	17.9	25.0
H	10.1	14.3	23.1	16.7	14.3	4.3	21.4	11.5	11.4
C	12.1	14.3	15.4	20.8	21.4	21.7	19.0	14.1	15.9
D	7.1	28.6	7.7	14.6	7.1	0.0	23.8	11.5	11.4

Note: Values are shaded if preference is significantly different from 50%.

TABLE 7. Summary of the On-River Survey of Recreational River Users by Drainage.

	FWP Drainage									
Photograph	**Beaver-Head** (%) (***n*** **= 63**)	**Big-Hole** (%) (***n*** **= 70**)	**Bitter-Root** (%) (***n*** **= 129**)	**Black-Foot** (%) (***n*** **= 19**)	**Madison** (%) (***n*** **= 21**)	**Mussel-Shell** (%) (***n*** **= 15**)	**Upper Clark Fork** (%) (***n*** **= 83**)	**Upper Flathead** (%) (***n*** **= 67**)	**Upper Missouri** (%) (***n*** **= 83**)	**Upper Yellowstone** (%) (***n*** **= 13**)
A	100	97.1	100	100	100	93.3	95.2	98.5	98.8	92.3
G	98.4	94.3	87.6	100	100	86.7	91.6	92.5	98.8	100
F	87.3	82.9	66.7	73.7	57.1	86.7	74.7	79.1	77.1	76.9
E	52.4	30.0	17.1	15.8	23.8	33.3	39.8	19.4	48.2	30.8
B	39.7	31.4	8.5	57.9	19.0	33.3	15.7	29.9	55.4	53.8
H	25.4	12.9	8.5	0.0	19.0	46.7	18.1	22.4	42.2	15.4
C	25.4	5.7	2.3	0.0	23.8	6.7	9.6	3.0	30.1	7.7
D	22.2	5.7	2.3	0.0	19.0	0.0	6.0	6.0	18.1	15.4

Notes: FWP, Montana Department of Fish, Wildlife, and Parks.
Values are shaded if preference is significantly different from 50%.

significantly between respondents with different postmarks. Two notable exceptions were respondents' preference for photograph B (240 mg Chl a/m^2) in Billings *vs*. Missoula, and preference for photograph F (150 mg Chl a/m^2) in Wolf Point *vs*. Missoula. Conversely, for the On-River survey, many significant differences were evident between drainages. Such differences occurred for each photograph and results from every drainage differed with one or more other drainages.

In no instance, in either survey, is there a significant difference where a photograph preference from one location was desirable (>50%) and the preference from another location indicated the same photograph was undesirable (<50%) – or vice versa. Rather, significant differences indicated differences in the degree of acceptability – or unacceptability – between locations. For instance, 34.3% of Billings respondents considered photograph B (240 mg Chl a/m^2) to be desirable compared with 17.9% of Missoula respondents; this difference is significant, but the majority of both Missoula and Billings respondents consider the algae level depicted to be undesirable. Therefore, whereas comparisons indicate some variation in photograph preferences among locations – more so in the On-River survey than the By-Mail survey – results were consistent at the level of 50% (simple majority).

Comparisons by Residency

Table 8 summarizes photograph preference for 382 Montana residents surveyed and 181 nonresidents encountered. All preferences are meaningful in that they are significantly different from 50% ($p < 0.05$). Within each group, most preferences were significantly different between photograph pairs; exceptions exist at the extremes, i.e., photograph A (40 mg Chl a/m^2) *vs*. G (110 mg Chl a/m^2) and C (400 mg Chl a/m^2) *vs*. D (1,280 mg Chl a/m^2), and also for photographs E (200 mg Chl a/m^2) *vs.* B (240 mg Chl a/m^2). Results from each group are concordant with associated Chl a levels. Finally, in no instance is there a significant difference in photograph preference between Montana residents and nonresidents. Therefore, results for each group are meaningful and, between each other, show comparable preferences for the photographs presented.

Comparisons Between Surveys

Figure 2 shows the percent desirable responses for each photograph as observed in each survey. Photographs are ordered in progression of Chl a levels depicted. Both surveys exhibit a similar pattern of response; however, there are notable differences. For most individual photographs – A, F, E, H, and C – percent desirable response differed significantly between the two surveys. Surveys only agree on the level of preference for photographs G, B, and D. Also, there appears to be confusion in the By-Mail survey between photograph E (200 mg Chl a/m^2) *vs*. B (240 mg Chl a/m^2) and photograph H (300 mg Chl a/m^2) *vs*. C (400 mg Chl a/m^2). Whereas both surveys are concordant with Chl a levels depicted, the By-Mail survey does not depict a perfect relationship. Nevertheless, there is clear threshold in both surveys between those photographs considered desirable (>50%) and those considered undesirable (<50%) – a level of interpretation relevant to a simple majority. Significant differences among preference levels noted above are only indications of differences in degree of acceptability – or unacceptability. Therefore, from the perspective of a simple majority, results between surveys can be considered comparable and supportive of one another.

DISCUSSION

Sample frames provided for coverage of most Montanans and most recreational users of wadeable rivers and streams in the state. Sampling design yielded results that met statistical design criteria for precision of results. Furthermore, sample size tended to provide sufficient power for detecting differences between photographs, between groups, and between surveys. Survey form design, selection of photographs, and pretesting followed acceptable protocols that limit unintentional bias through survey execution (Dillman, 2000). Independent evaluation of photograph order indicated that unintentional bias was not introduced. Overall, the largest potential source of survey error is acknowledged to be attributable to nonresponse in the By-Mail survey. Efforts to characterize nonrespondent perceptions were unsuccessful, and a discussion of potential effects follows.

We carried out an anonymous survey, believing that it was appropriate for a regulatory government agency, would result in more accurate answers (Kerin and Peterson, 1977) and, therefore, reduce measurement error. But anonymity precluded strict adherence to the five-contact TDM (Dillman, 2000), consequently reducing the number and changing the manner of our multiple contacts. Multiple contacts are one of the most effective ways to reduce nonresponse error (Dillman, 1991), although anonymity can also reduce nonresponse error (Kindra *et al.*, 1985). Response rates were comparable in eastern

TABLE 8. Summary of the On-River Survey of Recreational River Users by Residency.

Photograph	Montana Residents (n = 382)			Nonresidents (n = 181)		
	Number Desirable	Number Undesirable	Percent Desirable	Number Desirable	Number Undesirable	Percent Desirable
A	376	6	98.4	177	4	98.2
G	354	28	92.7	173	8	93.6
F	291	91	76.2	136	45	75.8
E	123	259	32.2	56	125	31.8
B	115	267	30.1	49	132	29.1
H	78	304	20.4	36	145	20.2
C	40	342	10.5	25	156	11.5
D	33	349	8.6	18	163	9.1

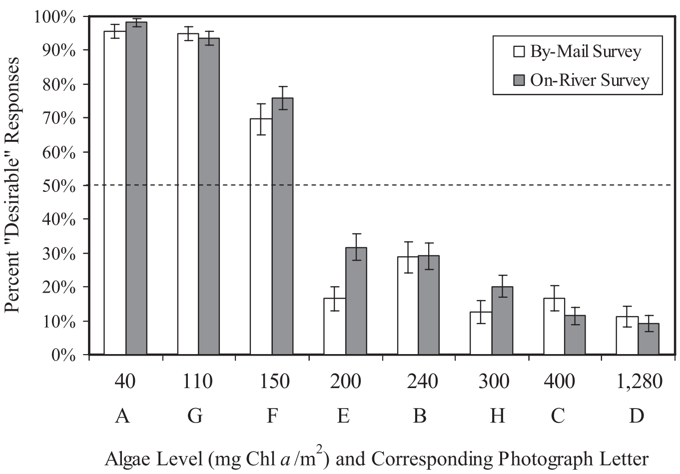

FIGURE 2. Percent Desirable Responses From the By-Mail and On-River Surveys. Letters designating the survey photographs are sequenced from lowest to highest algae level. Error bars are the 95% confidence level of each proportion, expressed as percent error.

(ca. 20%) and western (ca. 18%) Montana (each distinct geographic regions of the state; more on this below), indicating that a regional bias in nonresponse was not introduced. Overall, one can only speculate as to how nonresponse was affected by a reduced number of contacts (likely decreased response rate) *vs.* assured participant anonymity (likely increased response rate).

In general, the population represented by nonrespondents would have to exhibit a profoundly different perception of algae in wadeable rivers and streams to alter the proportions depicted in the Results at a meaningful level. Perceptions of algae by nonrespondents would have to be opposite those exhibited in the By-Mail and On-River surveys to meaningfully alter the trends observed in Figure 2. We assert that this is unlikely to occur and the fact that two different surveys – By-Mail and On-River – provide comparable results supports this assertion. A more likely outcome from the inclusion of nonrespondent perceptions would be either to shift preferences up or down (due to more or less tolerance to algae), or moderate overall preferences (due to overall indifference to algae). In either case, the proportions would still likely be significantly different than 50% (i.e., show a preference), but they would be less likely to be significant among photographs (i.e., not be distinct preferences). Overall, the same general grouping of desirable and undesirable photographs depicted in Figure 2 would likely go unchanged.

In the Introduction, we outlined some water quality regulations in the U.S. and abroad intended to protect against nuisance algal blooms and proliferations. Eutrophication of rivers and streams is a phenomenon that often leads to nuisance algae conditions. Eutrophication is the enrichment of a water body by nitrogen and phosphorus that frequently leads to increased primary productivity, i.e., increased plant growth and decay (e.g., Welch *et al.*, 1989; Chessman *et al.*, 1992; Welch, 1992; Sosiak, 2002; Dodds, 2006). So, how does the algae level found to be desirable for recreation identified in the present study (≤150 mg Chl a/m^2) compare with algae levels found in streams having varying degrees of eutrophication? Biggs (1996) reports that a group of un-enriched streams in New Zealand have a typical range of 0.5-3 mg Chl a/m^2 (median = 1.7 mg Chl a/m^2), whereas moderately enriched streams normally range from 3 to 60 mg Chl a/m^2 (median = 21 mg Chl a/m^2), and enriched streams are usually in the range of 25-260 mg Chl a/m^2 (median = 84 mg Chl a/m^2). The acceptability threshold from the present study falls within the enriched category of these streams. In Montana, Suplee *et al.* (2005) define a process for identifying reference streams and list 129 such sites around the state. Reference stream sites are, by definition, minimally impacted by human activities (Hughes *et al.*, 1986; Stoddard *et al.*, 2006) and therefore should not be very enriched relative to natural conditions. In western Montana, a region dominated by the Rocky Mountains where most streams have gravel substrates, good gradient, and support trout fisheries, 26 reference streams had a range of mean benthic algal Chl *a* levels from 3 to 75 mg Chl a/m^2 (median = 14 mg Chl a/m^2). In contrast, in northeastern Montana, which is part of the Northern Great Plains (Hunt, 1974) and is dominated by warm-water fish species (e.g., walleye) and low-gradient prairie streams, eight reference streams had a range of mean benthic algal Chl *a* levels from 2 to 302 mg Chl a/m^2 (median = 24 mg Chl a/m^2), with 97% of the sampled reaches falling below 150 mg Chl a/m^2. So in Montana, it appears that the algae level at the recreation nuisance threshold (150 mg Chl a/m^2) is much higher than what is found in mountainous reference streams, and is only rarely found in prairie reference streams.

Dodds *et al.* (1998) present a classification scheme for rivers and streams modeled after the classic one for lakes (oligotrophic or low productivity; mesotrophic or midrange productivity; and eutrophic or productive) (Wetzel, 1975). The Dodds classification was derived from a benthic algae cumulative frequency distribution for 200 streams from North America and New Zealand of varying degrees of eutrophication, and places the breaks for the three classes at the lower, middle, and upper thirds of the dataset. Interestingly, the boundary between mesotrophic and eutrophic streams was given as 200 mg Chl a/m^2 (maximum) (Dodds *et al.*, 1998) and matches the first benthic algae level in our study considered undesirable (200 mg Chl a/m^2). Further, the cumulative frequency distribution of an enlarged version of the Dodds *et al.* (1998) dataset shows that, across a set of worldwide temperate rivers and streams of varying degrees of eutrophication, there is an inflection point around 150 mg Chl a/m^2 (mean); algae levels above this value are generally uncommon (Dodds *et al.*, 2002). Thus, benthic algae levels characterized in the literature as uncommon and representing the onset of eutrophic conditions in temperate streams worldwide correspond with what the public perceived to be, in our study, the onset of excessive algal growth.

As in our study, environmental perception studies involving streams and public waters have often focused on visual characteristics that may affect public acceptability. Studies show that river and lake water color and clarity clearly influence suitability for swimming, water clarity in particular showing a distinct threshold beyond which most feel the water is unsuitable (Smith and Davies-Colley, 1992; Smith *et al.*, 1995a,b). Public enjoyment of rivers and beaches is diminished more by solid waste contaminants (e.g., toilet paper, bottles, and cans) in the water than up on the banks (House, 1996), and varying levels of solid litter at water sites (artificially placed for a study) diminish recreational values and lead participants to incorrectly assume the water itself is polluted (Dinius, 1981). Regarding the present work, participants in the On-River survey were clear about what they did not like about some of the photographs. This is illustrated by the fact that 78% had a comment about how their recreation would be interfered with by the algae levels they deemed undesirable. Some listed several reasons, but for simplicity we tally here only their first-mentioned reason; 33% stated fishing was affected (e.g., snags lures, etc.), 23% indicated wading impacts (e.g., slippery, dangerous, and would wrap around legs), 11% cited swimming interference (e.g., looks unsuitable and would get entangled), 11% stated strictly aesthetic reasons, 2% stated boating interference (e.g., entangles paddles), and 20% had comments not readily classifiable into the aforementioned groups.

The public majority showed a high degree of consistency in our study regarding what constitutes desirable and undesirable algae levels, regardless of their location in the state. For example, the majority of citizens from Billings (i.e., eastern prairie region of Montana) found benthic algae levels greater than 150 mg Chl a/m^2 to be unacceptable, as did people in mountainous western Montana (i.e., Butte, Missoula, and Kalispell). But the geography and nature of rivers and streams of these two regions is very different, and benthic algae levels from reference streams of each region have different ranges. Due to these geographic differences we had expected significant regional differences in majority public opinion, however this was not the case; only the degree to which specific algae levels were desirable or undesirable changed. Similarly, both resident and nonresident respondents identified the same maximum threshold for a desirable algae level (150 mg Chl a/m^2). These results suggest that our findings can be applied beyond Montana to small rivers and streams in northern and southern temperate regions that are of a similar nature to those shown in Appendix A.

In conclusion, statistical analysis of responses establishes that meaningful preferences were evident for the photographs presented. Proportions of "desirable" responses (i.e., those indicating that an algae level was acceptable for recreational use of a river or stream) indicated either significant satisfaction or dissatisfaction with the levels of Chl *a* depicted; except in isolated cases, preferences between photographs were significant; and, levels of preference exhibited concordance with the algae levels (i.e., more algae, less desirable). Furthermore, results showed that the acceptability of the algae levels depicted in each photograph were consistent among locations and between residents and nonresidents; acceptability was also consistent between the two surveys. It can be meaningfully concluded that, among Montanans and recreational users of Montana rivers and streams, benthic algae levels less than or equal to 150 mg Chl a/m^2 represent desirable levels for recreation while 200 mg Chl a/m^2 and higher levels are undesirable for recreational activities.

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APPENDIX A

The Eight Photographs Used in the Survey. The By-Mail survey form was a five page pamphlet with the text (shown below) on the front cover and the eight photographs (two per page) inside. Adjacent to each picture were two choices (desirable/undesirable) and a space for comments. Here, the photographs are lettered and shown in the same order as they appeared in the survey. The dimensions of the pictures have been slightly modified to accommodate journal publication.

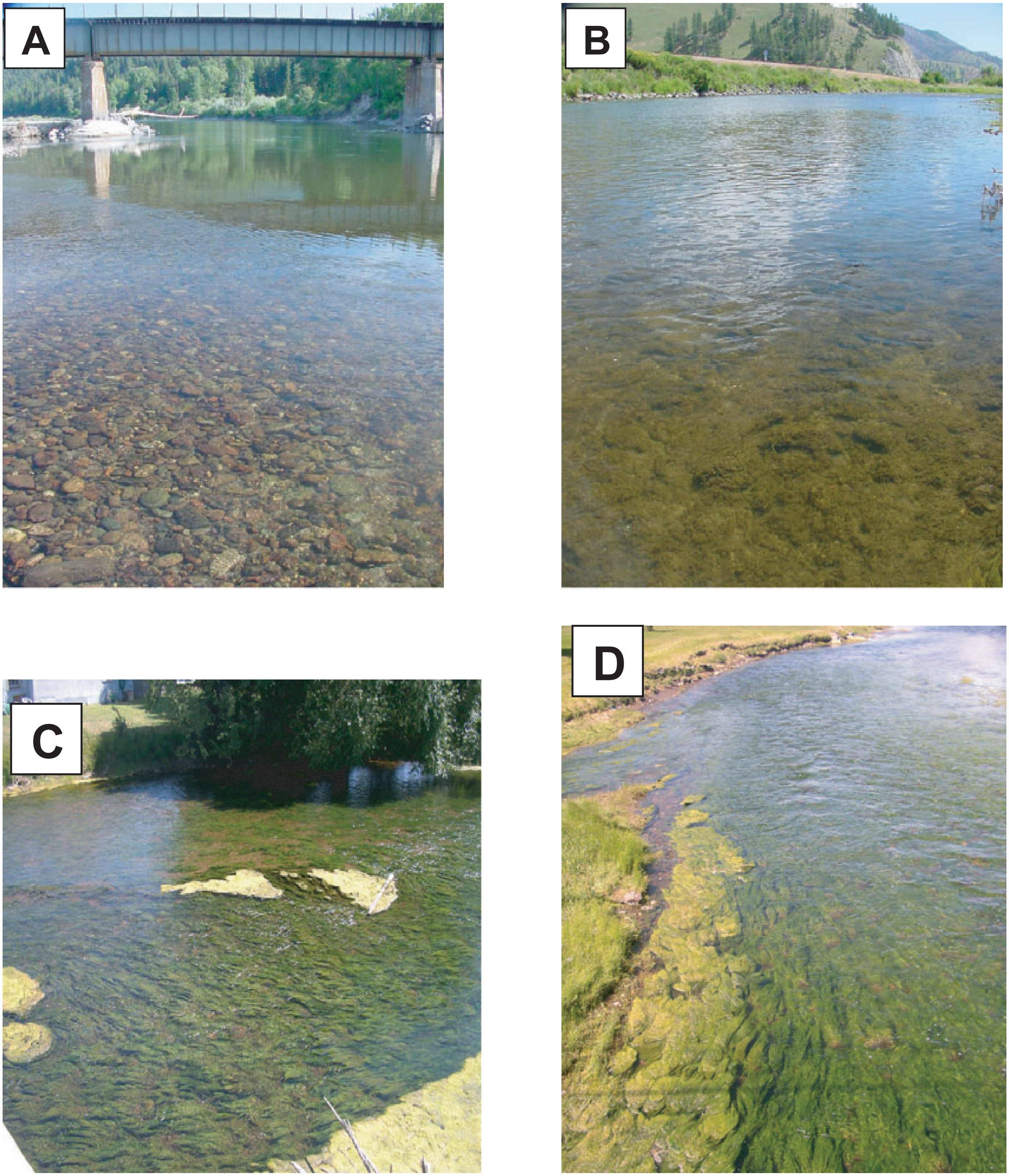
OPINION SURVEY: ALGAE LEVELS IN MONTANA RIVERS & STREAMS

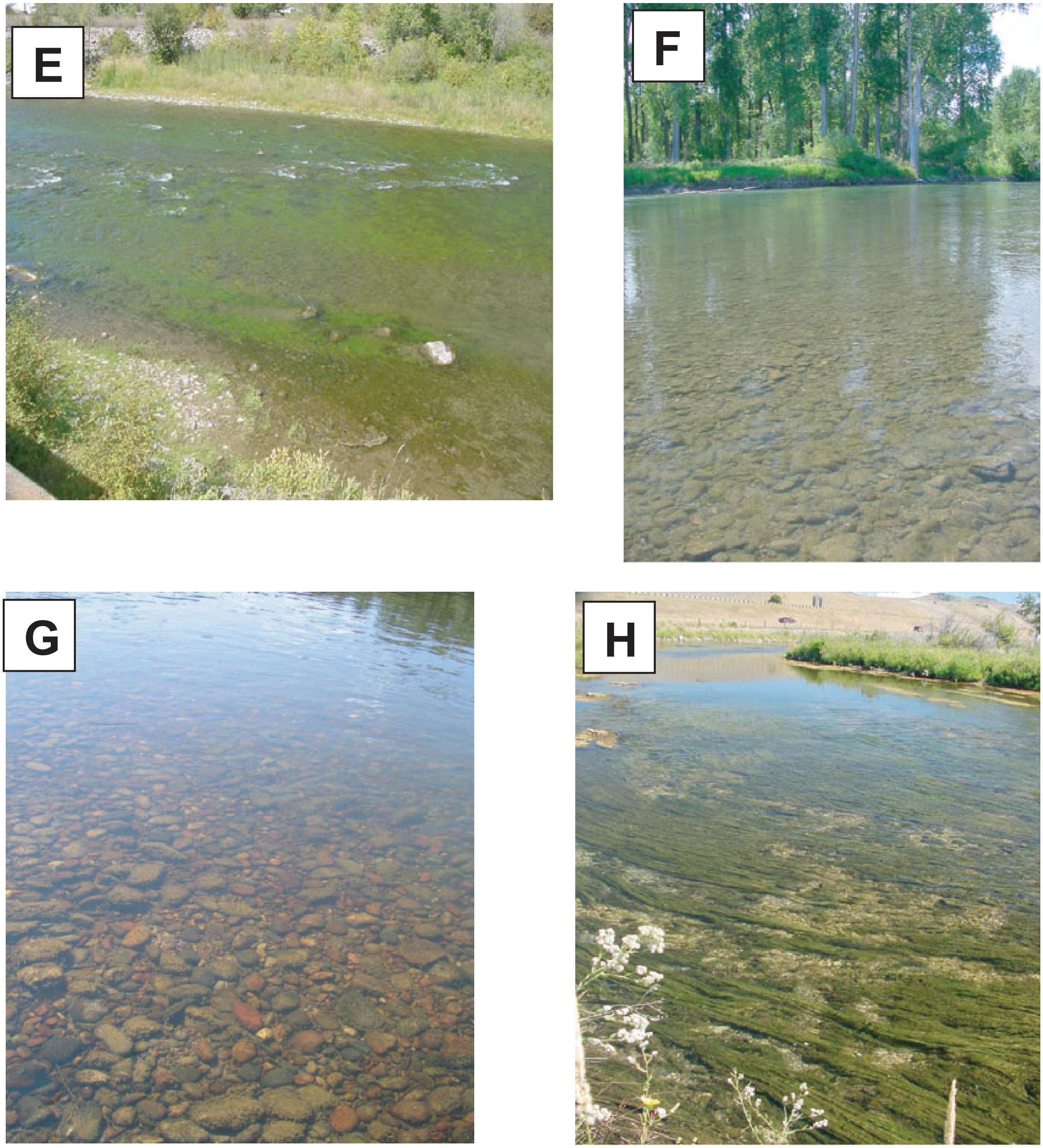
Dear Montana Citizen:
Montanans recreate in and on rivers & streams in many ways, from swimming to fishing to boating. Algae are often found in our rivers & streams, and may have the potential to affect people's recreation in different ways. The Montana Department of Environmental Quality (DEQ) would like to determine if and when river & stream algae become a nuisance to water-related recreation in Montana. The University of Montana has agreed to conduct this survey.

Inside this survey booklet are some pictures that represent different types and levels of common attached algae you might encounter in Montana rivers & streams. We would like your opinion of these pictures.

As you look over each picture, please think about whether the algae level shown would be desirable or undesirable in relation to your main recreational use of rivers & streams. Then, check the appropriate box next to each picture, and write down a few words in the space provided to tell us why. Please know that you and your responses will remain anonymous.

The DEQ will use this information to determine if and when river & stream algae become a nuisance to water-related recreation. If some levels of algae are found to be undesirable to Montana river & stream users, then the DEQ would take steps to assure that pollution sources causing those levels are properly addressed. However, we would like you to know that if a river or stream's algae levels are naturally high, the DEQ would take no action.


A
B
C
D


E
F
G
H

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Montana Department of
ENVIRONMENTAL QUALITY

MEMO

1520 E 6th Avenue
PO Box 200901
Helena MT 59620-0901
(406) 444-6697 • FAX: (406) 444-3836

To: Montana Board of Environmental Review

CC: Eric Urban, Water Quality Standards Section Supervisor

From: Michael Suplee, Ph.D., and Kyle Flynn, P.H., Environmental Science Specialists, Montana Dept. of Environmental Quality

Date: 3/19/2014

RE: Derivation of the Seasonal 14Q5 Low-flow Design Flow for Wadeable Streams and Large Rivers

When Montana Pollutant Discharge Elimination System (MPDES) permits are developed, a low-flow design flow is routinely used to calculate a permittee's allowable discharge concentrations. The Department uses a seven-day, ten-year design flow (7Q10) for permitting pollutant discharges (ARM 17.30.635(2)). But this low-flow was designed primarily for toxics and has year-round application. Existing rule directs the Department to identify a low-flow design flow specific to nutrients (ARM 17.30.635(2)). Therefore, we explored alternative design flows that might be more appropriate for discharges containing nitrogen and phosphorus (nutrients) and which could be applied seasonally to the proposed base numeric nutrient standards in MAR notice No. 17-356.

The result of our work was the seasonal 14Q5 low-flow design flow. Rules (e.g., ARM 17.30.635) have been modified to include the seasonal 14Q5 and these modifications are also found in MAR notice No. 17-356. The seasonal 14Q5 low-flow is **specific to discharges containing nutrients**. The purpose of this memo is to describe the process by which this nutrient-specific low-flow design flow was developed.

Development of the 14Q5 Low-flow Design Flow for Discharges Containing Nitrogen and Phosphorus

The most important low-flow design period for nutrients is the summer and fall baseflow period (growing season), when water quality is most likely to be impaired by excess nutrients. Streams in Montana tend to reach stable baseflow, elevated temperatures, greatest water clarity, and maximum photoperiod at about the same time, beginning in late June or early July (Suplee et al., 2007). In large rivers this period generally begins later, usually around August 1st (Flynn and Suplee, 2013). The point in time in the fall/early winter when this growing season ends is somewhat subjective, but based on rapidly declining temperatures, diminished light levels, etc., sometime in October is probably appropriate. Given these considerations, then, the growing season is the most logical time for the application of nutrient standards and a seasonal low-flow design flow.

Algal growth rates govern the time required to reach a given algal biomass. They are dependent on temperature, light, and nutrient limitation, and are the precursor to all the attendant eutrophication responses. It is therefore necessary to constrain loadings (i.e., limit nutrient concentrations) over durations when nuisance growth and associated water quality excursions are expected to (and can physically) occur. Since bottom-attached (benthic) algae have been shown to be very influential to river and stream primary productivity (Stevenson et al., 1996), benthic algal growth rates from the literature (**Table 1**) were used in conjunction with a simple analytical model to derive a suitable duration for appraisal of river and stream water-quality. The model (presented below) provides a low-flow design flow supportive of river beneficial uses.

Table 1. Enrichment Studies and Associated Net-specific Growth Rates Adjusted to 20 Degrees C. Growth rates were corrected to the reference temperature using the Arrhenius equation (Chapra et al., 2008).

Algae Type	Net Specific Growth Rate at 20˚C (k, day^{-1})	Reference	Location	Comment
Diatoms	0.50	Klarich (1977)	Yellowstone River, MT	Near Huntley Billings WWTP
Diatoms	0.55	Bothwell and Stockner (1980)	McKenzie River, OR	5% kraft mill effluent
Cladophora	0.71	Auer and Canale (1982)	Lake Huron, MI	Harbor Beach WWTP
Green algae	0.52	Horner et al. (1983)	Lab Flume	Laboratory N & P addition
Diatoms	0.42	Bothwell (1985)	Thompson River, BC	Downstream of WWTP
Diatoms	0.62	Bothwell (1988)	S. Thompson River, BC	Flume with N & P addition
Diatoms	0.58	Biggs (1990)	South Brook, New Zealand	Downstream of WWTP
Diatoms	0.45	Stevenson (1990)	Wilson Creek, KY	Agricultural stream after spate

Growth of stream and river benthic algae typically follows a general pattern of colonization, exponential growth, and autogenic sloughing and loss (Stevenson et al., 2006). The net accrual portion (i.e., colonization and growth) can be readily modeled using a first-order exponential net growth equation (**Equation 1**), with space limitation (**Equation 2**), per Chapra et al. (2010),

$$\frac{da_b}{dt} = a_b \phi_{Sb} k \tag{1}$$

$$\phi_{Sb} = 1 - \frac{a_b}{a_{b,\max}} \tag{2}$$

where a_b = benthic algal biomass (mg Chl*a*/m^2), ϕ_{Sb} = a space limitation factor (dimensionless), k = temperature dependent first-order net-specific growth rate (day^{-1}), and $a_{b,\max}$ = maximum biomass carrying capacity (mg Chl*a*/m^2). Equations 1 and 2 can be combined and solved analytically (**Equation 3**),

$$a_b(t) = \frac{a_{b,\max} \exp^{kt}}{\frac{a_{b,\max}}{a_{b,init}} + \exp^{kt} - 1} \tag{3}$$

where $a_b(t)$ = benthic algal biomass (mg Chl*a*/m^2) at a defined point in time after growth initiation, $a_{b,init}$ = initial biomass condition (mg Chl*a*/m^2), and t = time (days), so that peak biomass (PB) and time to peak biomass (T_{PB}) (per Stevenson et al., 2006) can be identified (**Figure 1**). For the design flow determination, an initial biomass of 0.1 mg Chl*a*/m^2 was assumed for all growth calculations with an $a_{b,max}$ of 1000 mg Chl*a*/m^2 (Horner et al., 1983).

We further define additional points of interest in the accrual curve (**Figure 1**), namely nuisance biomass (NB) and time to nuisance biomass (T_{NB}). These occur between the initial colonization phase and PB and reflect the point where a nuisance response would occur absent of nutrient limitation. For a nutrient control ... d so that PB never re

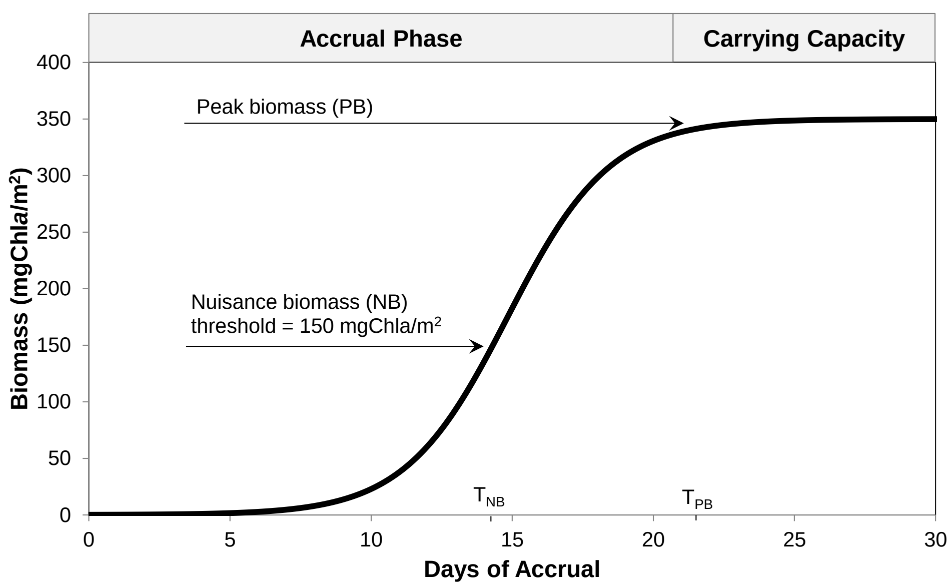

Figure 1. Modeled Accrual Phase for Benthic Algae Showing Colonization, Exponential Growth, and Peak Biomass. Time to peak biomass and nuisance biomass are shown on the abscissa.

Studies with moderate nutrient enrichment and time-variable benthic algal biomass measurements were compiled (**Table 1**) so that T_{NB}, PB, and T_{PB} could be estimated for nutrient concentrations similar to the proposed standards. We only considered studies that reported water temperature so that corrections to a standard reference temperature (20˚C) could be made, and the Arrhenius equation was used to make these adjustments (Chapra, 2008). Light was not believed to be a limiting factor in the compiled studies. Temperature-normalized growth coefficients (k; day^{-1}, $20^{o}C$) averaged 0.55 ± 0.09/day (95% confidence level) and, using **Equation 3** above, yielded times-to-nuisance biomass from 11-17 days, with an average of 14 days (**Figure 2**). Fourteen days closely matched the T_{NB} determined for the Yellowstone River (Klarich, 1977) and was selected as the duration interval. The 14-day T_{NB} applies to shallow areas (< 0.5 m) of large rivers, as well as to wadeable streams which normally have shallow depths in summer.

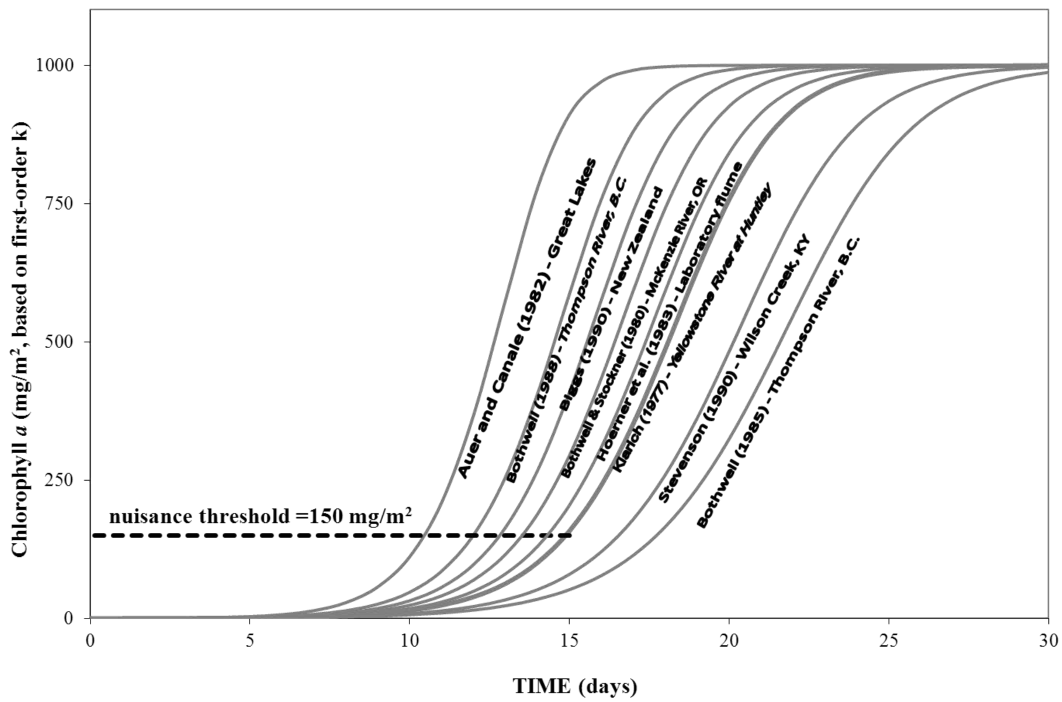

Figure 2. Estimated Time to Nuisance Algal Biomass under Moderately Enriched Conditions. Each curve was generated using the k values in Table 1. Time to nuisance biomass was approximately 14 days.

The time to nuisance biomass estimate (14 days) could actually be lower or higher than 14 days and warrants further consideration. The time to nuisance depends, in part, on the initial biomass used. It is possible that the initial biomass we used for the growth curves (0.1 mg Chl$a$/m^2; **Figure 2**) was too low, and the algae standing crop more common in summer (5-50 mg Chl$a$/m^2) would rise to a nuisance level more quickly than estimated. But if the proposed nutrient standards induce a lower level of enrichment than assumed (i.e., a reduced k, or growth coefficient), the time to nuisance biomass is extended and would lengthen the associated duration beyond 14 days. These two uncertainties will tend to counter-balance one another, and we concluded that 14 days is a good approximation of the central tendency of the modeled results.

We then considered the 14-day duration in the context of the results from a whole-stream enrichment study carried out by the Department in a Montana stream (Suplee and Sada de Suplee, 2011). In the enrichment study, peak algal biomass at the location in the study reach with the most algae—as documented by photo series and quantitative measurement—occurred about 20 days after N and P dosing began. Dosing was set at moderately-enriched levels and the algal biomass peaked at the location at 1,092 mg Chl*a*/m^2, a density nearly identical to the maximum value we assumed in **Equation 3**. (The biomass peak comprised filamentous algae, not diatoms.) Initial algal biomass in the stream at the study site was around 40 mg Chl*a*/m^2. The average stream water temperature over the time period was 21.8°C (range 16.2°C to 28.9°C), very close to the reference temperature of 20°C used in **Equation 3**. Note that the 20-day time-to-peak-biomass from the stream enrichment study closely aligns with the time-to-peak (T_{PB}) biomass in the modeled results (**Figure 2)**. Taken together, the laboratory studies (**Table 1)**, the modeled results **(Figure 2)**, and the results from the stream-enrichment study indicate that 14 days is an appropriate duration for nutrient control to maintain benthic algae below nuisance levels.

Recurrence frequency of low-flow events is the second consideration. The U.S. Environmental Protection Agency (USEPA) recommends a site-specific, biologically-driven approach for permitting discharges, where the average concentration of a toxic pollutant to which aquatic life can be chronically exposed without deleterious effects over a 4-day period should not occur more than once every 3 years (Stephan et al., 1985). Four days equates to the duration of exposure, once in three years the allowable recurrence frequency. In theory, this approach ensures excursions of toxic pollutants are uncommon enough that sufficient time passes for the aquatic community to recover in the interim years. Excess nutrient concentrations also lead to biological changes and impacts which then require time for recovery, therefore a recurrence frequency of about once every three years is probably a good starting point for nutrient pollution. Accordingly, we selected a seasonal (July 1to October 1) 14Q5 flow as the design flow for application to nutrient standards to be slightly protective. The 14-day duration reflects the time it can take to achieve nuisance biomass in wadeable streams and shallow parts of large rivers if nutrients are elevated (fewer days would be more-protective, more days less-protective). The 5-year recurrence frequency is close to USEPA's long-standing recommendation (i.e., once in three years) while being slightly protective. And, the seasonal 14Q5 flow is routinely reported by the U.S. Geological Survey (McCarthy, 2004), therefore it is readily available for use in MPDES discharge permits.

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Montana Department of
ENVIRONMENTAL QUALITY

MEMO

1520 E 6th Avenue
PO Box 200901
Helena MT 59620-0901
(406) 444-6697 • FAX: (406) 444-3836

To: Montana Board of Environmental Review

CC: Eric Urban, Water Quality Standards Section Supervisor

From: Michael Suplee, Ph.D., and Kyle Flynn, P.H., Environmental Science Specialists, Montana Dept. of Environmental Quality

Date: 3/19/2014

RE: Benthic algae biomass levels protective of fish and aquatic life in western Montana streams

In Suplee and Watson (2013), an algal biomass value of 125 mg chlorophyll *a*/m^2 is used in western Montana ecoregions to help derive the numeric nutrient criteria for those areas. Excessive bottom-attached algae in streams is a classic symptom of nutrient over-enrichment and controlling such algal growth is one of the objectives of numeric nutrient standards. This memorandum addresses how the 125 mg chlorophyll *a*/m^2 algae biomass value was determined by the Department.

Between 2009 and 2011, the Department carried out a whole-stream nitrogen and phosphorus addition study in a stream in southeastern Montana (Box Elder Creek). The project's objective was to determine what types of impacts to stream beneficial uses occur as a direct result of elevated nutrient concentrations. As documented in Suplee and Sada de Suplee (2011), bottom-attached (benthic) algae grew to much higher levels in the nutrient-dosed reaches than it did in the control reach, reaching a maximum of 127 mg chlorophyll *a*/m^2 of benthic algae[1] in the most highly dosed reach. One of the study's most notable findings was that this high biomass of benthic algae then senesced *en masse* at the end of the growing season, leading to declines in dissolved oxygen (DO) concentration to as low as 1.37 mg/L near the stream bottom. Low DO occurred in what we surmise to be a series of disconnected patches (pools or deposition zones) along the stream bottom; reaches with extremely low DO were found in areas where the stream had slower velocities. It was also documented that the only viable DO sink was the large volume of dead and decaying algae on the stream bottom (confirmed by visual observation and photographs), because water-column BOD_5 samples taken at the time were all less than detection[2]. Thus, elevated nutrient concentrations led to excessive benthic algal growth which, when it senesced and died at the end of the growing season, caused exceedences of the state's dissolved oxygen standards (DEQ, 2012).

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[1] This value represents the average for the reach. The reach average was determined from eleven replicate chlorophyll a measurements collected systematically at eleven transects spaced along the reach.

[2] Sediment oxygen demand (SOD) was not a significant DO sink in Box Elder Creek as evidenced by the near-saturation levels of water-column DO observed in the control reach throughout the study.

Dissolved oxygen standards are intended to protect fish and associated aquatic life. The finding that unusually high levels of benthic algae can lead to seasonal crashes in DO in wadeable streams is important because it demonstrates a direct link between elevated nutrient concentrations, resultant algae growth, and probable harm to fish and aquatic life that is disconnected in time with the initial nutrient loadings. But because the study was undertaken in southeastern Montana, it was necessary to carry out additional analyses to see if its findings were applicable to western Montana streams, which are generally colder and have steeper gradients.

To explore the effect of water temperature on the DO impacts observed in the eastern Montana study, we developed a modified Streeter-Phelps (1925) analytical model of the Box Elder Creek nutrient dosing reach. The physical basis of the model was then used to evaluate the response of hypothetical streams in western Montana at an elevation of 1,219 m (4,000 ft), with mean daily water temperature of 7 ^{o}C (a common temperature in the region), and with varying physical characteristics (gradient and velocities, reflecting different reaeration behavior). Methods, discussion and recommendations are provided below.

1.0 Simulation of Nutrient Dosing Study Findings for Cooler Water Temperatures, Higher Dissolved Oxygen Saturation, and Differing Reaeration Rates

As documented in Suplee and Sada de Suplee (2011), benthic algae senesced *en masse* at the end of the growing season which led to observed DO levels as low as 1.37 mg/L near the stream bottom in the High Dose reach (HD reach) in a depositional area (glide) with steady laminar flow. It was also documented that the only viable DO sink was the large volume of dead and decaying algae on the stream bottom. Thus, senesced algae are a significant and important DO sink when it comes to eutrophication, and we refer to it here as “senesced algae oxygen demand” (SAOD). The terminology has been used to differentiate it from normal sediment oxygen demand (SOD) which is associated with the oxygen consuming properties of organic material in a stream’s bottom sediments.

The questions addressed in this section are:

1. *If the nutrient dosing study had been carried out in an identical western Montana stream but with cooler average temperatures and higher dissolved oxygen saturation, would impacts to Montana’s dissolved oxygen standards still have occurred?*

2. *In the hypothetical stream with cooler water temperatures and higher dissolved oxygen saturation simulated in No. 1 above, what would be the effect of differing reaeration rates on the dissolved oxygen concentrations?*

Continuously monitored data (by YSI 6600 sonde) at the headwaters of the Box Elder High Dose (HD) reach (triangle **A** in **Figure 1-1**, just upstream of the nutrient addition point) showed that DO was very close to saturation entering the HD reach (measured as 8.8 mg DO/L @ 12 cm off the bottom; water temp = 15.3^{o}C; elevation 921 m). By the time water flowed down to the HD reach YSI sonde, DO concentration 12 cm off the bottom had dropped to 1.37 mg/L. The residence time between the two points was about 20 min as identified through velocity measurements. Box Elder Creek is a Rosgen C4 channel (Rosgen, 1996).

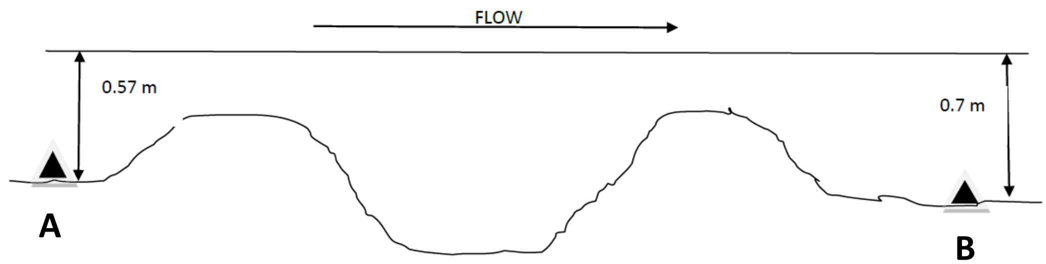

Figure 1-1. Longitudinal diagram of Box Elder Creek high-dose reach. The upstream YSI sonde (**A**, on left) was placed just upstream of the point where nutrients were added. There is a span of 110 m between it and the High Dose YSI sonde (**B**, on right). Two riffles and one pool were found in the reach between the two sondes.

Data indicate that it is unlikely that 1.37 mg DO/L persisted surface to bottom; rather, a vertical gradient in oxygen concentration probably occurred. Such a gradient is typical of steady-state conditions of diffusive mass transfer typified by Fick's first law. Given the mass of decaying algae observed, we inferred that DO was zero mg/L on the bottom of the channel whereas, at the surface, it was still probably at or near saturation (as observed upstream). A simple linear relationship was fit between the three DO-depth points (**Figure 1-2**; representing the linear change in oxygen over depth, i.e., dO_2/dz), and we believe a linear fit is very reasonable. Indeed, if poorly mixed (as typified in diffusion problems) a linear gradient would occur over the water column. In this instance, the linear model most reasonably fits the data.

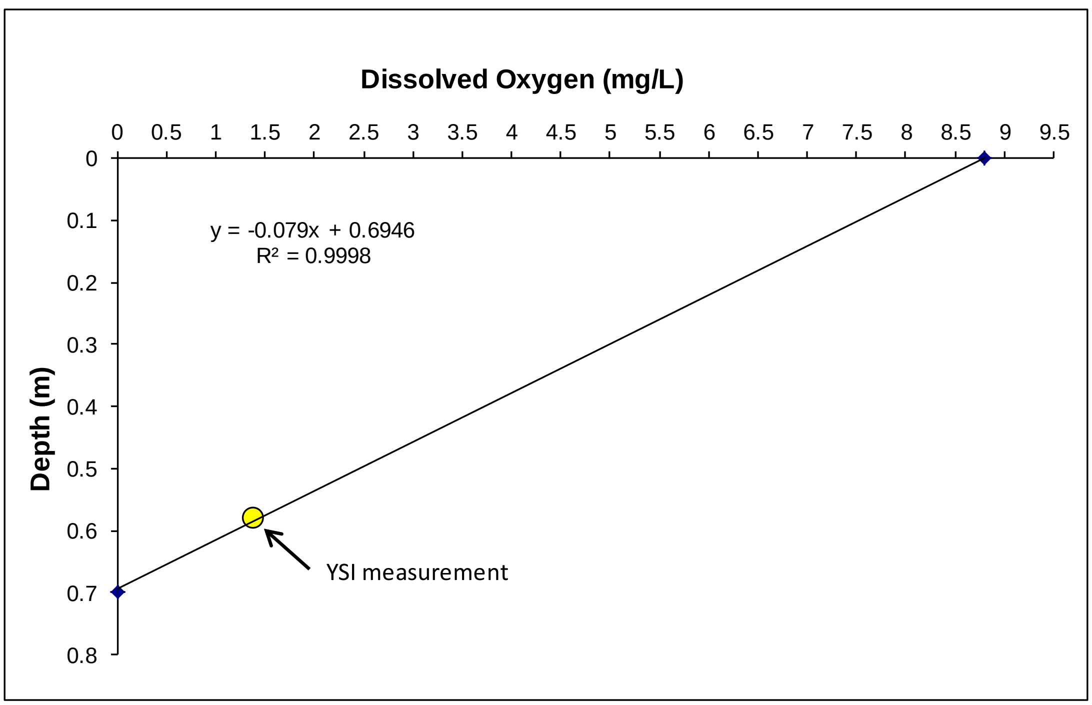

Figure 1-2. Estimated surface to bottom DO profile at the HD reach, 12:01 am, 10/6/2010.

Thus we have some knowledge about the oxygen gradient through the water column as well as the average oxygen concentration at the site in **Figure 1-2** given the previous assumptions. When using the linear assumption, the midpoint between 0 (bottom) and 8.8 (surface) mg DO/L is effectively the mean DO concentration of the site (i.e., 4.4 mg/L). An analytical solution to the Streeter-Phelps model with constant SAOD was then developed. With this new model we could evaluate whether differences in reaeration rates and algae growth rates between eastern and western Montana streams preclude (or do not preclude) the use of an algae level of 125 mg chlorophyll *a*/m^2 to assess aquatic life impacts in western Montana streams. The algae level was slightly lowered, from 127 to 125 mg chlorophyll *a*/m^2, to provide a threshold that is somewhat more protective; after all, we documented DO impacts at 127. Assuming plug flow (i.e., where advection is dominant) and a channel with uniform slope and cross-sectional area, the 1-D mass balance for DO over a differential element (Δ x) is

$$\Delta V \frac{DO}{dt} = J_{in}A_c - J_{out}A_c \pm \text{reaction}$$

where ΔV = volume of element (m^3), A_c= channel area (m^2), DO = dissolved oxygen concentration (g/m^3), and J_{in} and J_{out} = flux of DO in and out of element due to advection (g/m^2 d). Flux in and out of the element is defined as

$$J_{in} = U(DO)$$

$$J_{out} = U(DO + \frac{dDO}{dx}\Delta x)$$

where U =channel velocity (m/d) and Δx = incremental distance (m).

By adding first-order reaction rate (i.e., reaeration) and a zero-order term for senesced algae oxygen demand (SAOD [$gO_2/m^2/d$]), the equation looks like

$$\Delta V \frac{DO}{dt} = U(DO)A_c - U(DO + \frac{\partial DO}{\partial x}\Delta x)A_c + k_a\overline{(DO_{sat}\text{-}DO)}\Delta V - \frac{SAOD}{H}\Delta V$$

where k_a = first-order rearation coefficient (/d), DO_{sat}-DO = the average DO deficit, and H = channel depth (m).

Collecting terms, dividing by $\Delta V = A_c \Delta x$, and taking the limit as $\Delta x \rightarrow 0$ yields

$$\frac{DO}{dt} = -U\frac{\partial DO}{\partial x} + k_a\overline{(DO_{sat}\text{-}DO)} - \frac{SAOD}{H}$$

At this point, it should be noted that the first-order reaeration rate k_a is actually a function of the liquid mass transfer velocity [k_l, m/d] divided by H, which relates atmospheric flux to surface area. Also, because of the high Henry's constant of O_2, and the fact that oxygen in the atmosphere is constant, exchange is strongly liquid-film controlled. Consequently, DO at saturation (DO_{sat}) can be well characterized by altitude and temperature only. Finally, to simplify the differential equation, we reformulate DO concentration as dissolved oxygen deficit $D = DO_{sat}\text{-}DO$ [mg O_2/L], which switches the sign of the DO input/output terms. Under steady state conditions the temporal derivative goes away and we are left with

$$0 = -U\frac{dD}{dx} - k_a D + \frac{SAOD}{H}$$

The above differential equation can then be solved by using integrating factors and yields the equation below, where D_0=the initial *DO* deficit (mgO_2/L) and x equals the distance downstream (m) from initial conditions:

$$D = e^{-\frac{k_a}{U}x} \left(D_0 + \frac{SAOD}{Hk_a}\left(e^{\frac{k_a}{U}x} - 1\right)\right)$$

Finally, substitution of k_a with the approximation from Owens et al. (1964) makes the equation appropriate for small streams (Covar, 1976) (in metric units, where *U* is in m/s):

$$k_a = 5.32\frac{U^{0.67}}{H^{1.85}}$$

Thus the equation is applicable to small shallow streams where the oxygen generation and consumption processes are primarily reaeration and SAOD. It should be noted that reaeration is temperature adjusted using the Arrhenius equation with a theta (θ) of 1.024 (Chapra, 1997). Also, since we have omitted respiration and photosynthesis from our equation, the results are probably only appropriate for night-time conditions only. Finally, note that algal respiration does consume oxygen in dark reactions through carbon oxidation, but it was not included in the model.

Following model development, we then calibrated the analytical model to data from Box Elder Creek to estimate SAOD expected from an accumulation of dead/decaying algae, which could then be transferred to other streams. In this instance the calibrated SAOD was very high, approximately 92 $gO_2/m^2/day$. Using the Arrhenius equation and a theta (θ) of 1.047 (applicable to BOD decomposition; Chapra, 1997), this zero-order rate equates to 76 $gO_2/m^2/day$ at a stream temperature of 7°C. In other words, the biological decomposition rate of the dead algae determined from the Box Elder Creek dosing study has been reduced mathematically to a colder water temperature. We refer to this new, adjusted SAOD as $SAOD_{COLD}$.

Figure 1-3 below compares the model output for Box Elder Creek vs. the simulated, carbon-copy western Montana stream evaluated through the model. Conditions were kept the same except that the western-Montana simulation was colder (7° C, vs. 15.3° C in Box Elder Cr.) and at a higher elevation (1,219 m vs. 921 m at Box Elder Cr.).[3] In general, we see a longitudinal decline in DO concentration

[3] Velocity in both streams was set at 0.09 m/s, depth at 0.27 m, as measured in Box elder Cr. We also used measured DO (8.8 mg/L) for initial conditions in Box Elder Cr. as observed just upstream of the nutrient-dosed reach (at the sonde shown as A in Figure 1-1); DO at saturation in Box Elder Cr., at 15.3°C and the site elevation, is very close (8.97 mg/L). In the western-MT simulation, DO was set at 10.4 mg/L (saturation at the temp. and elevation given). The ka values for Box Elder Creek and its western-MT simulation were 11.1/day and 9.1/day, respectively.

reflective of a waterbody flowing over a very large diffuse SAOD source. The zero-order SAOD used for the western MT stream is $SAOD_{COLD}$. The model output presents average water column DO (i.e., the midpoint of the surface-to-bottom DO gradient in **Figure 1-2**) longitudinally, and shows how that average would decline over space due to SAOD. Note that the DO standard is actually exceeded sooner in the western Montana simulation (occurring 75 m downstream as opposed to 100 m in Box Elder Creek). This happens because the western Montana DO standard is set at a higher concentration (8 mg DO/L vs. 5 mg DO/L in much of eastern Montana) because it is intended to protect salmonid fishes.

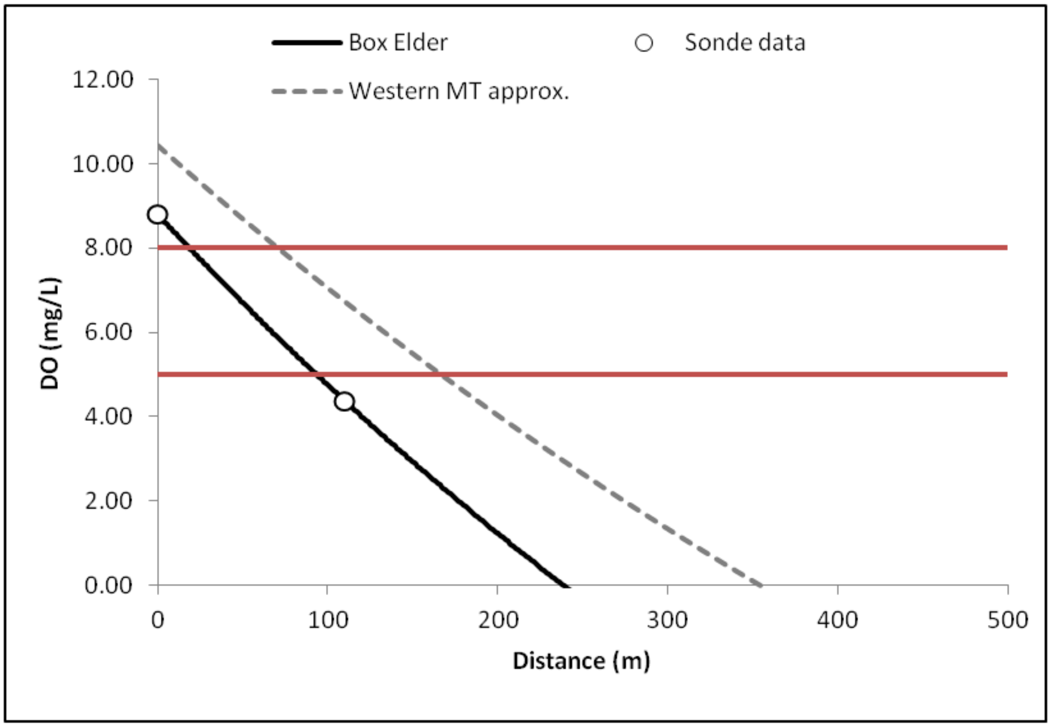

Figure 1-3. Model output for two scenarios: (1) Box Elder Creek nutrient dosing study, including YSI sonde data from the HD reach which were used to calibrate the model, and (2) the simulation of an identical stream but at 7° C and at 1,219 m elevation. Red horizontal lines show the DO standards typical for each region (lower red line for eastern MT, upper red line for western MT).

1.1 Simulations Using Different Reaeration Rates

To evaluate the potential effect of the calibrated SAOD presented above, we evaluated a number of stream types commonly encountered in western Montana. The primary difference between the evaluated streams and the Box Elder Creek calibration was the dependence of reaeration rate on channel configuration, and (as before) the effect of colder temperatures and higher altitude on oxygen saturation and biological decomposition rates. Western Montana streams were segregated using the Rosgen stream classification system which integrates factors such as slope, channel width/depth ratio, substrate, etc. Given the dependence of the reaeration coefficient on channel depth and velocity, Manning's equation was used to determine effective velocities for various channel configurations:

$$U = \frac{1}{n} R^{2/3} S^{1/2}$$

where U = velocity in m/sec, R = hydraulic radius in m, S = water surface slope in m/m (Dunne and Leopold, 1978, but in SI units), and n = Manning's coefficient. Rosgen (1996) provides descriptive statistics for many of his stream types (e.g., C4, F4 channels), and values (e.g., surface water slope, cross-sectional area) representing the central tendency of each group were selected and input into Manning's relationship in order to derive a representative velocity for the stream class group (**Table 1-1**). Roughness coefficients between 0.05 and 0.06 were selected which, from our experience, are reflective of streams during low-flow conditions (recall that the roughness coefficient actually is not independent of flow and depth). Chow (1959) reports variation in Manning's *n* with stage and also that weeds (i.e., macrophytes and attached algae) in stream channels induce somewhat higher Manning's *n* values. Given that our simulated streams would have fairly thick mats of filamentous algae, slightly higher-than-textbook Manning's *n* values are well justified[4].

Table 1-1. Stream Channel Characteristics for Different Representative Rosgen Stream Channels Used in the Model.

Rosgen Stream Type	X-sectional area (m^3)	Width (m)	Depth (m)	Surface Water Slope (m/m)	Manning's *n* used	Velocity (m/sec)
B4	1.54	5.77	0.27	0.0200	0.05	1.11
C3	3.09	11.55	0.27	0.0023	0.06	0.32
C4	2.70	10.09	0.27	0.0045	0.06	0.45
C5	2.51	9.39	0.27	0.0005	0.06	0.14
E3	0.88	1.16	0.76	0.0100	0.06	0.80
E4	0.54	0.71	0.76	0.0100	0.06	0.65
E5	0.56	0.73	0.76	0.0010	0.06	0.21
F4	1.95	7.31	0.27	0.0018	0.06	0.28
G4	0.74	2.78	0.27	0.0200	0.06	0.87

The calculated velocities and depths for the stream groups (**Table 1-1**) were used to recalculate the reaeration coefficient in the model and evaluate the effects on longitudinal DO decline. As before, the simulations were run under the assumption that the stream, irrespective of whether it was C4, B4, etc., was at 7° C and at an elevation of 1,219 m. We tested Rosgen B, C, E, F, and G stream types. Following initial testing, it was clear from the B-channel results that the higher gradient A-channels did not need to be evaluated since their gradients/velocities would overcome SAOD. We did not test D channels as Rosgen (1996) provides no descriptive statistics. It was found that C and F channels would be vulnerable to low oxygen problems (i.e., their velocities were insufficient to overcome $SAOD_{COLD}$), the Rosgen C4 channel somewhat less so than C3 or C5. In contrast, A, B, E, and G channels would probably not be impacted by low DO. It is not clear whether D channels would be vulnerable to DO problems or not; some of the lower gradient ones very likely would. **Figure 1-4** below contrasts results from three examples; a hypothetical B4 channel, C3 channel, and F4 channel. As can be seen, high velocities in the B4 channel lead to higher reaeration rates capable of overcoming the $SAOD_{COLD}$; in contrast, the C3 and

[4] Box Elder Creek, which is a Rosgen C4 channel with a D_{50} of 45 mm (coarse gravel), required an even higher Manning's *n* (0.08) in order to match the Manning's equation result to measured average stream velocity and slopes. Thus, the use of 0.06 in our calculations for Rosgen stream types is quite reasonable.

F4 channel become impacted (relative to their DO standards) in about the same way Box Elder Creek was.

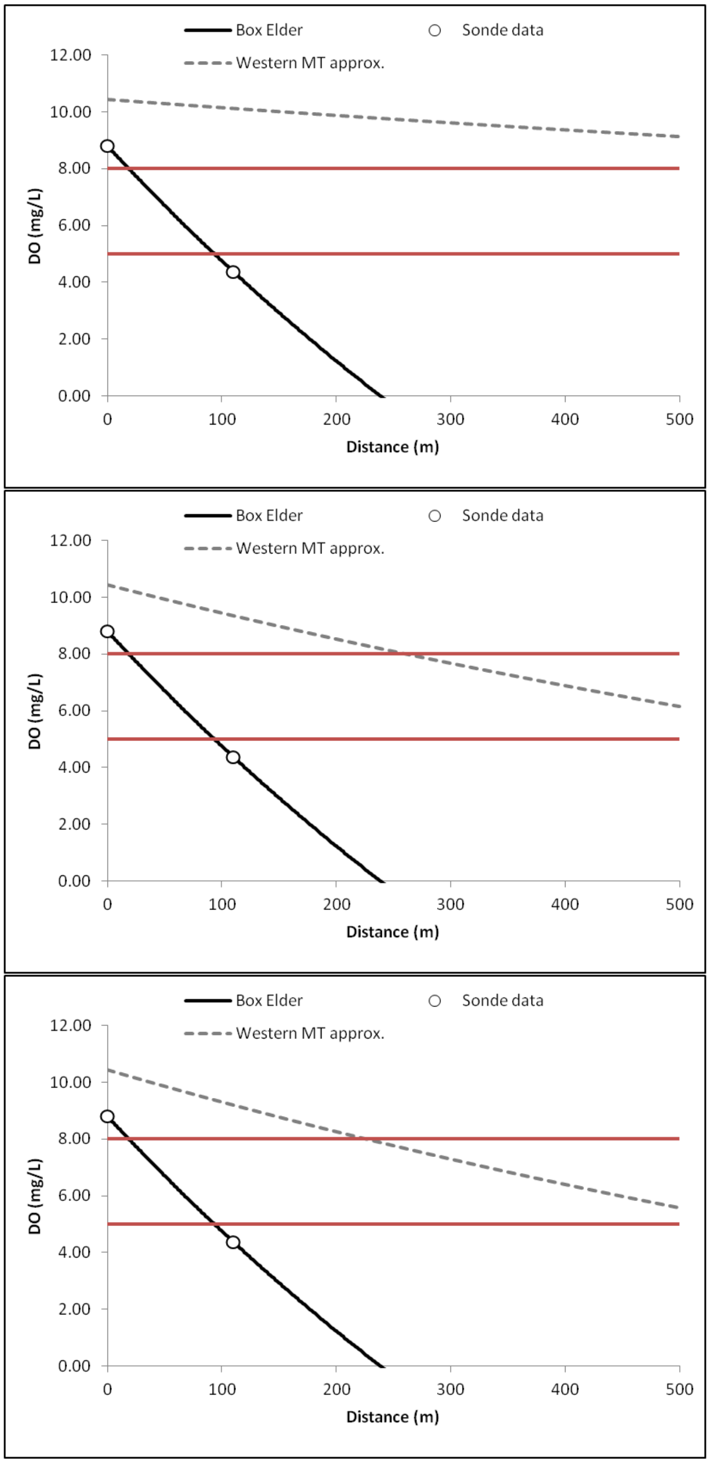

Figure 1-4. Modeled DO impacts as a function of Rosgen stream class. Upper Panel. Rosgen B4 channel. Dissolved oxygen would not fall below the 8.0 mg DO/L standard for a long distance and DO impacts are almost certainly negated. Middle Panel. Rosgen C3 channel. Violations of the DO standard occur within 300 m. Lower panel. Rosgen F4 channel. Dissolved oxygen impacts are similar to the C3 channel, but occur in an even shorter distance.

1.2 Discussion

Others discuss end-of-growing season senescence of plants and its effect on stream water quality. Jewell (1971) notes in streams in England that “At the end of the growing season, or when the weeds are killed, their decomposition may exert heavy demands on the oxygen resources of a water”. Novotny and Bendoricchio (1989) observe that “oxygen deficiency is highest and most troublesome in streams where shallow productive zones are followed be deeper sections”. The latter statement largely conforms to what we observed, where senesced algae accumulated in the glides and pools of the nutrient-dosed Box Elder Creek reach, and these areas manifested discontinuous areas of low DO longitudinally along the reach.

The reach-average level of benthic algae leading to the DO problem at the Box Elder Creek HD reach (127 mg Chl*a*/m^2) has been observed in eutrophied streams in western Montana, and quite often *Cladophora* provides a substantial proportion of this biomass (just as was observed in the Box Elder study). Thus, a key biological characteristic of the plains stream we studied, in terms of the biomass and algae type, is quite comparable to western Montana streams. Therefore, we believe it can be reasonably concluded that if the study had been carried out in an identical stream in western Montana, but at 7^o C and with DO at saturation of 10.4 mg/L, one would have seen exceedences of the DO standards following a similar pattern.

The SAOD we calculated is far higher than sediment oxygen demand (SOD) reported in the literature (highest SOD located was 21.4 g O_2/m^2/day; Ling et al., 2009). But as mentioned at the start, SAOD is not SOD in the normal sense and the rates are not strictly comparable. Novotny and Olem (1994) report that feedlot runoff (a highly organic, putrescible material) can have BOD_5 of 1,000-12,000 mg/L. If this material were all to settle to the stream bottom in a stream having the same average depth as Box Elder Creek and then exert its DO demand, BOD_5 values of this magnitude would equate to 53-640 g O_2/m^2/day (at 20^o C). The SAOD we calculated for decomposing benthic algae (equal to 133 g O_2/m^2/day @ 20^o C) clearly falls to the low side of this range and is, therefore, a reasonable estimate. It should also be noted that the true SAOD may be higher, but much more localized spatially along the reach (i.e., we calibrated it assuming SAOD over the entirety of the reach).

In conclusion, it appears that the Box Elder dosing study, had it been carried out under identical circumstances except for lower water temperature and higher DO saturation, would have led us to similar conclusions about the impacts of senesced algae oxygen demand and associated effects on stream DO dynamics (**Figures 1-2, 1-3**). However, our simulations indicate that the gradients of some western Montana streams are sufficiently high that SAOD can, for all practical purposes, be overcome (**Figure 1-4**, upper panel). Based on our findings, we recommended the following:

1. The 125 mg Chla/m^2 threshold should apply to all Rosgen C and F channels, as their group characteristics (velocity, depth, etc.) appear to be insufficient to overcome DO impacts from senesced algae oxygen demand;
2. The 125 mg Chla/m^2 threshold would not apply to Rosgen A, B, E and G channels, as their group characteristics (velocity, depth, etc.) appear to be sufficient to overcome DO impacts from senesced algae oxygen demand; and

3. No recommendation is made for Rosgen D channels at this time. These channels are not so commonly encountered in Montana and cases will need to be evaluated case-by-case.

Note that in streams where the 125 mg Chl*a*/m^2 threshold does not apply, the recreationally-derived benthic algal threshold (150 mg Chl*a*/m^2) does. Because 125 mg Chl*a*/m^2 links directly to DO impacts, it is associated with the fish and associated aquatic life beneficial use; in contrast, 150 mg Chl*a*/m^2 applies to the recreation use (per Suplee et al., 2009).

2.0 Final Department-recommended Benthic Algae Threshold

A meeting was held between management and technical staff in March 2012 to consider these findings and their implication for making stream assessment decisions. Careful consideration was given to the totality of information provided by the Box Elder nutrient dosing study, the modeling results discussed above, and the benthic algae threshold for protecting the recreation use (150 mg Chl*a*/m^2). Our analysis showed that some western Montana streams would be vulnerable to low DO problems if benthic algae reached 127 mg Chl*a*/m^2, but other streams would not. There was relatively little difference in magnitude between the level of algae that would prevent impacts to fish and aquatic life (~125 mg Chl*a*/m^2) vs. the level which protects recreational (150 mg Chl*a*/m^2). Further, monitoring staff who have completed many stream assessments involving benthic algae have indicated that in the vast majority of cases benthic algae levels are either well below or well above the thresholds in question (i.e., borderline cases near the thresholds that would a require more detailed analysis and data collection are uncommon). Thus, in order to preclude a complex, two-threshold system requiring Rosgen stream class identification in all cases, the Department instead decided that in western Montana streams a single benthic Chl*a* threshold of 125 mg Chl*a*/m^2 (site average) would be used. Values above this threshold are considered impacts to both the aquatic life and the recreational beneficial uses, as documented in Suplee and Sada de Suplee (2011). And as noted at the start of this memorandum, 125 mg Chl*a*/m^2 has since been used to help derive the proposed base numeric nutrient standards for western Montana ecoregions (Suplee and Watson, 2013; draft Department Circular DEQ-12A).

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DEFINING LARGE RIVERS IN MONTANA USING A WADEABILITY INDEX

Prepared by Kyle Flynn, P.H.[1], and Michael Suplee, Ph.D.[2]

EXECUTIVE SUMMARY

A simple, single parameter and physically-based approach for determining river wadeability is proposed by the Montana Department of Environmental Quality (DEQ) to differentiate between wadeable and non-wadeable rivers for the purpose of watershed management. Data from 54 different rivers and 157 sites were compiled to identify key attributes of wadeability including baseflow annual discharge, mean baseflow hydraulic depth and velocity, and the product of the two, herein referred to as the wadeability index. Very consistent relationships between the wadeability index and baseflow annual discharge (r^2=0.91) were found, indicating that hydraulic geometry, and its independent variable discharge, are suitable approximations for assessing wadeability. Data analysis also revealed that a statistically significant changepoint occurs within the wadeability function ($p << 0.001$) around which a wadeable/non-wadeable threshold can be determined at the 90% confidence interval. This threshold correlates to an approximate baseflow annual discharge of 1,500 cfs, depth of 3.15 ft, or wadeability index of 7.24 ft^2/s, and compares well with other wadeability indices proposed in the literature. Using the above criteria, eight rivers in the state were determined to be non-wadeable in at least one portion of their extent. They are: the Bighorn, Clark Fork, Flathead, Kootenai, Madison, Missouri, South Fork of the Flathead, and Yellowstone rivers. As a result, a demarcation upon which future monitoring, modeling, and assessment methodologies for large rivers in Montana is now established.

[1] Lead Hydrologist, Information Management and Technical Services Section
[2] Environmental Science Specialist, Water Quality Standards Section

INTRODUCTION

The concept of river wadeability is a relatively important component in the field of river management. This is largely due to implications regarding the design and selection of monitoring procedures or equipment, and the importance of how water-quality management endpoints are formulated. Several attempts have been made to define wadeability, of which Wilhelm et al. (2005) and Flotemersch et al. (2006) provide very good reviews. Attributes commonly used to define wadeability include stream order (Strahler, 1957), drainage area, or site specific characteristics such as depth or width. State-level definitions are as follows: in Michigan, a non-wadeable or large river is defined as a reach which cannot be waded across its length or from bank to bank (Wilhelm et al., 2005); in Wisconsin, non-wadeability constitutes a river with more than three kilometers of continuous channel that is too deep to sample by wading during summer baseflow (Lyons et al., 2001); in Ohio, a large river is one that exceeds a drainage area of 1600 km^2 (Ohio EPA, 1989); and, in Idaho, average width at baseflow, average depth at baseflow, average greatest depth, site discharge, mean annual site discharge, and site drainage area have all been considered (Grafe, 2002). Average depth of one meter (Stalnaker et al., 1989), width of 50 meters (Simonson et al. 1994), or a river order of six or greater (Vannote et al., 1980; Sheehan and Rasmussen 1999) have also been suggested.

Clearly, the available literature suggests that a foundation for determining wadeability already exists. However, most of the methods detailed above require that the assessor is physically at the river, or alternatively, are unreliable predictors of wadeability (Flotemersch et al., 2006). Due to these reasons, DEQ wishes to devise a simple, single parameter index that defines wadeability up-front, using readily available data. Therefore we have initiated our own investigation to define appropriate wadeability measures for Montana rivers.

BACKGROUND

The demarcation between wadeable and non-wadeable rivers is ambiguous and indistinct (Wilhelm et al., 2005). As a result, DEQ simply defines a large river as *one that is unwadeable during the summer and early fall baseflow period*. Techniques to distinguish between wadeable/non-wadeable thresholds, as well as what constitutes the baseflow period, are described below.

The Wadeability Index (WI)

In its simplest form, wadeability is dependent on river flow and channel hydraulic geometry. Resultant force is exerted on a person by oncoming flow which must be overcome by the wader to avoid toppling or instability. Abt et al. (1989) conducted tests on human wading subjects in a re-circulating flume, and defined a measure of human stability under different flow conditions as the product number (P.N.). Defined as the multiplicand of the water depth and velocity, P.N. is a direct measure of wadeability, and therefore has been re-coined the wadeability index (WI) for our purposes. WI is calculated as follows (Equation 1) where, d = mean depth (ft) and v = mean velocity (ft/s):

$$WI = d \times v \ \text{ft}^2/\text{s} \qquad \text{(Equation 1)}$$

Results from Abt et al., (1989) indicate that WI values ranging from 7-20 ft^2/s cause instability in human subjects (i.e., the inability to stay upright), and the relative ability of an individual to avoid toppling depends on their height, weight, and compensation skills. A lower limit for stability was suggested at approximately 7 ft^2/s. Consequently, this WI threshold is a logical starting point for partitioning between wadeable and non-wadeable rivers in Montana using safety, and the ability of an assessor to stay upright when wading as the sole measure. Ideally, if the variables described above (i.e. depth and velocity) could be easily ascertained for a site, wadeability could readily be determined.

Existing Measures of Wadeability

For all practical purposes, the only extant measures of river wadeability are made by the U.S. Geological Survey (USGS) as part of their field measurement program. Observations are made for purpose of determining discharge as part of routine gaging activities, and mean depth (*d*), as identified as the cross-sectional area of the measurement divided by the measurement width, and mean velocity (*v*), the quotient of the discharge divided by the cross-sectional area, are all reported. Leopold and Maddock (1953) effectively demonstrate how these observations can be used to approximate the hydraulic geometry of natural river channels using simple power functions (Equations 2 and 3), thereby forming the initial basis of our supposition:

$$d = aQ^b \qquad \text{(Equation 2)}$$

$$v = cQ^d \qquad \text{(Equation 3)}$$

where, Q = discharge (ft^3/s), a, b = experimentally determined coefficient and exponent related to depth, and c, d = experimentally determined coefficient and exponent related to velocity. The utility of these relationships lies in the fact that for any given discharge, the channel hydraulic geometry, and subsequently WI can be determined in a predictable way.

METHODS AND MATERIALS

Data Compilation

All USGS gage sites within the state of Montana having the nomenclature “river”, as well as a handful of selected locations outside the state were compiled by DEQ from the USGS National Water Information System (NWIS, 2010). This was done for the purpose of developing site coefficients and exponents for Equation 2 and 3, and included data from 54 different rivers and 157 gaging sites (Figure 1). For each site, the information necessary to determine wadeability index, i.e. mean cross-sectional velocity, discharge, and top width were acquired over the period of record for the gage. Only sites with 10 field observations or more, and at least 10 years of published streamflow statistics (McCarthy, 2004), or provisional streamflow statistic data from NWIS, were considered. Quality control (QC) for each site was completed through review of the constructed depth and velocity rating curves. Errant values were corrected if identifiable typos

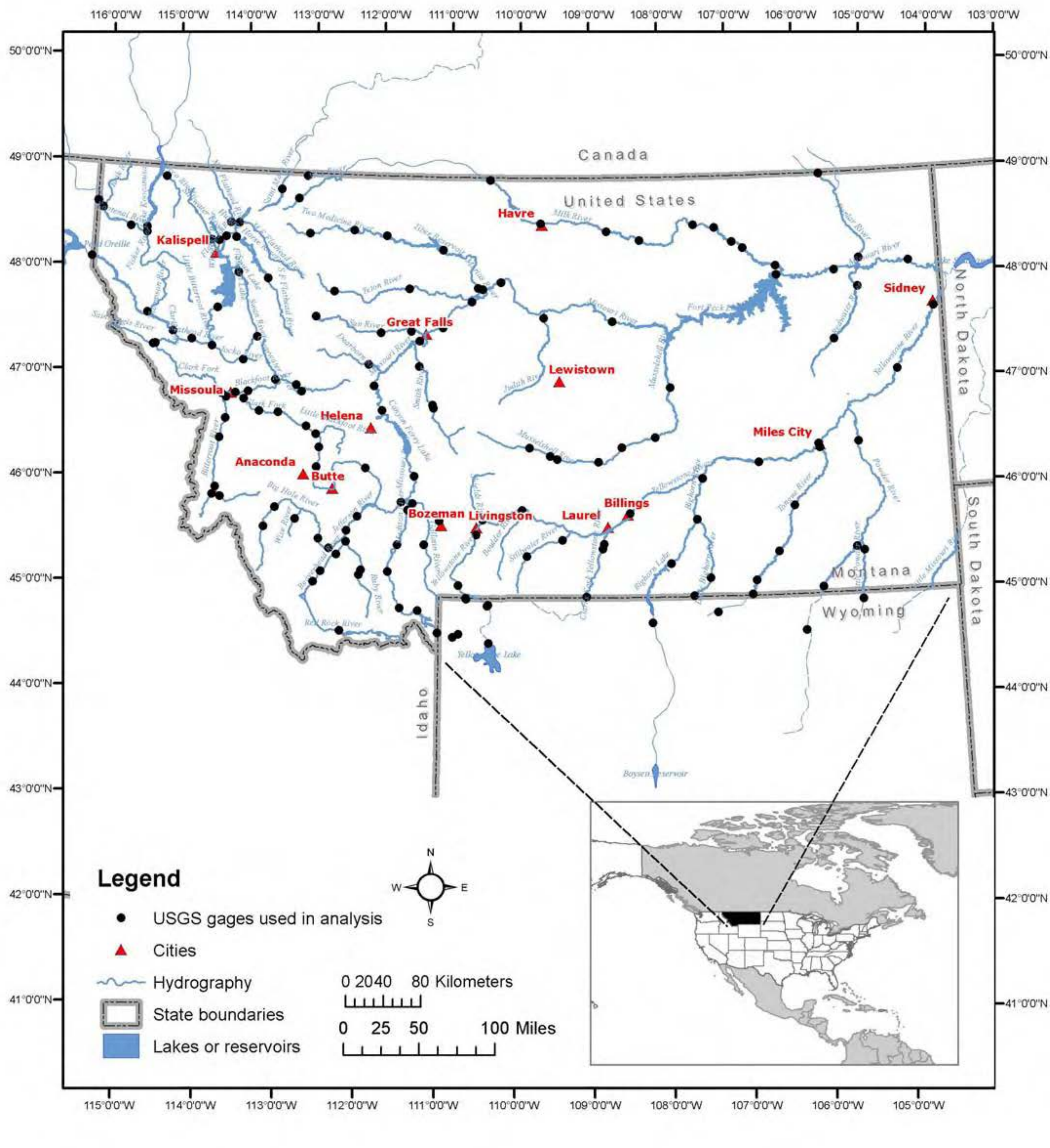

Figure 1. Locations of USGS gage sites used in compilation of Montana's wadeability index.

were found, or were removed in instances where the error or source of the error could not be identified. All field measurements were included in the data compilation regardless of channel control condition or measurement type, and no efforts were made to remove high flow data unless it was clearly apparent that a shift in the rating curve had occurred. While it is possible that some error is introduced into our analysis due to these conditions, the number of sites used in the analysis likely overrides any effects from a handful of anomalous stations.

Relationship between Discharge and Hydraulic Geometry

The relationship between discharge and hydraulic geometry at each gage was determined using the least squares method within Microsoft Excel®. Depth and velocity were evaluated against discharge, and yielded very good site specific regressions with individual coefficients of determination (r^2) ranging from 0.35-0.99 and 0.15-0.99 for depth and velocity, with an average r^2 for all sites of 0.82 and 0.81, respectively. Figure 2 shows an example of the rating curve for USGS 12363000, Flathead River at Columbia Falls MT (top) as well as a combined rating curve for all 157 sites and 54 rivers (bottom). The coefficients and exponents of most sites are very similar to those reported by Leopold and Maddock (1953). The number of observations at each gage site, and r^2 for each rating curve, are shown in Appendix A.

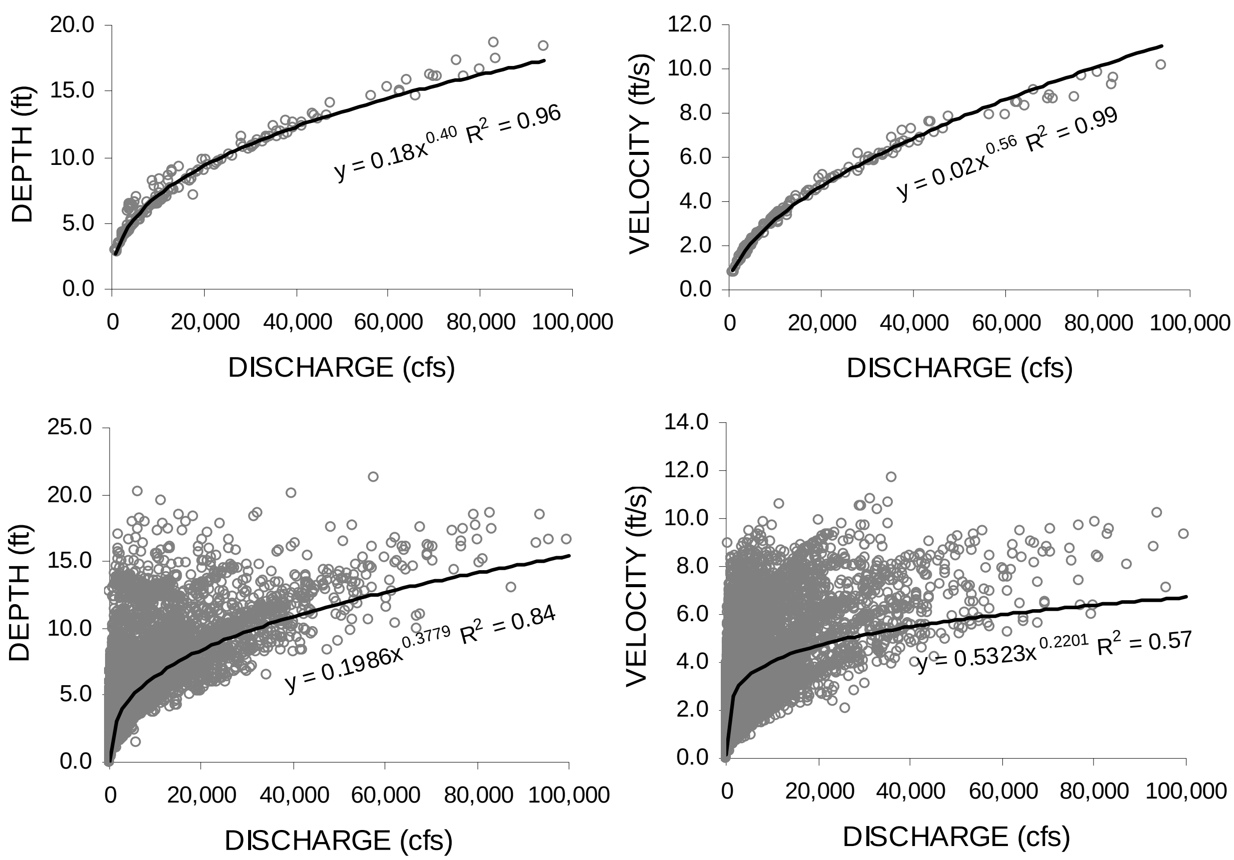

Figure 2. Top. Field observations and site rating curve for USGS station 12363000, Flathead River at Columbia Falls MT, based on 190 observations. Bottom. Field observations and rating curves for all sites.

Computation of the Wadeability Index (WI)

Following development of the site rating curves, the wadeability index was computed for each location using the baseflow annual discharge to provide a Q *vs.* WI dataset. Suplee et al. (2007) defined the onset of baseflow for wadeable streams in Montana as the point of inflection on the recession or falling limb of the mean daily hydrograph. Using this same definition for our analysis, the baseflow period for large rivers in Montana begins on August 1st (Fig. 3) and terminates on October 31st when water temperatures fall below those required for growth of the nuisance algae genus *Cladophora* sp. (Fig. 4). The WI was subsequently computed for this period using the mean of the mean monthly discharges reported by USGS.

Computed WIs ranged from 0.28-36.48 ft^2/s for all rivers examined, and averaged 5.87 ft^2/s (Appendix A). Based solely on the product number and human stability indexes developed by Abt et al. (1989), this in itself suggests that a large proportion of rivers within the state are wadeable during the baseflow period. Further analysis was then completed to determine if there was a statistically-definable change point between wadeable and non-wadeable rivers such that wadeability could be distinguished *a priori*, as detailed in the next section.

Changepoint Analysis of the Wadeable-to-Non-Wadeable Threshold

We carried out changepoint analysis on the baseflow annual discharge Q (x) *vs.* WI index (y) dataset to determine if there was a statistically-definable point where a shift in the wadeability function occurs. Changepoint analysis is a non-parametric statistical method with origins in tree-based modeling used for classification purposes (Breiman et al., 1984). The changepoint method works to minimize deviance within a dataset of paired (x, y) data (Venables and Ripley, 1994; Qian et al., 2003) and systematically bifurcates the dataset at different points along the continuum of x data until the point where the deviance of the resulting two datasets is less than that of the whole dataset. The changepoint statistic can be used to identify shift points or thresholds in x, relative to y, which are not readily identified using other methods (e.g., least squares regression). The defined changepoint was then compared to thresholds defined elsewhere in the literature (e.g., the approach used by Abt et al. [1989], or USGS safety policies).

A statistically significant ($p << 0.001$) changepoint occurred at a baseflow annual discharge of 2,670 ft^3/s, with a 90% confidence interval ranging from 1,320 to 6,540 ft^3/s (Fig. 5). This corresponds to a wadeability index of 10.5 ft^2/s with 90% CI's of 6.5 and 12.1 ft^2/s. Using the lower bound of the 90% confidence interval as a general limit, a baseflow discharge of 1,500 ft^3/s was selected as the wading threshold due to the fact that this falls between the changepoint of 2,670 ft^3/s and its lower bound of 1,320 ft^3/s. Using the lumped depth-discharge regression equation in Fig. 2, this corresponds to a depth of 3.15 ft, and an overall WI of 7.24 ft^2/s.

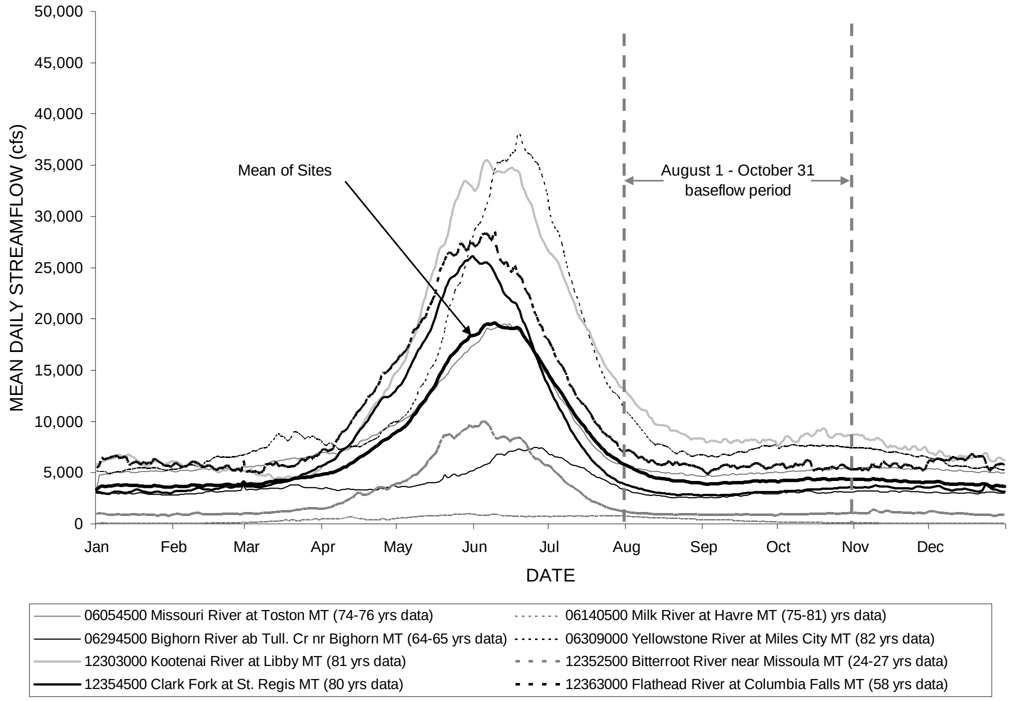

Figure 3. Mean daily hydrographs used to establish the baseflow period.

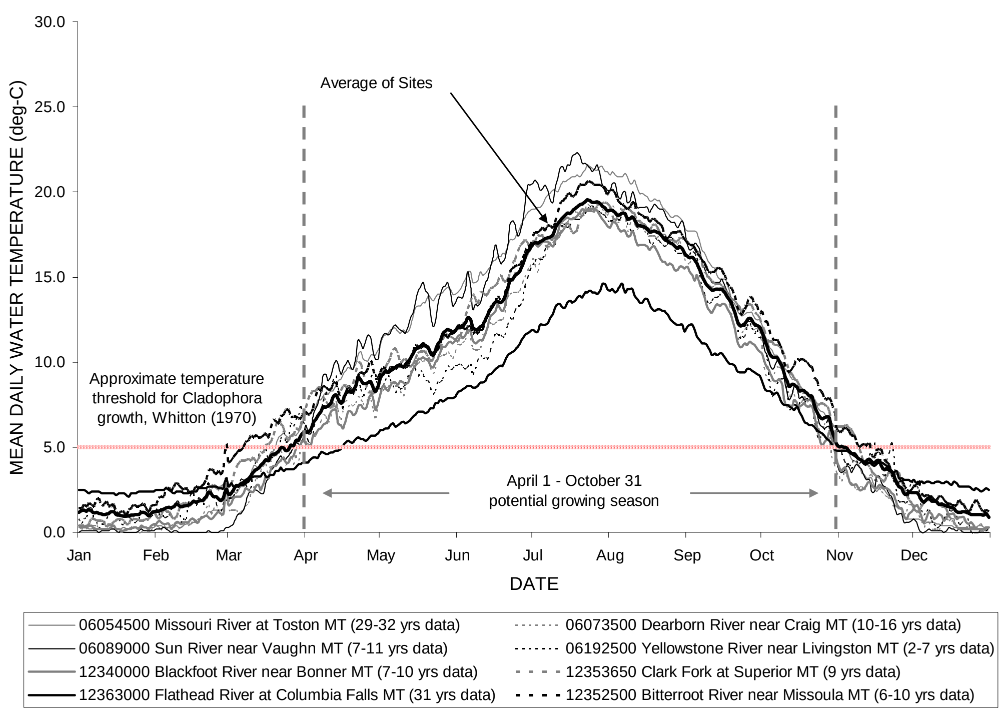

Figure 4. Mean daily water temperatures used to establish the baseflow period.

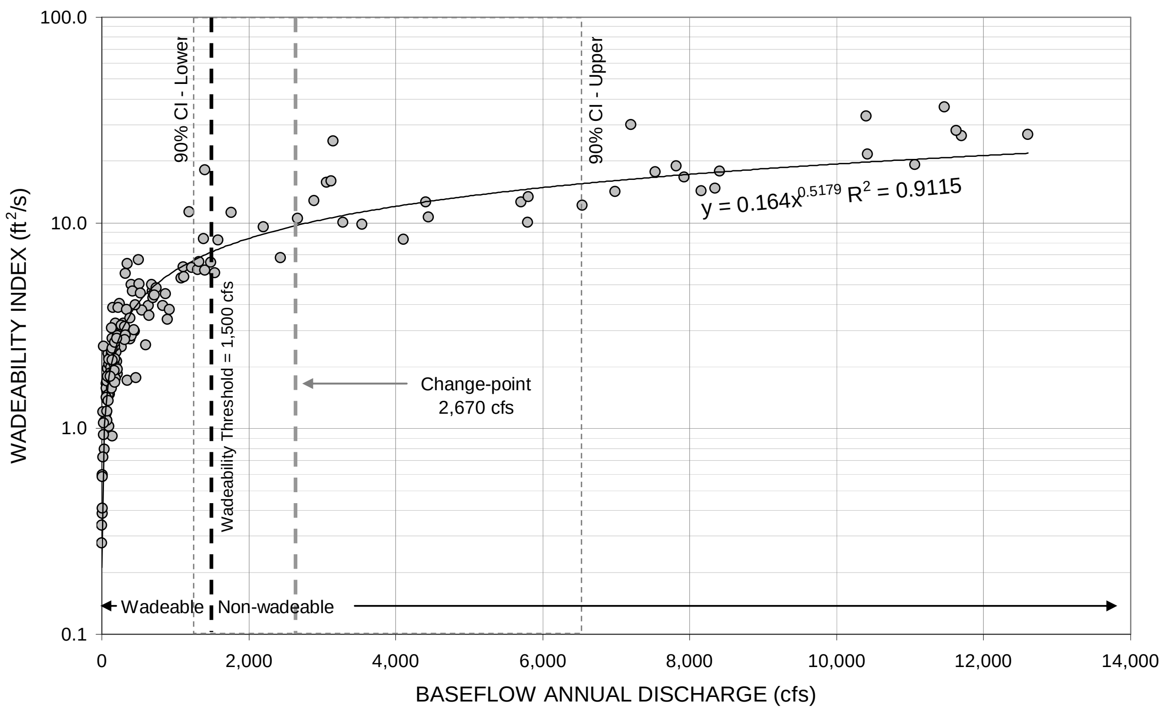

Figure 5. Baseflow annual discharge Q vs. wadeability index (WI) for the 54 sites examined in this study.

DISCUSSION

The wadeability index and wadeable/non-wadeable threshold for rivers in Montana identified through the changepoint analysis (7.24 ft^2/s) correspond well with WI values suggested in the literature. Abt et al. (1989) show that a person of fairly small stature (125 lbs, 68 inches tall), can no longer sustain themselves when WIs are as low as 7.56 ft^2/s. The USGS (safety Memo WRD99.92) uses a WI value of 10 ft^2/s as a safe wading limit. Therefore, our selected depth and WI thresholds are reasonable, and lean somewhat to the protective (i.e., safety-oriented) side. Based on the baseflow annual discharge threshold of 1,500 ft^3/s (which corresponds to a WI of 7.24 ft^2/s, or depth of 3.15 ft, as identified above), the rivers and associated segments determined to be non-wadeable for the purpose of water quality monitoring or watershed management in Montana are shown in Table 1.

Table 1. Non-wadeable river segments within the state of Montana.

River Name	*Segment Description*
Big Horn River	Yellowtail Dam to mouth
Clark Fork River	Bitterroot River to state-line
Flathead River	Origin to mouth
Kootenai River	Libby Dam to state-line
Madison River	Ennis Lake to mouth
Missouri River	Origin to state-line
South Fork Flathead River	Hungry Horse Dam to mouth
Yellowstone River	State-line to state-line

A final caveat that should not go unmentioned regarding this wadeability determination is that in some instances the mean depth, velocity, or wadeability of a specific river segment will inevitably differ from this analysis. This can be readily discerned in review of the residuals from the best-fit equation in Fig. 5 (data not shown). Discrepancies will occur due to the idealized mathematical descriptions of channel hydraulic geometry and wadeability, natural site variability, or the dynamic behavior of rivers. While the relationships presented herein are approximations only, they do provide useful information about river wadeability in general.

CONCLUSION

This document outlines an approach for defining wadeability based solely on baseflow annual discharge, a parameter that is easily estimated regardless of location, drainage area, or hydrologic considerations. In order to develop the appropriate dependencies, the following was completed: (1) hydraulic geometry relationships for 157 sites were evaluated from USGS field observations, (2) the wadeability index was calculated for each site using the mean baseflow annual discharge (i.e. product of the mean hydraulic depth and velocity), and (3) the data were then examined statistically and partitioned into a wadeable and non-wadeable population with the assistance of non-parametric changepoint methods. As a result, a baseflow annual discharge of 1,500 cfs, depth of 3.15 ft, and wadeability index of 7.24 ft^2/s were found to be a suitable threshold for wadeability, and compare well with the literature.

The process described above proved to be a useful tool in distinguishing between wadeable and non-wadeable river segments in Montana. Eight rivers within the regulatory constraints of the state were determined to be non-wadeable using this approach. These included the Bighorn, Clark Fork, Flathead, Kootenai, Madison, Missouri, South Fork of the Flathead, and Yellowstone rivers. As a result, a framework upon which future monitoring, modeling, and assessment methodologies for large rivers in Montana has now been established.

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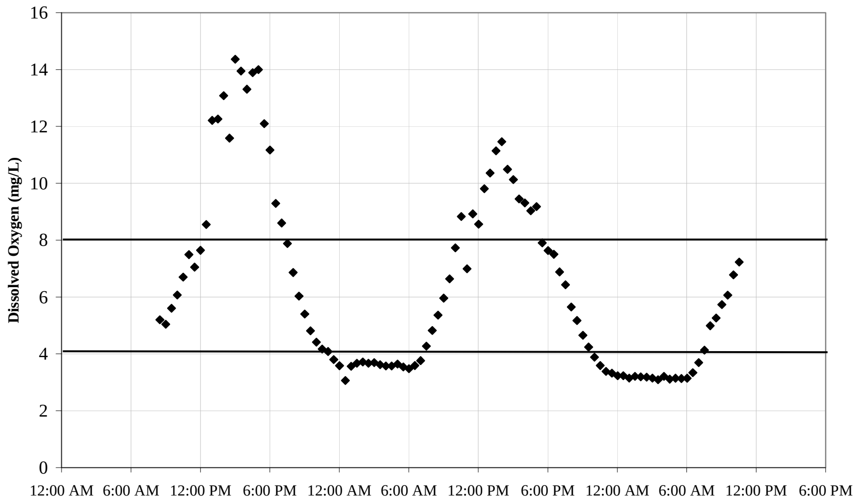
Appendix A. Channel data used in the development of wadeability index determinations. Data represent approximations to average channel conditions, and vary depending on site specifics, and location where field measurements are made.

Site ID	*Gage Location*	*Number of Field Measure-ments*	*Depth (r^2)*	*Velocity (r^2)*	*Baseflow Annual Discharge (cfs)*	*Baseflow Annual Depth (ft)*	*Baseflow Annual Velocity (ft/s)*	*Wade-ability Index (ft^2/s)*
05017500	St. Mary River near Babb MT	293	0.90	0.93	688	2.03	2.28	4.6
06012500	Red Rock R bl Lima Reservoir nr Monida MT	104	0.82	0.65	150	2.05	1.88	3.9
06016000	Beaverhead River at Barretts MT	261	0.88	0.90	497	1.90	3.47	6.6
06017000	Beaverhead River at Dillon MT	77	0.70	0.83	349	2.04	3.11	6.3
06018500	Beaverhead River near Twin Bridges MT	320	0.56	0.69	402	1.81	2.77	5.0
06019500	Ruby River above reservoir near Alder MT	328	0.80	0.78	119	1.18	1.93	2.3
06020600	Ruby River below reservoir near Alder MT	322	0.83	0.81	243	1.58	2.56	4.0
06023000	Ruby River near Twin Bridges MT	23	0.35	0.50	184	1.35	2.40	3.2
06024450	Big Hole River bl Big Lake Cr at Wisdom MT	208	0.75	0.64	54	0.90	1.32	1.2
06024540	Big Hole River below Mudd Creek nr Wisdom MT	140	0.79	0.77	139	0.91	1.01	0.9
06024590	Wise River near Wise River MT	23	0.91	0.99	82	1.42	1.38	2.0
06025500	Big Hole River near Melrose MT	312	0.84	0.84	444	1.57	1.90	3.0
06026210	Big Hole River near Glen MT	83	0.98	0.99	350	1.27	1.35	1.7
06026500	Jefferson River nr Twin Bridges MT	156	0.89	0.73	1,107	2.45	2.49	6.1
06027600	Jefferson River at Parsons Bdg nr Silver Star MT	27	0.75	0.92	831	1.92	2.06	3.9
06033000	Boulder River near Boulder MT	254	0.86	0.81	32	0.85	0.93	0.8
06036650	Jefferson River near Three Forks MT	339	0.71	0.73	1,230	3.06	1.98	6.0
06036905	Firehole River near West Yellowstone MT	140	0.42	0.43	278	1.87	1.57	2.9
06037000	Gibbon River near West Yellowstone MT	104	0.77	0.49	96	1.11	1.99	2.2
06037500	Madison River near West Yellowstone MT	179	0.63	0.78	432	1.21	2.45	3.0
06038500	Madison River bl Hebgen Lake nr Grayling MT	195	0.87	0.76	1,193	3.58	3.16	11.3
06038800	Madison River at Kirby Ranch nr Cameron MT	155	0.83	0.95	1,310	1.62	3.66	5.9
06040000	Madison River near Cameron MT	10	0.89	0.91	1,580	2.20	3.73	8.2
06041000	Madison River bl Ennis Lake nr McAllister MT	242	0.45	0.68	1,760	3.34	3.35	11.2
06042600	Madison River at Three Forks MT	46	0.79	0.89	1,390	2.94	2.84	8.4
06043500	Gallatin River near Gallatin Gateway MT	248	0.97	0.99	511	1.97	2.55	5.0
06048700	East Gallatin R bl Bridger C nr Bozeman MT	140	0.86	0.86	59	1.04	1.59	1.7
06052500	Gallatin River at Logan MT	348	0.86	0.80	637	1.59	2.49	4.0
06054500	Missouri River at Toston MT	226	0.99	0.99	3,543	3.77	2.61	9.8
06065500	Missouri River below Hauser Dam nr Helena MT	86	0.85	0.91	2,887	4.65	2.75	12.8
06066500	Missouri River bl Holter Dam nr Wolf Cr MT	417	0.86	0.93	4,413	4.99	2.53	12.6
06073500	Dearborn River near Craig MT	190	0.80	0.87	67	1.00	1.19	1.2
06076690	Smith River near Ft Logan MT	267	0.84	0.62	103	1.09	1.89	2.1
06077200	Smith River bl Eagle Cr nr Fort Logan MT	137	0.86	0.84	112	1.03	1.43	1.5
06077500	Smith River near Eden MT	47	0.92	0.89	160	1.14	1.63	1.9
06078200	Missouri River near Ulm MT	139	0.71	0.72	4,107	5.39	1.54	8.3
06078500	N F Sun River nr Augusta MT	51	0.79	0.89	142	1.11	1.72	1.9
06085800	Sun River at Simms MT	150	0.82	0.65	177	1.14	1.55	1.8
06089000	Sun River near Vaughn MT	475	0.92	0.76	464	1.19	1.48	1.8
06090300	Missouri River near Great Falls MT	269	0.95	0.97	5,797	3.67	2.74	10.1
06090800	Missouri River at Fort Benton MT	236	0.93	0.96	5,707	5.60	2.25	12.6
06091700	Two Medicine River bl S F nr Browning MT	342	0.72	0.90	121	1.15	1.61	1.9
06093600	Two Medicine River near Cut Bank MT	49	0.70	0.68	266	1.22	2.05	2.5
06099500	Marias River near Shelby MT	436	0.85	0.70	384	1.65	1.64	2.7
06101500	Marias River near Chester MT	260	0.67	0.56	872	2.26	2.00	4.5
06102050	Marias River near Loma MT	55	0.53	0.76	1,084	2.27	2.37	5.4
06102500	Teton River bl South Fork nr Choteau MT	117	0.64	0.83	88	1.04	2.24	2.3
06108000	Teton River near Dutton MT	372	0.80	0.60	70	1.08	1.42	1.5
06108800	Teton River at Loma MT	118	0.76	0.74	9	0.50	0.77	0.4
06109500	Missouri River at Virgelle MT	151	0.99	0.99	6,540	4.92	2.46	12.1
06114700	Judith River nr mouth, nr Winifred MT	81	0.70	0.83	236	1.52	1.90	2.9
06115200	Missouri River near Landusky MT	228	0.86	0.81	6,983	6.06	2.35	14.2
06120500	Musselshell River at Harlowton MT	246	0.76	0.77	71	0.97	1.42	1.4
06122800	Musselshell River nr Shawmut MT	115	0.90	0.77	62	1.16	1.35	1.6
06123030	Musselshell River ab Mud Cr nr Shawmut MT	91	0.85	0.80	35	0.83	1.30	1.1
06126050	Musselshell River near Lavina MT	134	0.89	0.78	122	1.11	1.79	2.0
06126500	Musselshell River near Roundup MT	236	0.83	0.65	132	1.25	1.85	2.3
06127500	Musselshell River at Musselshell MT	183	0.91	0.54	108	1.26	1.36	1.7
06130500	Musselshell River at Mosby MT	246	0.89	0.78	101	0.96	1.76	1.7
06132000	Missouri River below Fort Peck Dam MT	30	0.94	0.98	11,467	9.15	3.99	36.5
06132200	S F Milk River near Babb MT	253	0.73	0.60	16	0.71	1.02	0.7
06133500	N F Milk River ab canal nr Browning MT	269	0.45	0.15	17	0.87	1.37	1.2

Site ID	Gage Location	Number of Field Measure-ments	Depth (r^2)	Velocity (r^2)	Baseflow Annual Discharge (cfs)	Baseflow Annual Depth (ft)	Baseflow Annual Velocity (ft/s)	Wade-ability Index (ft^2/s)
06135000	Milk River at Eastern Crossing of Int Bndry	491	0.84	0.86	335	1.59	1.84	2.9
06140500	Milk River at Havre MT	351	0.81	0.27	424	3.03	1.54	4.7
06154100	Milk River nr Harlem MT	248	0.87	0.58	321	4.68	1.21	5.7
06155030	Milk River near Dodson MT	197	0.94	0.48	135	2.19	1.41	3.1
06155900	Milk River at Cree Crossing nr Saco MT	74	0.84	0.64	75	1.03	1.65	1.7
06164510	Milk River at Juneburg Bridge near Saco MT	232	0.92	0.48	259	1.60	1.79	2.9
06172000	Milk River near Vandalia MT	26	0.94	0.61	226	3.42	1.13	3.9
06172310	Milk River at Tampico MT	183	0.93	0.56	163	1.51	1.45	2.2
06174500	Milk River at Nashua MT	270	0.95	0.66	298	1.48	2.19	3.2
06177000	Missouri River near Wolf Point MT	220	0.88	0.77	11,700	9.34	2.84	26.5
06177500	Redwater River at Circle MT	135	0.80	0.43	2	0.44	0.77	0.3
06177825	Redwater River near Vida MT	10	0.79	0.80	6	0.49	0.83	0.4
06178000	Poplar River at International boundary	233	0.75	0.57	3	0.41	0.67	0.3
06181000	Poplar River near Poplar MT	262	0.77	0.73	27	0.91	1.03	0.9
06185500	Missouri River near Culbertson MT	123	0.85	0.87	11,067	7.59	2.52	19.1
06186500	Yellowstone River at Yellowstone Lk Outlet YNP	249	0.91	0.90	1,407	2.43	2.43	5.9
06187550	Yellowstone River at Tower Junction YNP	14	0.58	0.99	1,407	5.59	3.23	18.1
06188000	Lamar River nr Tower Ranger Station YNP	210	0.90	0.74	265	1.62	1.97	3.2
06190540	Boiling River at Mammoth YNP	166	0.65	0.91	27	1.41	1.79	2.5
06191000	Gardner River near Mammoth YNP	243	0.87	0.95	142	1.25	2.19	2.7
06191500	Yellowstone River at Corwin Springs MT	278	0.80	0.89	2,197	3.59	2.66	9.6
06192500	Yellowstone River near Livingston MT	322	0.77	0.93	2,670	3.85	2.73	10.5
06195600	Shields River near Livingston MT	323	0.91	0.92	139	1.11	1.96	2.2
06200000	Boulder River at Big Timber MT	231	0.87	0.93	219	1.47	1.93	2.8
06202510	Stillwater River above Nye Creek nr Nye MT	74	0.88	0.91	197	1.54	1.52	2.4
06205000	Stillwater River near Absorakee MT	256	0.85	0.92	677	1.77	2.83	5.0
06207500	Clarks Fork Yellowstone River nr Belfry MT	276	0.95	0.95	403	1.70	1.65	2.8
06208500	Clarks Fork of the Yellowstone at Edgar MT	201	0.85	0.89	542	1.77	2.12	3.8
06208800	Clarks Fork Yellowstone River nr Silesia MT	103	0.93	0.90	698	1.74	2.48	4.3
06214500	Yellowstone River at Billings MT	322	0.88	0.88	4,447	3.44	3.10	10.6
06279500	Bighorn River at Kane WY	257	0.90	0.91	1,537	2.48	2.30	5.7
06287000	Bighorn River near St. Xavier MT	420	0.79	0.91	3,060	5.40	2.91	15.7
06289000	Little Bighorn River at State Line nr Wyola MT	250	0.74	0.97	103	1.33	1.63	2.2
06290500	Little Bighorn R bl Pass Cr nr Wyola MT	202	0.87	0.90	114	1.16	1.60	1.9
06294000	Little Bighorn River near Hardin MT	249	0.64	0.52	134	1.23	1.94	2.4
06294500	Bighorn River above Tullock Cr, nr Bighorn MT	169	0.49	0.57	3,123	6.74	2.36	15.9
06295000	Yellowstone River at Forsyth MT	154	0.41	0.75	7,533	7.75	2.27	17.6
06298000	Tongue River near Dayton WY	479	0.93	0.98	91	1.24	1.18	1.5
06306300	Tongue River at State Line nr Decker MT	284	0.81	0.81	214	1.21	1.52	1.8
06307500	Tongue River at Tongue R Dam nr Decker MT	257	0.91	0.92	314	1.65	1.90	3.1
06307616	Tongue R at Birney Day School Br nr Birney MT	224	0.94	0.77	322	1.67	1.71	2.8
06307830	Tongue River below B. Bridge, nr Ashland MT	75	0.82	0.52	337	1.60	2.37	3.8
06308500	Tongue River at Miles City MT	282	0.83	0.70	210	1.29	1.63	2.1
06309000	Yellowstone River at Miles City MT	251	0.93	0.91	8,350	4.89	3.01	14.7
06317000	Powder River at Arvada WY	422	0.73	0.84	99	0.70	1.46	1.0
06324500	Powder River at Moorhead MT	356	0.80	0.84	183	1.14	1.52	1.7
06324710	Powder River at Broadus MT	102	0.92	0.90	172	1.37	1.50	2.1
06324970	Little Powder River ab Dry Cr near Weston WY	361	0.84	0.54	7	0.53	1.13	0.6
06325500	Little Powder River near Broadus MT	74	0.67	0.53	9	0.52	1.12	0.6
06326500	Powder River nr Locate MT	241	0.91	0.86	214	1.10	1.76	1.9
06327500	Yellowstone River at Glendive MT	43	0.77	0.98	7,923	6.60	2.53	16.7
06329500	Yellowstone River near Sidney MT	299	0.76	0.89	8,157	6.40	2.23	14.3
12301300	Tobacco River near Eureka MT	269	0.89	0.95	116	1.12	1.57	1.7
12301933	Kootenai River bl Libby Dam nr Libby MT	122	0.98	0.99	11,633	11.74	2.39	28.0
12303000	Kootenai River at Libby MT	368	0.85	0.94	12,600	4.71	5.69	26.8
12302055	Fisher River near Libby MT	51	0.85	0.86	132	1.17	1.34	1.6
12304500	Yaak River near Troy MT	364	0.95	0.96	187	1.46	1.20	1.7
12305000	Kootenai River at Leonia ID	236	0.93	0.95	7,820	4.94	3.84	19.0
12323800	Clark Fork near Galen MT	210	0.75	0.89	79	1.22	1.47	1.8
12324200	Clark Fork at Deer Lodge MT	317	0.83	0.95	175	1.37	1.59	2.2
12324590	Little Blackfoot near Garrison MT	371	0.74	0.87	64	0.94	1.50	1.4
12324680	Clark Fork at Goldcreek MT	333	0.83	0.93	314	1.47	1.85	2.7
12331800	Clark Fork near Drummond MT	214	0.88	0.83	453	1.65	2.42	4.0
12331900	Clark Fork near Clinton MT	156	0.75	0.76	526	1.93	2.35	4.5
12334550	Clark Fork at Turah Bridge near Bonner MT	244	0.78	0.74	739	1.88	2.57	4.8

Site ID	***Gage Location***	***Number of Field Measure-ments***	***Depth (r^2)***	***Velocity (r^2)***	***Baseflow Annual Discharge (cfs)***	***Baseflow Annual Depth (ft)***	***Baseflow Annual Velocity (ft/s)***	***Wade-ability Index (ft^2/s)***
12338300	N F Blackfoot R ab Dry Gulch nr Ovando MT	127	0.88	0.98	183	1.18	1.42	1.7
12335100	Blackfoot R ab Nevada Cr nr Helmville MT	104	0.72	0.67	171	1.18	1.62	1.9
12339450	Clearwater River near Clearwater MT	164	0.85	0.95	74	1.14	0.96	1.1
12340000	Blackfoot River nr Bonner MT	265	0.88	0.93	712	2.18	2.04	4.4
12340500	Clark Fork above Missoula MT	246	0.92	0.95	1,483	3.17	2.02	6.4
12342500	W F Bitterroot River nr Conner MT	298	0.90	0.98	178	1.36	1.85	2.5
12343400	E F Bitterroot River nr Conner MT	41	0.71	0.88	120	1.28	1.40	1.8
12344000	Bitterroot River near Darby MT	351	0.83	0.97	383	1.61	2.15	3.4
12350250	Bitterroot River at Bell Crossing nr Victor MT	137	0.83	0.50	441	1.89	1.60	3.0
12351200	Bitterroot River near Florence MT	76	0.94	0.88	891	2.00	1.69	3.4
12352500	Bitterroot River near Missoula MT	161	0.84	0.75	919	2.42	1.56	3.8
12353000	Clark Fork below Missoula MT	246	0.98	0.99	2,437	4.40	1.54	6.8
12354000	St. Regis River near St. Regis, MT	72	0.92	0.84	147	1.09	1.97	2.2
12354500	Clark Fork at St. Regis MT	248	0.97	0.98	3,283	4.46	2.25	10.0
12355500	N F of Flathead nr Columbia Falls MT	421	0.97	0.97	1,320	2.32	2.79	6.5
12358500	M F Flathead River nr West Glacier MT	269	0.98	0.98	1,119	2.17	2.52	5.5
12359800	S F Flathead R ab Twin C nr Hungry Horse MT	274	0.96	0.95	647	2.09	1.70	3.6
12362500	S F of Flathead River nr Columbia Falls MT	136	0.84	0.66	3,147	8.24	3.03	24.9
12363000	Flathead River at Columbia Falls MT	190	0.96	0.99	5,810	5.70	2.35	13.4
12365000	Stillwater River near Whitefish MT	162	0.95	0.71	139	2.78	0.88	2.5
12366000	Whitefish River near Kalispell MT	428	0.71	0.87	87	1.11	1.23	1.4
12369200	Swan River near Condon MT	157	0.87	0.95	76	1.12	1.09	1.2
12370000	Swan River nr Bigfork MT	323	0.94	0.97	595	1.64	1.55	2.6
12372000	Flathead River near Polson MT	412	0.42	0.99	7,203	13.12	2.29	30.1
12381400	S F Jocko River near Arlee MT	187	0.76	0.87	29	0.94	1.13	1.1
12388200	Jocko River at Dixon MT	159	0.87	0.97	180	1.34	1.96	2.6
12388700	Flathead River at Perma MT	118	0.89	0.96	8,407	8.80	2.02	17.8
12389000	Clark Fork near Plains MT	195	0.94	0.98	10,420	11.44	1.89	21.6
12389500	Thompson River near Thompson Falls MT	296	0.81	0.87	207	1.25	2.18	2.7
12391950	Clark Fork River below Cabinet Gorge Dam ID	56	0.84	0.99	10,400	21.25	1.56	33.1

Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers



Prepared By

Michael Suplee, Ph.D. — Montana Department of Environmental Quality
Vicki Watson, Ph.D. — University of Montana
Arun Varghese and Josh Cleland — ICF International

November 2008

Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers

Prepared By

Michael Suplee, Ph.D.
Montana Department of Environmental Quality
Water Quality Standards Section
1520 E. 6th Ave, Helena, MT 59620-0901

Vicki Watson, Ph.D.
Environmental Studies
University of Montana
32 Campus Drive, Missoula, MT 59812

Arun Varghese and Josh Cleland
ICF International
9300 Lee Highway, Fairfax, VA 22031-1207

November 2008

Montana Department of Environmental Quality

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EXECUTIVE SUMMARY

Beneficial uses are valuable characteristics of a stream or river resource that, directly or indirectly, contribute to human welfare. Examples of beneficial uses include drinking water, fish and aquatic life, and recreation. Beneficial uses are established in law and reflect the societal values embodied in those laws. The intent of water quality criteria, in turn, is to assure a level of water quality that will protect the beneficial uses. Some beneficial uses are more sensitive to impacts (harm) than others; water quality criteria are required by law to protect the most sensitive use from harm. This document presents the science and technical analyses used to develop numeric nutrient criteria for wadeable streams in Montana.

It is well documented that the addition of nitrogen (N) and phosphorus (P) compounds to surface waters leads to a phenomenon referred to as eutrophication. Eutrophication is increased plant and algae growth and decay in a waterbody, and all of the consequential changes to the waterbody and the water quality that occur as a result. N and P criteria are set so that they protect streams from the *undesirable* aspects of eutrophication. Undesirable aspects of eutrophication include nuisance algae growth and reduced dissolved oxygen levels which impact fish and aquatic life.

Although N and P enrichment causes stream eutrophication, the manner in which eutrophication manifests itself in streams is influenced by other factors. These factors include stream temperatures, flow patterns, light levels, and grazing on algae and plants by fish and aquatic insects. Of these, the most important in Montana appear to be temperature and flow patterns. As such, both temperature and flow patterns were incorporated into the criteria development process; this will be discussed further when the criteria are presented at the end of this summary.

Montana already has several water quality standards that address undesirable aspects of eutrophication. (Water quality standards are, essentially, criteria that have been adopted into law.) However, the existing standards are either narrative (they describe a water quality condition that should be maintained, but provide no specifics, therefore they are open to varied interpretations), or they are numeric but address only *effect* variables (e.g., low dissolved oxygen levels). Thus, one is still required to determine the root cause of the effects and that root cause is commonly nutrient enrichment. Numeric nutrient criteria will improve upon the existing standards because they address the causes of eutrophication directly.

N and P concentrations in wadeable streams vary naturally in accordance with regional geology, soils, climate, and vegetation. To address this regional variation, DEQ tested three candidate mapping systems. The intended purpose of the mapping system was to assure that appropriate nutrient criteria are applied to different regions of the state given the natural spatial variation in nutrient concentrations. Omernik ecoregions, Strahler stream order, and underlying geology (lithology) were each evaluated to see which one was the best at maximizing the difference in stream nutrient concentrations between zones and minimizing the difference within zones. Among the three mapping systems, ecoregions were found to be superior to the others and are recommended as the system upon which nutrient criteria zones should be based.

In some parts of the state, mainly in the west, the most sensitive beneficial use is recreation. A public opinion study carried out by DEQ shows that the public majority in Montana does not want to see excessive bottom-attached algae growth in the gravel-bottomed, clear running, trout-fishery streams common in western Montana. For these types of streams, the nutrient criteria have been set to prevent nuisance algal levels (as defined by the public perception study) from developing and will, therefore, protect the recreation use. The criteria will also protect the fishery, which typically comprises fish such as trout, char, and whitefish, from the negative effects of excessive nutrient enrichment (e.g., low dissolved oxygen concentrations). The criteria will also better protect the agricultural use by reducing elevated algae levels that clog irrigation systems.

In the eastern part of the state, low gradient prairie streams are common. Wadeable prairie streams in Montana often become intermittent, commonly have mud bottoms, are turbid, frequently have substantial macrophyte populations, usually have filamentous algae but sometimes have only phytoplankton algae, and support catfish, walleye, chubs, bass, and other warm water fishes. Because prairie streams are fundamentally different in many ways from western Montana trout streams, the results from the algae public perception survey should probably not be directly applied to them. Prairie streams nevertheless have important and sensitive beneficial uses that need protection, like the diverse species of fish mentioned above. For these types of streams, the nutrient criteria have been set so that they will maintain dissolved oxygen concentrations that protect regional fish and aquatic life. The most sensitive use in prairie streams is therefore considered to be fish and aquatic life.

Fundamentally, the nutrient criteria are based on scientific stressor-response studies in which harm to a sensitive beneficial use is shown. All applicable stressor-response studies (N or P as stressor, beneficial use impact as response) that could be located were reviewed. This included regional studies as well as studies from other parts of the country and world. Some of these studies were carried out by DEQ.

We then compared the nutrient concentrations indicated by the stressor-response studies to regionally-applicable reference stream nutrient data. This analysis showed that there is a consistent relationship between nutrient concentrations that harm uses (as determined in the stressor-response studies) and nutrient concentrations observed in reference sites; namely, the most elevated nutrient concentrations observed in reference sites (e.g., those at the 87th percentile of reference) are equivalent to harm-to-use concentrations. It is not surprising that reference sites have some nutrient samples whose concentrations are higher than the harm-to-use threshold identified in stressor-response studies. In any population there are always low and high values that differ considerably from the population's central tendency; the important point is that nutrient concentrations in reference sites that are greater than the harm-to-use threshold occur infrequently, e.g. due to an atypical high-flow event in summer. It is when the harm-to-use concentrations occur commonly in a stream (e.g., 50% of the time) that eutrophication problems occur.

Owing to the collective relationship we observed between stressor-response studies and the their corresponding reference data, we recommend using nutrient concentrations linked to specified upper percentiles of the reference data (e.g., the 90th of reference) as criteria. This approach has the advantage that it helps overcome statistical uncertainties in any given stressor-response study, and makes certain that natural, regional effects on background nutrient concentrations are reflected in the criteria so that the criteria will not be overly stringent or insufficiently protective.

Table E1 below shows the recommended numeric criteria for different ecoregions. Both total P and total N are recommended, as co-limitation by both nutrients is common in rivers and streams. Nitrate + nitrite (NO_{2+3}) is also suggested because total N criteria — in the absence of NO_{2+3} criteria — may not achieve the water quality goals anticipated (nitrate is particularly important in prairie streams). Analysis shows that the total P criteria should generally maintain soluble P at appropriate levels.

The criteria only apply seasonally. This is because low temperatures in winter and high flow events during spring runoff tend to mute the local effects of eutrophication (plant growth slows dramatically in winter, and spring high-flow events prevent nuisance algal mats from developing). Therefore, the criteria have been set for the time period when eutrophication problems are most likely to occur (i.e., summer). Note also that we have included benthic (i.e., bottom-attached) algae criteria for the mountainous ecoregions (Northern, Canadian, and Middle Rockies, and the Idaho Batholith). The algae levels shown are based on DEQ's nuisance algae public-perception survey. In these mountainous ecoregions, the algae criteria should be adopted along with the nutrient criteria to assure protection of the beneficial uses. In the eastern prairie streams (Northwestern Glaciated Plains, Northwestern Great Plains, and Wyoming Basin), nutrient criteria are provided but benthic algae levels are not. As noted earlier, prairie stream nutrient criteria are intended to maintain dissolved oxygen levels already in state law and are not based on the algae public-perception survey. As for all water quality criteria, the numeric nutrient criteria will undergo periodic revision and update as more stressor-response studies are completed and more reference data are collected.

Table E1. Recommended Numeric Nutrient and Benthic Algae Criteria for Different Ecoregions of Montana.

		Nutrient Criteria			
Level III Ecoregion	Period When Criteria Apply	Total P (mg/L)	Total N (mg/L)	NO_{2+3} (mg/L)	Benthic Algae Criteria
Northern Rockies	July 1 -Sept. 30	0.012	0.233	0.081	150 mg Chl *a* /m^2 (36 g AFDW/m^2)
Canadian Rockies	July 1 -Sept. 30	0.006	0.209	0.020	150 mg Chl *a* /m^2 (36 g AFDW/m^2)
Middle Rockies	July 1 -Sept. 30	0.048	0.320	0.100	150 mg Chl *a* /m^2 (36 g AFDW/m^2)
Idaho Batholith	July 1 -Sept. 30	0.011	0.130	0.049	150 mg Chl *a* /m^2 (36 g AFDW/m^2)
Northwestern Glaciated Plains	June 16-Sept. 30	0.123	1.311	0.020	n/a
Northwestern Great Plains, Wyoming Basin	July 1 -Sept. 30	0.124	1.358	0.076	n/a

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GLOSSARY OF TERMS AND MEASUREMENT UNITS

Glossary of Terms

AFDW: Ash free dry weight. The weight (mass) of a material obtained by first drying the material at 105 ^{o} C and then heating the material to 500 ^{o} C for 1 hour.

Algae: Aquatic organisms that photosynthesize but lack a vascular system. Some are microscopic, others very large.

Aquatic Macroinvertebrate: An organism found in waterbodies, frequently associated with stream bottoms, not having a spinal column and which is visible with the naked eye.

ARM: Administrative Rules of Montana.

Baseflow: The portion of streamflow that comes from groundwater and not runoff.

Beneficial Use: A valuable characteristic of a stream or river resource that, directly or indirectly, contributes to human welfare.

Benthic: On or associated with the sediments or bottom of a body of water.

Biomass: The total mass or amount of living or dead organisms in a particular area or volume.

BOD (Biochemical Oxygen Demand): A measure of, as well as a procedure for determining, how fast oxygen is used up in water. BOD is usually measured as the rate of oxygen uptake by microorganisms in a water sample, at 20°C, in the dark, over a five day period.

CFR: Code of Federal Regulations.

Chlorophyll *a*: The major green pigment found in the chloroplasts of plants and algae.

Density: Quantity of a number per unit area, volume, or mass.

Diatom: Any one of a number of microscopic algae, which can live as single cells or in colonies, that are enclosed within two box-like parts or valves (called frustules) made of silica that fit together like the halves of a Petri dish.

Diel: Involving a 24-hour period that usually includes a day and the adjoining night.

Ephemeral Stream: A stream or stream segment which flows only in direct response to precipitation in the immediate watershed or in response to the melting of a cover of snow and ice and whose channel bottom is always above the local water table.

Eutrophication: The process of enrichment of a waterbody by nutrients, usually nitrogen and phosphorus containing compounds, and the resulting increase in primary productivity (algal and plant growth and decay). Some definitions include organic enrichment of a waterbody as part of eutrophication[1].

Fixation (Nitrogen Fixation): The process by which nitrogen is taken from its relatively inert gas form in the atmosphere (N_2) and converted into nitrogen compounds such as nitrate and ammonia.

Geospatial: Pertaining to the geographic location and characteristics of natural or constructed features and boundaries on, above, or below the earth's surface; especially referring to data that is geographic and spatial in nature.

Intermittent Stream: a stream or stream segment that is below the local water table for at least some part of the year, and obtains its flow from both surface run-off and ground water discharge.

Macrophyte: Macroscopic aquatic vascular plants capable of achieving their generative cycles with all or most of the vegetative parts submerged or supported by the water.

Mainstem: The principal river within a given drainage basin, in the case where a number of tributaries discharge into a larger watercourse.

MCA: Montana Code Annotated.

Metric: A characteristic of a biological assemblage (e.g., fishes, algae) that changes in some predictable way with increased human influence.

Narrative Water Quality Criteria: Statements codified in state law that describe, in a concise way, a water quality condition that must be maintained in order to protect beneficial uses.

Nonpoint Source: The source of pollutants which originates from diffuse runoff, seepage, drainage, or infiltration.

Numeric Water Quality Criteria: Quantified expressions of water quality, in state law, intended to protect a designated beneficial use or uses.

Organic Enrichment: From a water pollution perspective, the addition of decomposable plant or animal material, or their wastes, to a waterbody.

Perennial Stream: A stream or stream segment that has flowing water year-round except during extreme drought.

Periphyton: The microscopic flora and fauna that grow or are associated with the bottom of a body of water, and includes microscopic algae, bacteria, and fungi.

Phytoplankton: Free living, generally microscopic algae commonly found floating or drifting in waterbodies such as the ocean, lakes and streams.

Point Source: A discernable, confined, and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, or vessel or other floating craft, from which pollutants are or may be discharged.

Primary Productivity: The production of organic compounds from carbon dioxide, principally through the process of photosynthesis.

Redfield Ratio**:** The molecular ratio of carbon, nitrogen and phosphorus in phytoplankton. The molar ratio is 106:16:1 (C:N:P), or 47:7:1 by weight. The term is named after the American oceanographer Alfred C. Redfield.

Salmonids: Ray-finned fish, whose members include salmon, trout, chars, freshwater whitefishes and graylings.

Saturation: A state in which the gas concentration (e.g., oxygen) in a waterbody is in equilibrium with the local partial pressure of that gas in the atmosphere.

Standards (Water Quality Standards): In a water quality regulatory context, a term applicable to state waters referring collectively to their designated beneficial uses, criteria, and the non-degradation policy, all in Montana law.

Strahler Order: A simple hydrology algorithm used to define stream size based on a hierarchy of tributaries. Streams at the top of the watershed are labeled 1. When two order-1 streams join, they create an order-2 stream. When two order-2 streams join, they create an order-3 stream, and so on. If a stream of lower order (e.g., order-2) joins a stream of higher order (e.g., order-3), the order number of the latter does not change.

Wadeable: A stream whose Strahler order is first through (at most) sixth (1:100,000 map scale) in which most of the wetted channel is wadeable by a person during baseflow conditions.

Measurement Units

cm	centimeter
ft	feet
in	inch
hr	hour
L	liter
m	meter
m^2	square meter
m^3	cubic meter
mg	milligram
mm	millimeter
NTU	nephelometric turbidity units
µg	microgram

Section 1.0
Introduction

1.1 Background

The Montana Department of Environmental Quality (DEQ) has developed numeric water quality criteria intended to control excessive nutrient pollution in Montana's wadeable rivers and streams. This document describes why numeric nutrient criteria are needed and DEQ's technical basis for developing them. The criteria are the culmination of eight years of work by DEQ. Because of the potential regulatory implications of nutrient criteria, DEQ has taken a measured, cautious approach in developing them and has strived to base them on the best available science and data. A great deal of scientific understanding about the role of nutrients in stream ecology already existed at the onset of this work, in 2000. Since that time research carried out regionally, nationally, and internationally has only further increased the scientific knowledge base. Some of this scientific work has been carried out in Montana. Of equal or perhaps greater importance has been the increased clarification of how, and at what point, surface water resources are harmed by excess nutrients. Among water quality managers, the term "beneficial use" is often heard, and will be repeated throughout this document. Beneficial uses are valuable characteristics of a stream or river resource that, directly or indirectly, contribute to human welfare. Beneficial uses are known by other names (e.g. instream values[2], valued ecological attributes[3]), but one basic truth remains the same; determining the threshold that defines when an impact to a beneficial use has occurred requires both scientific understanding and value judgment. Thus, clarity about how, when, and why beneficial uses become harmed by nutrients is at the heart of setting numeric nutrient criteria. In writing this document, it was our intent that the reader gains a basic understanding of the ecological role of nutrients in streams, how beneficial uses become harmed by nutrients, and how the criteria are expected to prevent the latter from occurring.

1.2 Scope of the Criteria

The criteria discussed in this document apply specifically to *wadeable* streams. A wadeable stream is generally defined here as a stream whose Strahler order[4] is first through (at most) sixth (1:100,000 map scale) in which most of the wetted channel is wadeable by a person during baseflow conditions. This includes perennial streams that run all year and streams that become intermittent, i.e., a disconnected series of pools. The criteria do not apply to ephemeral channels, defined in Montana as a stream or stream segment which flows only in direct response to precipitation in the immediate watershed or in response to the melting of a cover of snow and ice and whose channel bottom is always above the local water table (ARM 17.30.602[12]). Nor do the criteria apply to large rivers, which are generally 7^{th} order or larger and in which most of the wetted channel is unwadeable during baseflow; examples include the Yellowstone River and the Missouri River (Table 1.1). Finally, the criteria do not apply to lakes or wetlands. Separate efforts are currently underway to develop criteria for large rivers and lakes, and will be addressed in a future document similar to this one.

Table 1.1 Large River Segments in Montana. This Table May Be Subject to Further Refinement.

River Name	Segment Description
Clark Fork River	From Rock Creek confluence (46.726, -113.683) to Idaho border*
Flathead River	From the Flathead Indian Reservation boundary to the mouth
Kootenai River	From Libby Dam to Idaho border
Missouri River	Entire length in Montana
Marias River	From Tiber Dam to the mouth
Musselshell River	From Flatwillow Creek confluence (46.928, -107.930) to Fork Peck Reservoir
Milk River	From Fresno Dam to the mouth
Yellowstone River	Entire length in Montana
Bighorn River	From Yellowtail Dam to Crow Indian Reservation boundary near St. Xavier, MT, and from the Crow Indian Reservation boundary near Hardin, MT, to the mouth
Tongue River	From Hanging Woman Creek confluence (45.321, -106.522) to the mouth
Powder River	From Wyoming border to the mouth

*Numeric nutrient and algal biomass standards already are in place from the Rock Cr confluence downstream to the Flathead River confluence (ARM 17.30.631)

1.3 Nutrient Criteria are Different than Other Water Quality Criteria

Nitrogen and phosphorus are essential nutrients for plants and animals. Without them, organisms would not be able to build the proteins and nucleic acids of their cellular structures or carry out the basic oxidation and reduction reactions that power each of their cells. Because nitrogen and phosphorus are necessary for all organisms, the effects these nutrients have in the aquatic environment are inherently different from the effects of toxic substances. Many substances (e.g., lead and mercury) are toxic to people or aquatic organisms in the tiniest of concentrations[5] and, traditionally, water quality criteria[a] for these types of elements and compounds have been derived by toxicologists using laboratory studies. Some nutrients (e.g., certain nitrogen compounds) can be, at fairly high concentrations, toxic to people and aquatic organisms as well. But criteria have already been established for those effects. The nitrogen and phosphorus criteria presented in this document are intended to control the undesirable aspects of an environmental effect referred to as eutrophication (or sometimes “cultural” eutrophication).

Eutrophication is the enrichment of a waterbody by (typically) nitrogen and phosphorus, leading to increased plant and algae growth and decay, and all the consequential changes to the waterbody and the water quality that occur as a result of this enrichment[1]. Enrichment of waterbodies by nutrients is not in and of itself negative. Many waterbodies are purposefully enriched in order to enhance their overall productivity; fertilization of commercial fish ponds to increase growth of certain fish species is such an example. Enrichment becomes detrimental — and therefore begins to fall within the realm of water quality criteria setting — when the effects manifested in a waterbody are undesirable relative to the uses the waterbody is intended for. Nitrogen and phosphorus concentrations in waterbodies have the interesting quality that as they increase they may initially enhance certain waterbody characteristics (e.g., cause fish to grow

[a] Water quality criteria are numeric or narrative expressions of water quality that, if achieved, assure that important characteristics of a water resource are not damaged.

larger) but then, at higher concentrations, lead to conditions that harm the value of the waterbody (e.g., result in low dissolved oxygen that impairs the fishery). Thus, setting numeric nutrient criteria requires an understanding of how eutrophication progresses with increasing nutrient concentrations, and at what point the detrimental effects begin to occur. This document will discuss each of these issues in detail.

1.4 Beneficial Uses and Criteria

Identifying when water quality changes begin to have detrimental effects on stream resources is key to setting appropriate water quality criteria. But even before a detrimental effect or criterion can be considered, some frame of reference is needed to help set the expectations. This frame of reference is established via the stream's beneficial uses. Beneficial uses are the valuable characteristics of a stream or river resource that, directly or indirectly, contribute to human welfare. They are established in law and reflect the societal values embodied in those laws. Some typical examples of beneficial uses in streams are public water supply, fish and aquatic life, agricultural use, and recreational use. Montana has had this very list of beneficial uses in law since 1955. Montana records its beneficial-use definitions in its Administrative Rules (ARMs) at 17.30.601 *et seq*. Beneficial uses became more uniformly defined nationally when the U.S. Congress amended the Federal Water Pollution Control Act in 1972, leading to what is commonly called the Clean Water Act. The U.S. Clean Water Act clearly states that water quality should be protected so that fish propagation and recreation (i.e., fishable and swimmable) would be achieved in all waterbodies, where possible.

Once the beneficial uses of waterbodies are established, the framework is in place to develop mechanisms to protect the beneficial uses from harm. This leads directly back to water quality criteria. As noted earlier, criteria are numeric or narrative expressions of water quality that, if achieved, assure that the uses are not harmed by human action. As might be expected, some uses are more sensitive to impacts than others. For example, both the fish and aquatic life use and the industrial use are beneficial water uses. But fish are much more sensitive to many water quality changes (e.g., increases in copper concentration) than most industrial processes would be to the same changes. Thus, the fishery use is — in this example — the most sensitive use. In Montana all uses are valued the same; i.e., all uses are treated equally. Water quality criteria in Montana are set to protect the most sensitive use, with the understanding that the less sensitive uses will be protected automatically; this approach is also required under Federal law (40 CFR 131.11). This method was used in developing the numeric nutrient criteria presented in this document.

As states, tribes, and the U.S. Environmental Protection Agency (EPA) have developed water quality criteria to protect beneficial uses, they have used a variety of techniques to identify when harm occurs to the use. These techniques are a function of the use in question and the specific technical approaches required. But these techniques all have in common the integration of value judgments and science. Beneficial uses and their associated impact (or harm) thresholds reflect societal values codified in law, while the scientific method provides understanding as to how particular water quality parameters affect streams and quantifies when those parameters harm the use. Figure 1.1 provides an example of harm to a beneficial use (salmonid fishes). Large day-to-night dissolved oxygen (DO) changes measured in this Montana stream were linked to human pollution causes; note how they go over 14 mg/L in the day and drop below 4 mg/L at night. It is well established scientifically that low DO concentrations harm fish[6,7]; for example, weight gain and food conversion (amount consumed to amount of weight gained) of young salmonid fishes drops rapidly when DO is below 4 mg/L[6]. For the stream in Figure 1.1, DO levels of 8 mg/L for juvenile fish and 4 mg/L for adult fish are instantaneous minima in Montana law[8] intended to assure proper egg development[b] and normal growth and activity for completing all life stages of fish like rainbow and brook trout. The DO concentrations in the stream in Figure 1.1 fall below these criteria and are, therefore, harming the beneficial use.

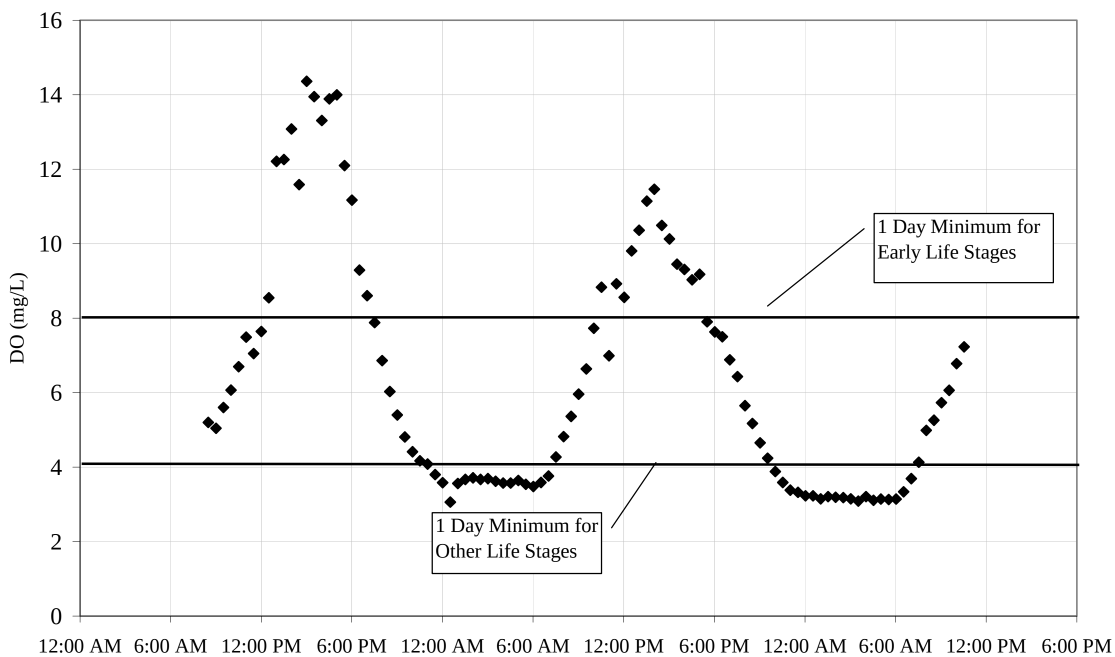

Figure 1.1 Diel Dissolved Oxygen (DO) Patterns in a Montana Stream, 2003. The DO concentrations at 8 and 4 mg/L are intended to protect juvenile and adult fish, respectively.

[b] Montana water quality standards require 8 mg DO/L in the water in order to achieve 5 mg DO/L in the inter-gravel region where embryonic, larval, and juvenile fish are commonly found.

Section 2.0
The Science of Stream Eutrophication

2.1 Stream Eutrophication, Nitrogen, and Phosphorus

Eutrophication (nutrient enrichment) management efforts focus on nitrogen (N) and phosphorus (P) concentrations and not on other nutrients (e.g., trace minerals, vitamins, etc.) for good reason. After much scientific research concerning which major nutrients most often limit or control primary productivity in freshwaters, there is general consensus that it is typically P and or N. Major nutrients required by organisms are carbon, oxygen, hydrogen, nitrogen, and phosphorus[9] and, at various times, limnologists (scientists that study fresh waters) have considered most of these elements as candidates for controlling productivity in freshwaters. In the 1960s there was considerable debate about the relative importance to primary productivity of inorganic carbon *vs.* phosphorus[10], but Schindler's influential work on whole-lake fertilization[11] showed that phosphorus limited lake productivity, and carbon did not. After Schindler's lake work in the 1970s, there was a general emphasis on P as the sole nutrient limiting productivity and controlling eutrophication in freshwater systems. However, N has in more recent decades been found to be of equal importance in rivers and streams, where N and P co-limitation appears to be common[12]. This has been further confirmed by whole-river and whole-stream fertilization experiments using N and P that demonstrate that production in flowing waters is strongly controlled by N and P[13,14]. Plant physiologists have identified a number of required micronutrients[15] that could potentially limit freshwater productivity, but these substances are needed in minute quantities and field studies show they do not limit stream production[16]; thus, it is very unlikely that they generally limit freshwater productivity[1,9,17]. Site-specific exceptions to the previous statement may exist, e.g. in some granitic alpine areas[9], but these are not widespread enough to warrant consideration for water quality management purposes.

Nitrogen and P enter streams by a variety of routes. Point sources (e.g., waste water treatment plants) with an end-of-pipe discharge are probably the most conspicuous. But they are not the only sources and, in certain situations, are not even the largest contributors. Since the end of WW II humans have dramatically increased the fixation of N (e.g., via the Haber-Bosh industrial process which converts unreactive atmospheric N to ammonia salts) to the point that human sources of fixed N now exceed natural sources on a global scale[18]. Most of this fixed N is used for growing food crops and, as a result, large amounts of N have entered streams via point and nonpoint sources. Agricultural sources are a major sources of N and P to streams, and are often a mix of organic and inorganic forms, while other nonpoint nutrient sources include soil and stream bank erosion, urban runoff and sprawl, land clearing and conversion, and loss of wetlands and the subsequent oxidation of their organic soils[7,17,19].

2.2 How Eutrophication Manifests in Streams

Eutrophication has a number of effects in flowing waters. One very typical change is the increased dominance by benthic (i.e., bottom attached) filamentous algae in temperate streams. The green algae *Cladophora spp.* in particular seems to benefit from increased nutrient enrichment. *Cladophora* (Figure 2.1) probably played a minor role in aquatic communities

Figure 2.1 Example of Heavy Cladophora Growth in a Wadeable Montana River (Aug., 2004).

before widespread cultural eutrophication occured[20,21], but dense growths of the alga have now become common in nutrient-enriched temperate streams worldwide, including in Montana[20,22-24]. Another common effect of eutrophication in streams and rivers is the increased magnitude of daily DO and pH oscillations due to the elevated productivity of phytoplankton, benthic algae, or both[25]. Aquatic insect (aquatic macroinvertebrate) populations often shift in response to increasing nutrient enrichment and there is a large scientific literature devoted to the relationships between macroinvertebrates and water quality. Very generally, sensitive macroinvertebrates, including mayflys (Ephemeroptera), stoneflys (Plecoptera), and caddisflys (Trichoptera), tend to be found in clean water having low nutrient concentrations and DO near saturation (i.e., without extreme daily oscillations). At the other end of the spectrum, many midge species (chironomids) are tolerant of heavy eutrophication and the associated conditions (e.g., low nighttime DO)[21,26-28]. Downstream of efficiently-treated wastewater discharges where the effluent contains mainly inorganic N and P (i.e., little organic matter), overall macroinvertebrate density and biomass increase, but the density of pollution sensitive species diminishes[29].

Fish populations are also affected by eutrophication. It has been shown that increasing N and P concentrations in streams can result in increased growth of fish, including salmonid fishes (e.g., trout, char)[13,14,30,31]. Phosphorus is shown to have a threshold effect on salmonid fish populations, whereby the number of fish found per 100 m of stream length initially increases with increasing P, but then drops after P reaches a certain threshold concentration[28] (Figure 2.2). Ultimately, if eutrophication becomes too severe, fish kills can occur due to low nighttime DO[32]. Studies also show that stream eutrophication (quantified as increasing benthic algae density) leads to greater accumulation of organochlorine pollutants (e.g., PCBs) in localized trout populations. This occurs because increased primary productivity in rivers slows downstream transport of the PCBs, allowing fish more time to ingest them[33,34].

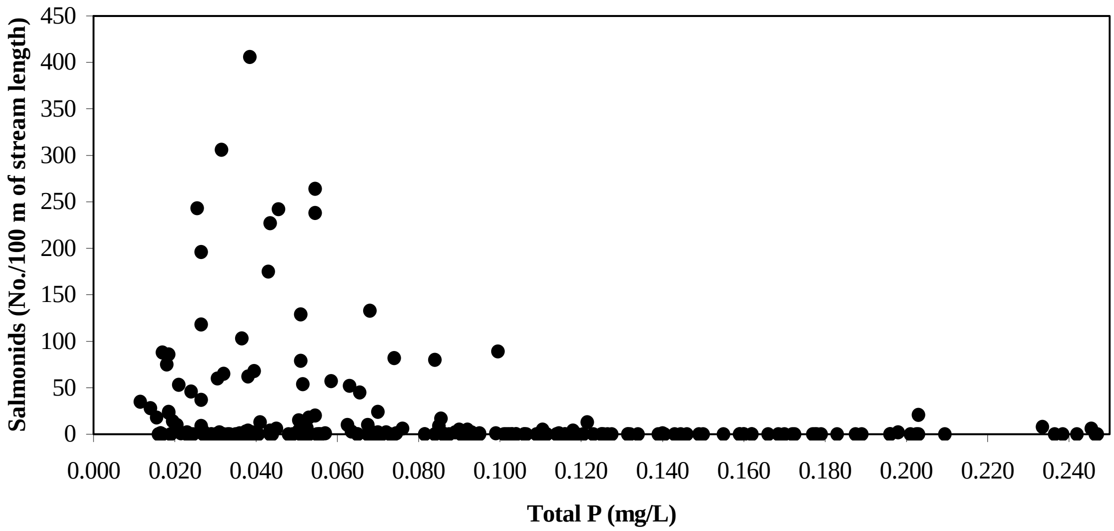

Figure 2.2 Number of Salmonid Fish Per Unit Stream Length in Wisconsin Streams. Modified from Figure 3 in Wang *et al.* (2007) to show greater resolution at lower total P concentrations. Presented with permission of the authors.

2.3 The Influence of Other Stream Environmental Factors on Eutrophication

Non-nutrient environmental factors influence the way eutrophication manifests itself in streams. At a very coarse scale, climate, source of flow, and geology set the stage for the types of streams that develop in a region[35]. Within any particular stream, factors external to nutrients that influence primary productivity include light, temperature, water velocity and high-flow events, grazing by fish and macroinvertebrates, and inter-specific competition among the plants[36]. Those we considered among the most important to Montana streams are detailed below.

When sufficient light reduction occurs, eutrophication effects are muted, at least locally, because aquatic plants driving the productivity become light limited[37-39]. However, light must be reduced by 60% or more from ambient levels in order to reduce algae proliferations[2,39]. Many smaller streams in western Montana have riparian canopies which substantially diminish light levels at the stream surface, but it is also true that un-canopied areas are commonly interspersed along their lengths as they flow through open meadows and pastures. Light attenuation is provided by riparian canopy, but it can also result from instream turbidity; Figure 2.3 is an example of the latter from a Montana river. In 2007, a tributary upstream of the Figure 2.3 data-collection site had a summertime high flow event and discharged highly turbid, sediment-laden water to the river. Productivity, measured indirectly as the magnitude of diel DO oscillation in the river[40], was notably dampened during the event (days 1 through 3) after which productivity increased as turbidity fell to more normal levels. Note the very high turbidity levels (> 600 NTU) that were needed to induce the dampening effect (Figure 2.3). Turbidity levels in western Montana streams are, outside of runoff, typically well below 50 NTU, and are not likely to limit instream productivity. In contrast, turbidity levels in eastern Montana prairie streams are sometimes above 100 NTU and midsummer values as high as 1,830 NTU have been measured in ones considered to have few human impacts[41,42]. Turbidity in these prairie streams correlate well with suspended sediment[41], therefore it is likely that Montana prairie streams experience periodic reduction of their productivity due to sediment turbidity.

Stream velocity and high-flow events interact with eutrophication and influence benthic algae in two distinct ways. Water velocity — up to a point — can allow larger algae mats to grow than is possible in quiescent water because the flow induces nutrients to reach algae cells at the base of the mat which might otherwise be starved of nutrients by the algae growing above them[2,43]. In contrast, high flow events beyond that which the algae are adapted to leads to reduced biomass via sloughing and scouring[44,45]. Benthic algae scouring can occur in Montana during the summer due to isolated high flow events caused by localized thunderstorms; this is especially true in eastern Montana where such storms are common and stream flows tend to be flashy. High flow events act as a reset mechanism, after which rapid re-growth of the algae can and often does occur[44].

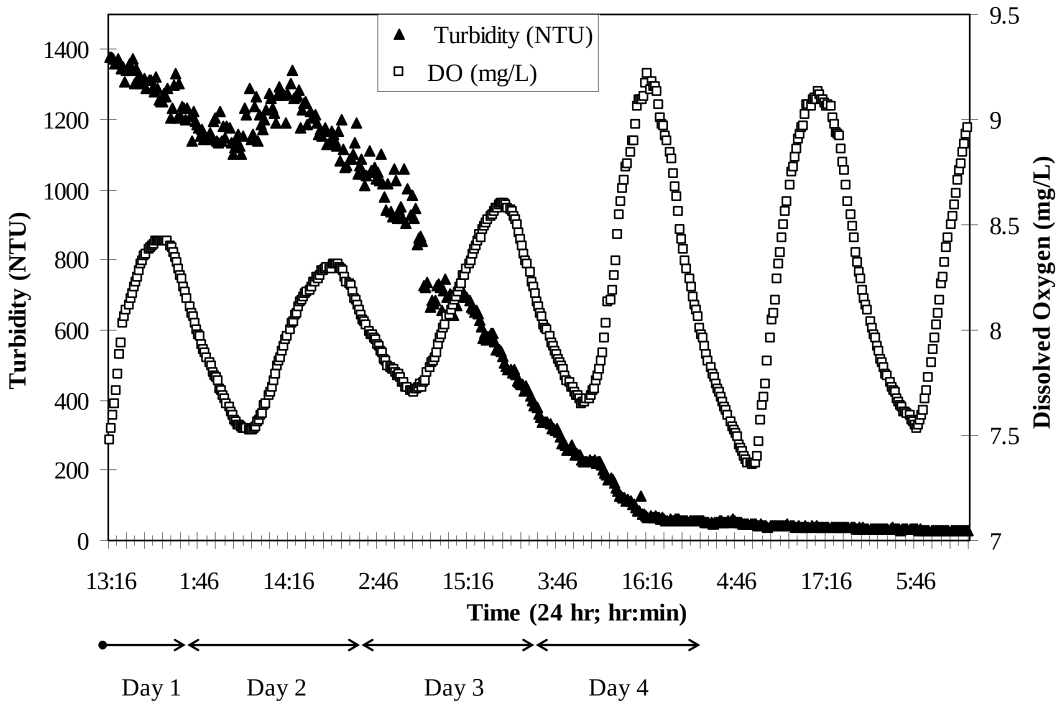

Figure 2.3 Influence of Light Attenuation on Productivity in a Montana River. Unusually high turbidity dampened DO changes over a three day period. During the 4th day, turbidity dropped to more typical background levels and the magnitude of the diel DO oscillations (i.e., productivity) increased markedly.

Grazing by fish and macroinvertebrates affects plant productivity in streams, however studies in this area of stream ecology are often conflicting and there is still much scientific debate. An excellent literature review is provided by Steinman[46] but only the most pertinent aspect of his review (biomass-grazing relationship) is summarized here. Most studies show — not surprisingly — that algal biomass decreases in response to fish and macroinvertebrate grazing, although a few studies show algae biomass actually *increases*. Algal biomass might not decline due to grazing because (1) grazer density and consumption rates are insufficient to induce a decline, (2) the type of grazing is not well matched to the dominant algae forms, and (3) other resources (e.g., nutrients) are limiting and biomass is low regardless of grazing. We are not aware of any published studies of grazing effects on aquatic plants carried out in Montana, and can only offer our own observations on this topic. Based on many years of working in Montana streams, it is our conjecture that grazing by fish and macroinvertebrates does not play a large role in reducing heavy algal growth in Montana streams. Snails are a common grazer[46], but we have not often observed high snail densities in either western or eastern Montana streams, probably due to harsh conditions in winter. An exception to this is in a few streams where the invasive New Zealand mud snail (*Potamopyrgus antipodarum*) has become established. Spring creeks also tend to have somewhat elevated snail densities. In near-pristine streams, biomass is probably constrained by low nutrients (reason No. 3 above). In other, more eutrophied streams, where we have observed very high algal biomass (often *Cladophora*) growing in thick mats, low grazer density, or the less-than-preferable nature of the algae as a food source, seem equally likely to explain why grazing has not observably diminished heavy benthic algae growth. Alternatively, benthic algal biomass might increase due to macroinvertebrate grazers, because grazers can stimulate *Cladophora* growth by removing epiphytes growing on the surface of the algal filaments[47,48].

Finally, there is inter-specific competition among stream aquatic plants. Of particular relevance in Montana is the competition observed between benthic plants and phytoplankton in the state's eastern prairie streams. Benthic algae and phytoplankton compete with each other for resources (light, nutrients, etc.) and, once one or the other of the two plant groups gains the upper hand, a positive feedback loop can ensue that leads to the near domination by that group[49]. This process appears to occur in Montana prairie streams, most often when they become intermittent[41]. An analogous regional example of this process described in the scientific literature is found in shallow lakes of the Canadian Boreal Plain[50]. These lakes switch status from year to year between (state 1) phytoplankton domination and turbid water, and (state 2) domination by submerged aquatic vegetation and clear water. Harsh winters (i.e., non-biological factors) tend to reset the lakes each year, and so the state (1 or 2) that dominates the following year is dependent on small perturbations present in the spring[50]. As mentioned, a similar phenomenon has been observed in Montana prairie streams[41](see also Discussion, Appendix A), and where prairie streams are eutrophied this can result in heavy blooms of phytoplankton with chlorophyll *a* concentrations as high as 515 µg Chl *a*/L[41]. Phytoplankton influence DO as do benthic plants, and in eutrophied prairie streams low DO seems equally likely to result from phytoplankton blooms or from heavy benthic plant growth.

Section 3.0
Existing *vs.* Proposed Approach to Controlling Stream Eutrophication

3.1 The Push for Numeric Nutrient Criteria at the National Level

Eutrophication has long been recognized as a major water quality problem by EPA, illustrated by the fact that the agency undertook a national eutrophication survey of streams just shortly after its creation in the early 1970s[51]. In the late 1990s EPA announced that all states and tribes must develop nutrient criteria for their respective waters, and by 2000 EPA had published a series of regionally-based numeric nutrient criteria recommendations[52]. As of this writing, EPA's current policy position is that each state and tribe should adopt numeric criteria, but they can develop and adopt the criteria according to mutually agreed upon plans and schedules. There are already a number of states that have adopted numeric nutrient criteria (e.g., Tennessee, Hawaii, Connecticut), and many more states are well along in the development process.

3.2 How Eutrophication has been Addressed in Montana Up Till Now, and How Numeric Nutrient Criteria Will Improve the State's Existing Water Quality Standards

Although DEQ is developing statewide numeric nutrient criteria for the first time, eutrophication has long been recognized as a problem and water quality laws exist to help address it. How have the negative aspects of stream nutrient enrichment been addressed in Montana to date? Montana has several water quality standards that generally apply to eutrophication, including a limited number of numeric nutrient criteria. Numeric nutrient criteria are established on large reaches of the Clark Fork River and define reach-specific nutrient concentrations and benthic algae biomass levels for the river (ARM 17.60.631). These were adopted in 2002 and are intended to prevent nuisance growth of benthic algae by limiting the river's N and P concentrations. Other numeric standards that are applicable to all state waters and that address eutrophication-related water quality problems are the numeric DO criteria[8], and the pH criteria (e.g., ARM 17.30.623[2][c]). These standards are intended to protect fish and aquatic life uses.

In addition to the regulations cited above, Montana has a narrative criterion that covers other unwanted aspects of eutrophication. Narrative criteria are codified statements that describe, in a concise way, a water quality condition that must be maintained. However, unlike numeric criteria, there are no quantitative values associated with narratives. The Montana narrative criterion that is applicable to eutrophication specifies that "State surface waters must be free from substances attributable to municipal, industrial, agricultural practices or other discharges that will create conditions which produce undesirable aquatic life" (ARM 17.30.637[1][e]). Narrative criteria have the advantage that they are flexible and cover many potential situations (even unforeseen ones), but because they lack specificity, they are open to varied interpretations.

An obvious question that arises is "why adopt statewide numeric nutrient criteria if Montana already has other criteria that address eutrophication?" DEQ asked this very question when it began to develop statewide numeric nutrient in 2000 and concluded that the existing criteria were not sufficiently addressing some types of water quality problems[53] (Clark Fork River criteria excluded). Clearly, something about the DO, pH, and narrative criteria was not and is not working when it comes to stream eutrophication, since eutrophication problems continue to be common in Montana[54]. When it comes to eutrophication, the shortfall of the numeric DO and pH criteria is that they are *effect*, or secondary, variables in streams, and require one to further seek the specific, primary cause of the impact. That is, DO and pH are being driven by other factors and those "other factors" are often excess nutrients. To properly implement the DO criteria where eutrophication is involved requires nighttime measurement of DO (when the minima usually occur), a cause-effect linkage between DO and nutrients, and an understanding of the nutrient concentrations that would prevent the low DO from occurring. Similarly, the pH criteria require an understanding of a waterbody's natural background pH, the degree of change from background, and the cause. Thus, if one knew the nutrient concentrations that could prevent exceedences of the DO and pH criteria in a waterbody, one has a good chance of actually attaining the DO and pH criteria because the root cause of the problem would be addressed. That is exactly what numeric nutrient criteria are intended to do.

The narrative criterion (ARM 17.30.637[1][e]) has more difficult implementation challenges than the DO and pH criteria do. In particular, there are no definitions in rule of what "undesirable" aquatic life is, or, if that could be determined, what the levels of this aquatic life should be held to. If undesirable aquatic life can be defined and maximum allowable levels of it are established, then the situation resembles that of DO and pH in that one needs then to determine appropriate nutrient concentrations where enrichment is involved. But because undesirable aquatic life has not been defined heretofore, the application of this criterion has been subject to individual interpretation and, consequently, debate. In developing the numeric nutrient criteria it was necessary to (1) identify and quantify undesirable (i.e., nuisance) aquatic life attributable to eutrophication, (2) determine a level of that aquatic life that would not harm the beneficial uses, and (3) identify the nutrient concentrations that would maintain said aquatic life at non-nuisance levels. As a result, the numeric nutrient criteria in this document closely reflect the spirit and intent of the narrative criterion but also provide sufficient detail to make it of practical value.

3.3 Recommended Nutrients, and Determining Compliance with the Criteria

Criteria for total N (TN), total P (TP), and nitrate + nitrite (NO_{2+3}) are proposed. Research shows that total nutrients provide better overall correlation to eutrophication problems in streams than do soluble nutrients[55-57] and, in addition, EPA has indicated that TN and TP are the minimum acceptable nutrient criteria[58]. Although total nutrients correlate best overall with eutrophication, soluble NO_{2+3} concentrations are also important. This is because there is the potential that if TN criteria are adopted in the absence of accompanying NO_{2+3} criteria, the water quality benefits desired would not be achieved. For example, if a point source discharge met the TN criterion but the discharge was almost all nitrate, water quality problems would in all likelihood continue. This is particularly true in eastern Montana prairie streams, where there are strong indications of nitrogen limitation and studies suggest that special attention should be paid to NO_{2+3} [59,60]. We believe that soluble P criteria need not be promulgated because, as will be shown in Section 6.3.2, the TP criteria concentrations being recommended should maintain soluble P at acceptable levels. Therefore, we recommend TN, TP, and NO_{2+3} for numeric nutrient criteria.

Recommendations are provided for determining compliance with the numeric nutrient criteria in Appendix H (for TMDLs and 303(d) listing) and Appendix I (for point sources) in a recently completed technical report[61]. Readers interested in the details of the recommended compliance-determination techniques should read the aforementioned appendices. The most important point to summarize here is that the criteria will have a certain allowable exceedance rate based on appropriate statistical evaluation of the data. Specifically, 20% of the data from a population of water quality data could exceed any given criterion and still be in compliance with the standard. This exceedence rate was determined empirically using Montana data and falls within the range of exceedence rates recommended by EPA[61]. A 12 sample minimum is recommended.

3.4 Difficulties with Numeric Nutrient Criteria

Numeric nutrient standards will help Montana to better protect beneficial uses and water quality. Numeric nutrient criteria are not without their own difficulties, however. In our opinion, the major issues are (1) the question of geographic specificity of the criteria, and (2) the achievability of the criteria using current wastewater treatment technologies.

Nutrient criteria concentrations proposed by DEQ will be different from place to place due to local differences in geospatial features (e.g., climate, geology, soils) and their combined effects on stream nutrient concentrations. DEQ has carried out substantial research to assure that the classification system used to differentiate nutrient expectations is regionally appropriate (much more on this, Section 4.0). But whenever a classification system is used, decisions have to be made as to whether it is better to lump individual categories, or to further split them. There are two ends to this continuum; at one end, there is a classification system so "split" that each stream falls in its own class (i.e., each stream is unique), while at the other end of the spectrum all streams are "lumped" together (i.e., all streams are the same). Neither of these is appropriate for nutrient criteria. Instead, DEQ strove to find a useful and practical balance between these two extremes when developing the numeric nutrient criteria.

Nevertheless, there will be cases where the criteria are not exactly appropriate for a given stream due to local conditions not sufficiently addressed by the classification system or because the classification is too coarse. Some of the localized, confounding environmental factors that change the way eutrophication is manifested in streams have already been presented (high flow events, shading, etc.; Section 2.3). The confounding environmental factor we consider most likely to render the criteria inappropriate is the near-field (i.e., proximate) effect of upstream dams (which are addressed by other MT laws, e.g. 75-5-306[2], §MCA), and upstream natural lakes. However, there are other state water quality laws (e.g., "it is not necessary that wastes be treated to a purer condition than the natural condition of the receiving stream…", 75-5-306[1], §MCA) not discussed in this document that can address any gross criteria misfits on a case-by-case basis. It is worth pointing out that the establishment of beneficial uses and water quality criteria in Montana have traditionally been somewhat "broad-brush". This approach has the advantage that *all* waterbodies are protected under law, but as a consequence criteria or use classes may not make sense for some specific waterbodies and warrants additional site-specific consideration.

The other major issue that has become clear is that nutrient concentrations that prevent the unwanted aspects of eutrophication are quite low relative to current wastewater treatment technologies. Scientific studies show that it only takes small amounts of nutrient enrichment to manifest changes in streams[43,62]; this region of the country appears to be particularly sensitive and the specifics of this will be detailed in Section 6.0. The implication for Montana is that the criteria will be difficult to achieve in some places, especially where a point-source discharge is a large proportion of a receiving stream's volume. DEQ is developing implementation policies that will help dischargers deal with this; those efforts are on going, and we will not present those here. It is also important to note that wastewater technologies are rapidly advancing, hence, lower and lower N and P concentrations can be routinely achieved for less money. DEQ anticipates that the numeric nutrient criteria are, ultimately, achievable, even if dischargers need time for treatment technologies to mature and costs to come down.

Section 4.0
How DEQ Selected a Geographic Stratification System to Apply Different Nutrient Criteria in Different Places

An essential step in setting numeric nutrient criteria involves deciding how to divide the state into regions or nutrient zones in which a single N or P criterion would apply. This section discusses why such a geographical stratification system is necessary and how it was developed.

4.1 Purpose of Developing a Geographic Stratification System

As mentioned in Section 3.4, one approach to setting numeric nutrient criteria would be to identify a single nutrient concentration (e.g., total P) that is protective of beneficial uses and apply that concentration as a uniform criterion across the state. Such a single state-wide numeric criterion approach would be, however, deficient for several reasons.

The natural sources of N and P in surface water are mainly geology, soils, and vegetation[35,63]. Climactic conditions, and other regional variables, may affect the rate at which N and P are released to the surface waters from these sources. But different regions of the state are endowed with different types of soil, different kinds of underlying geology and experience differing climates. As a consequence, nutrient concentrations in rivers and streams are expected to show a natural variability across the state even in the absence of impairment from human activities. It would be entirely natural for some regions to manifest higher background levels of nutrient concentrations than other regions. A single, state-wide criterion therefore has the serious disadvantage that it may either (1) require the attainment of nutrient levels that are below the natural background level for a region (imposing unnecessary and unrealistic attainment costs on local communities) or (2) allow the build-up of nutrient concentrations that are above an acceptable background level for that region (possibly leading to the problems associated with eutrophication described in Section 2.2).

Furthermore, the association between nutrient concentrations and beneficial uses involves regional factors. Owing to the regional variability of plant and animal species and their differing ability to adapt to environmental conditions, a nutrient concentration that does not compromise beneficial uses in one region may indeed affect beneficial uses significantly in another. Therefore, the very process of determining a single concentration that does not compromise beneficial uses may be misguided and inaccurate if applied blindly from one region to another.

For these reasons, a more scientifically accurate approach to setting nutrient criteria would require partitioning the state into zones that share common characteristics such as comparable background nutrient concentrations in their surface waters. One way to do this would be to divide the state into areas which share the same basic soil, geology, vegetation, climate, and regional topographical features. In each of these zones, a single criterion for each nutrient can be reasonably applied because it is reasonable to conclude that the background concentration within each zone is similar, and that the beneficial uses, expected relationships between causal (nutrient) and response variables (e.g., benthic algae growth), and effect thresholds would be comparable.

4.2 Conceptual Approaches for Developing a Geographic Stratification System

There are potentially an infinite number of ways in which the state could be divided, or stratified, into nutrient zones. The most effective stratification methodology, however, would be the one that maximizes the difference in concentration between zones and minimizes the variance within zones. The specific statistical tests which can be used to measure the performance of a proposed stratification methodology are described in Section 4.5.

DEQ began by considering three conceptual approaches for determining the nutrient zones:

- A purely empirical approach based on iterative random delineation of geographical strata followed by statistical analysis of stream monitoring data;
- An empirical approach based on statistical methods known as factor analysis which uses stream monitoring data together with a host of regional environmental variables to create geographic strata; and
- A combined approach based on extant (i.e., existing) geographic classification systems which are subsequently verified and refined by the analysis of empirical data from within the proposed extant zones.

These approaches require the availability of a database of observed nutrient concentrations evenly sampled from all areas of the state from streams classified as "reference" or "background" streams. As will be discussed in greater detail in Section 6.2, reference sites represent our best approximation of stream condition in the absence of noteworthy human disturbance or alternation, although they are not all pristine.

The data requirements of the first two approaches are likely to be much greater than the third approach. The potential approaches to delineating nutrient zones are discussed in further detail in Sections 4.2.1, 4.2.2 and 4.2.3 below. The preferred approach is identified in Section 4.2.4.

4.2.1 The Random Delineation Approach to Establishing Regional Nutrient Zones

This approach is based on repeatedly dividing the state into random nutrient zones and then testing to determine which of the iterations maximize inter-zone nutrient concentration differences and minimize intra-zone differences. To use an analogy, the random delineation of nutrient zones may be thought of as equivalent to splashing several different colors of paint onto a map of the state, assuring that none of the color blobs overlap. Each color would represent a nutrient zone. This process is repeated a large number of times. The map resulting at the end of each iteration would represent one possible classification scheme for nutrient zones. Each resulting map would have to be evaluated using the available database to assess its performance as a stratification methodology. The map in which nutrient zones are most effective at maximizing inter-zone differences and minimizing intra-zone differences would be selected as the best scheme for delineating nutrient zones.

The most significant advantage of this empirical approach is that it has the potential to generate a nutrient zone classification system which performs better at maximizing inter-zone variance and minimizing intra-zone variance than a system generated by any alternative method for a given dataset. In other words, through the sheer power of number-crunching this technique can come up with the best classification scheme for nutrient zones as measured by statistical performance metrics.

The empirical approach based on random delineation has several drawbacks. Most importantly, it does not take advantage of any established theories of water quality. The exclusive focus on numerical analysis means that the results will only be as good as the available database — this could result in high levels of uncertainty and misleading results. Another disadvantage is that the empirical approach is computationally intensive. Also, the size and shapes of the resulting zones could be impractical to implement.

4.2.2 The Factor Analysis Approach to Establishing Regional Nutrient Zones

The factor analysis approach attempts to identify the regional variables which best explain the variance in background nutrient concentrations using complex statistical methods broadly known as factor analysis. Factor analysis is a term encompassing a family of statistical techniques which reduce multiple variables (such as geology, climate, etc) into smaller groups to minimize the variance of the dependent variable (in this case nutrient concentrations) within these groups. In the context of nutrient concentrations, the groups resulting from a factor analysis can effectively be thought of as a basis for defining nutrient zones. Unlike an extant classification system (described next), factor analysis combines regional variables without reference to theoretical concepts from water quality science.

The advantage of this approach is that it is less computationally intensive than the empirical random delineation method described above. A second advantage is that it seeks to determine associations and correlations between the dependent and independent variables without reference to any existing classification system; it is possible that this approach could therefore reveal associations overlooked by standard nutrient theory.

The disadvantage of the factor analysis approach is that is will only be as good as the database upon which it operates. Errors and imbalances in the database will result in inaccurate nutrient zones and high levels of uncertainty. The methods of factor analysis are also statistically complicated and would require a high input of analytic effort. Nutrient zones based on factor analysis may be difficult to communicate to the public and complicated to implement.

4.2.3 The Combined Approach to Establishing Regional Nutrient Zones

The combined approach to establishing nutrient zones endeavors to take advantage of theoretical knowledge of water quality science as well as empirical data on nutrient concentrations. As described above, theoretical considerations lead to the expectation that areas with similar soils, geology, climate and other regional environmental factors are likely to have similar background nutrient concentrations in their surface waters. In the combination approach to establishing nutrient zones, several potential extant classification systems are proposed. The extant classifications are selected based on their likelihood of properly reflecting the natural variability of stream nutrients concentrations. These alternative extant classification systems are then evaluated based on the available empirical evidence to determine which performs best in terms of maximizing inter-zone variance and minimizing intra-zone variance. Thus, the combination approach is a dual approach comprising two distinct steps:

Step 1: Propose alternative extant classification schemes that, based on theoretical considerations from water quality science, are likely to be good candidates for segregating stream nutrient concentrations.

Step 2: Evaluate the performance of the proposed classification schemes using empirical data to determine which scheme is best able to maximize inter-zone variance and minimize intra-zone variance; statistical tests will also ascertain whether the proposed scheme is statistically significant.

The primary advantage of the combined approach is that it seeks to balance theoretical knowledge from water quality science with empirical field observations in the delineation of nutrient zones. The proposed extant classification schemes are expected to stratify the state into zones with similar background nutrient concentrations in surface waters based on well established theoretical principles of science. The empirical analysis which follows will then confirm whether the proposed schemes do achieve this objective and will identify the best performing stratification methodology amongst the options analyzed. Thus, the approach is not overly subject to uncertainty and error resulting from errors or imbalances in the available water quality database.

A potential disadvantage of the combination approach is that the final result for a stratification methodology will only be as good as the proposed alternatives. That is, the approach will choose the best of a given set of alternatives. It is possible that the best possible approach may not have been included amongst the set of extant classification systems tested.

4.2.4 DEQ's Preferred Approach

Based on the advantages and disadvantages described above, DEQ decided against pursuing either the random delineation or the factor analysis for determining nutrient zones because of the high data requirements, statistical uncertainties, computational intensity, and analytic complexity. DEQ instead favored the combined approach (Section 4.2.3), which builds on theoretical concepts of water quality science using rigorous empirical analysis.

4.3 Extant Classification Systems

The first step in using the combined approach is to identify potential extant classification systems for nutrient zones. When choosing amongst potential extant classification systems, one of DEQ's choices was Omernik ecoregions[63]. Designed to serve as a spatial framework for environmental resource management, ecoregions denote areas within which ecosystems (and the type, quality, and quantity of environmental resources) are generally similar[63,64]. The ecoregion concept is based on the premise that ecologically similar regions can be identified through analysis of the patterns and composition of biotic and abiotic factors that affect or reflect differences in ecosystem quality and integrity. These factors include geology, physiography, vegetation, climate, soils, land use, wildlife, and hydrology. James Omernik carried out a national eutrophication survey of streams in the 1970s[51] and, following up on that work, developed the first of his ecoregion maps. The stated purpose of the ecoregion maps was to classify streams for more effective water quality management[63]. Thus, from the outset, ecoregions were developed and designed for making decisions about streams and their water quality.

A Roman numeral classification scheme has been adopted for different ecoregion scales. Level I is the coarsest level, dividing North America into 15 ecological regions, whereas at level II the continent is subdivided into 52 classes (*Available on the web at:* http://www.epa.gov/wed/pages/ecoregions.htm). Level III and level IV are the hierarchical levels evaluated as part of this analysis. Montana contains parts of 7 level III ecoregions and also 85 level IV ecoregions (Figure 4.1).

In addition to examining the usefulness of ecoregions as a classification system for establishing nutrient zones, DEQ also evaluated two other extant classification systems; Strahler stream order[4], and lithologic groupings[65] (i.e., underlying geology). Strahler stream order groups streams of a similar dimension and flow, and had the potential to delineate meaningful nutrient zones if in fact nutrients in streams were largely a function of upstream watershed area. Lithology groups land areas with a similar underlying geology, and since the rocks through which streams flow are known to influence their water chemistry (including nutrients, e.g. nitrogen[66]), lithology was considered to be a good candidate for nutrient zones.

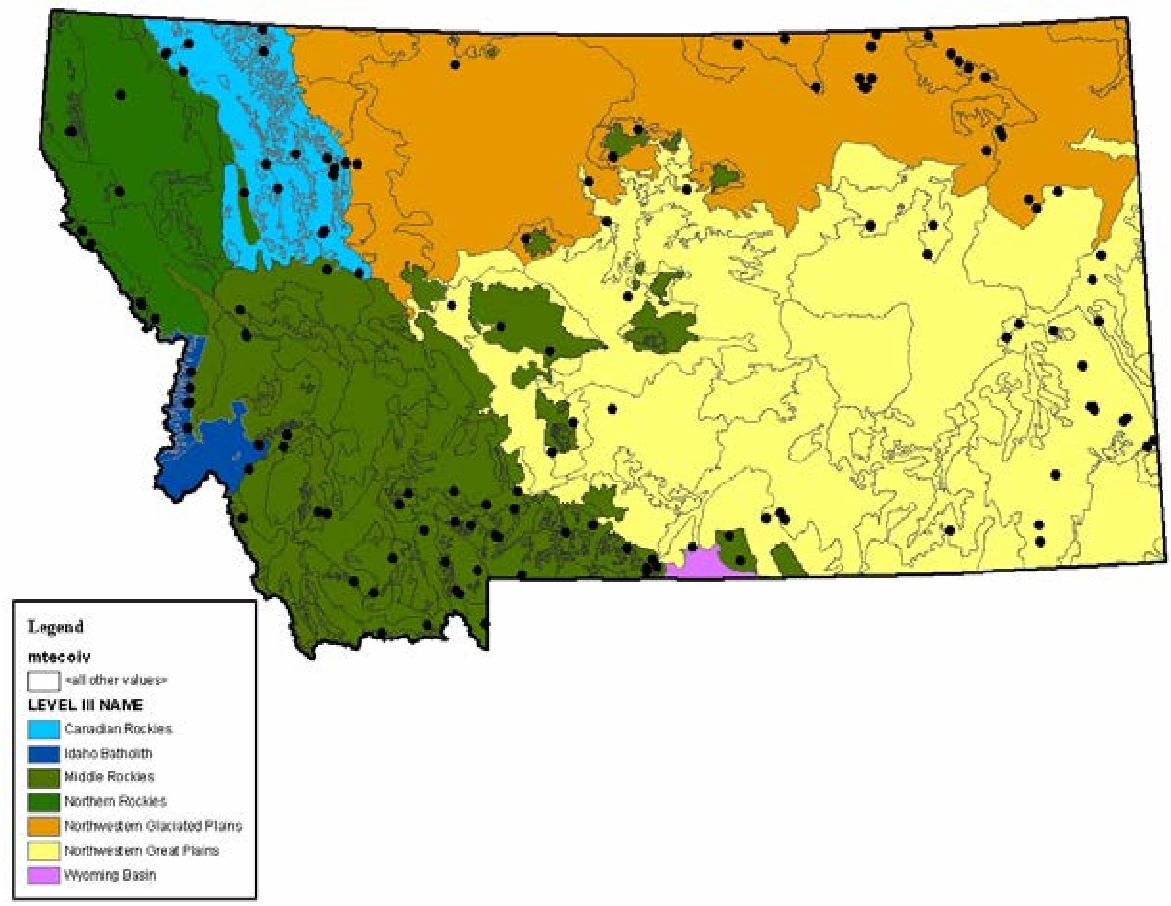

Figure 4.1 Omernik Level III Ecoregions in Montana. Level IV ecoregions are shown as outlined areas within each level III ecoregion. Reference stream sites in Montana through 2007 are identified with black dots.

4.4 Historical Database of Montana Stream Monitoring Data

The extant classification systems proposed in Section 4.3 needed to be verified and supported by statistical tests using empirical data. The main data source for the analyses was from the U.S. EPA's Storage and Retrieval (STORET) database. The database was supplemented with all river and stream nutrient data from DEQ found in modernized STORET, which were collected from 2000 to 2004. Additional data sources included Montana river and stream data collected by the University of Montana, Utah State University, the U.S. EPA's Environmental Monitoring and Assessment Program (EMAP[67]), and reference-stream nutrient data up through 2007. Greater detail on the database and the quality control measures used to assemble it are on page 454-456 of Suplee *et al.*[60]. The database contained approximately 13,000 sampling sites and over 140,000 total records. In the following summaries of data analysis, the term "general population" refers to all observations from both reference and non-reference sites. The term "reference population" refers to nutrient data only from reference sites (reference sites are further discussed in Section 6.2).

4.5 Statistical Methods used to Test the Geographic Nutrient Zones

The extant classification systems proposed in Section 4.3 can be used to generate different combinations of stratifying and sub-stratifying hierarchy for the creation of nutrient zones. For example, one hierarchy could define zones based on each Strahler stream order within level III ecoregions. An alternative potential hierarchy could define zones based on each level IV ecoregion within each level III ecoregion.

With several potential stratification hierarchies, statistical tests are necessary to identify the best-performing hierarchy and to confirm that the proposed hierarchy is indeed statistically significant. As described earlier, the best performing hierarchy is one that stratifies the state into nutrient zones such that the inter-zone variability of nutrient concentrations is maximized and the intra-zone variability of nutrient concentrations is minimized; the hierarchy must also be statistically significant and not just the result of sampling variance.

Non-parametric and parametric statistical methods were both used to examine whether the proposed extant classification hierarchy did indeed result in nutrient zones in which nutrient concentrations were different from one other at an adequate level of statistical significance. A test was considered statistically significant if we could deduce with 95% confidence that the observed differences were indeed reflective of true differences in the data groupings and not the result of sampling variance. Parametric tests were preferred if the distributional requirements of the underlying data were satisfied; the results of parametric tests provided more information than non-parametric tests especially for examining the statistical significance of sub-stratifying methodologies. The non-parametric tests have the advantage, however, of not requiring the underlying data to adhere to any particular statistical distribution. If more than one proposed classification system is found to be statistically significant, it is possible to investigate which classification maximizes inter-zone variability by assessing which classification results in the most nutrient zones in which nutrient concentrations are different from one another at an adequate level of statistical significance. These tests are discussed in Section 4.5.1 and 4.5.2. Statistical measures were also designed to measure the performance of the alternative extant classification methodologies at reducing intra-zone variance. These tests are discussed in Section 4.5.3.

4.5.1 Non-Parametric Tests for Determining Differences Between Stratified Populations

A stratification methodology may be considered statistically significant if there are differences in nutrient concentrations between the zones defined by the methodology, i.e., if at least one zone may be considered to have a higher or lower median concentration than the other zones. In order to test for statistically significant differences between the median nutrient concentrations of different strata within a given stratification hierarchy, we used the non-parametric Kruskal Wallis test. This test is very similar to a one-way ANOVA in which the data are replaced by their ranks. The main advantage of the Kruskal Wallis test is that it does not require the populations to be normally distributed although it does assume that the data in each grouping follow a similarly shaped distribution. The Kruskal Wallis test is an extension of the Mann-Whitney test (also known as the Wilcoxson Rank sum test) to three or more data groupings. The test was used only on the median database. (As described above, the median database is a version of the water quality database in which each sampling station is represented by a single median value for each season for each nutrient grouping. The median database reduces imbalances in sampling frequency and is likely to obey the distributional assumptions of the parametric tests.)

A 95% confidence level was used to identify statistically significant differences. If the test indicated the existence of statistically significant differences in median concentrations between the strata, a *post-hoc* non-parametric multiple comparison test was implemented[68]. These procedures helped determine which strata could be considered different from one another.

4.5.2 Parametric Tests for Determining Differences Between Stratified Populations

We used analysis of variance (ANOVA) to test for statistically significant differences between the mean nutrient concentrations of different strata for a given stratification methodology. Although similar to the non-parametric Kruskal Wallis test, ANOVA offers the substantial advantage of being able to test for the statistical significance of sub-stratifying methodologies. ANOVA procedures are most accurate when the underlying populations are normally distributed with equal variance in each stratum. For our analysis, tests showed that ANOVA results could be considered robust to the observed levels of non-normality and inequality of variance (see page 11, Varghese and Cleland[69]).

ANOVA was implemented only on the median database because this database reduced imbalances in sampling frequency and better obeyed the distributional assumptions of the parametric tests. A 95% confidence level was used to identify statistically significant differences. If the test indicated statistically significant differences in mean concentrations between the strata, a post-hoc parametric multiple comparisons of means was performed using the Bonferroni adjustment. In order to test the statistical validity of sub-stratification, we used a nested ANOVA model with sub-strata nested within the main strata. We then used the Wald test to test the significance of the sub-stratification term in the nested model. The Wald test is a way of testing the significance of particular explanatory variables in a statistical model, including nested variables.

The coefficient of determination, represented as R^2, is the proportion of the total variability in the dependent variable that is accounted for by the model. It is an indicator of the goodness of fit of a model. An $R^2 = 1$ indicates that the model accounts for all the variability of the values of the dependent variables in the sample data. At the other extreme, an $R^2 = 0$ implies that the model explains none of the variability. This measure must be used with caution. Statisticians warn that a high R^2 does not assure a valid relation just as a low R^2 does not mean the model is without value.

The R^2 and adjusted R^2 statistics were computed for all ANOVA runs in our analysis. While indicative of a model's goodness of fit, these measures should not be used alone to select between alternative statistically valid stratification methodologies, because adding variables or sub-strata to a model will always improve the R^2 measure. Instead, once a set of statistically significant stratification methods have been determined, the selection of the optimal method should also consider *a priori* ecological, biological, and hydrogeologic considerations, and practical ease of applicability.

4.5.3 Computation of Measures of Variance for Alternative Stratification Methods

Section 4.5.1 and 4.5.2 discussed tests to ascertain whether statistically significant differences existed between proposed nutrient zones. To assess the performance of alternative stratification methodologies in minimizing variation within nutrient zones, two measures of intra-zone variance were computed: the mean coefficient of variation and the coefficient of efficiency.

For each stratification methodology, the mean coefficient of variation (MCV) was computed as follows, based on a definition provided in Robertson *et al*.[70]:

$$MCV = \sqrt{\frac{\sum (CV^2 \times n)}{N}}$$

$$CV = \frac{StDev}{\overline{X}}$$

where,

CV is the coefficient of variation of each group (or area);
n is the number of observations in each group;
N is the total number of observations in all of the groups;
StDev is the standard deviation of each group; and
$\overline{X}$ is the mean concentration of each group.

A shortcoming of the MCV measure is that it is likely to improve (i.e., show lower absolute values) with increasing stratification. Therefore it would only be appropriate to use the MCV to assess the performance of alternative stratification schemes if the schemes divide the state into roughly equal numbers of strata.

Hydrologists have proposed the coefficient of efficiency as a means of evaluating the goodness-of-fit of hydrologic and hydroclimatic models[71]. This measure is defined as follows:

$$COE = 1 - \frac{\sum_i (O_i - P_i)^2}{\sum_i (O_i - \overline{O})^2}$$

where,
O_i = Value of the ith observation
P_i = Predicted value corresponding to the ith observation (equal to the mean of the observations in the stratum of the ith observation)
$\overline{O}$ = Grand mean of observed values

Thus, the COE in this analysis will equal the ANOVA R^2. This measure can vary from minus infinity (poor model) to 1.0 (perfect model). Like the MCV, the COE has the shortcoming of being likely to improve (increase) with increasing stratification. Therefore it would only be appropriate to use the COE to assess the performance of alternative stratification schemes if the schemes divide the state into roughly equal numbers of strata.

Although the MCV and COE will usually be negatively correlated (i.e., high MCV associated with low COE and vice versa), there may be exceptions to this trend. These exceptions may occur because the MCV is weighted by the number of observations in each group and because the COE is more sensitive to departures from the grand mean.

4.6 Other Issues Affecting the Choice of Nutrient Zones

Although the chosen approach and the supporting statistical analysis may suggest the adoption of a particular system of geographical stratification, there are other pertinent issues affecting the choice of nutrient zones. These issues include the optimal number of nutrient zones, whether specific zones are required for specific nutrients, and whether there is the need for season-specific nutrient zones. These issues are discussed in greater detail in this section.

4.6.1 The Ideal Number of Regional Nutrient Zones

In developing any geographic stratification methodology an important issue that must be addressed is the appropriate scale and therefore number of nutrient zones. The disadvantages of a single zone, or too few zones, have been described. Too many zones are undesirable because they require extensive data; without a sufficient number of empirical observations from each proposed zone, the numeric nutrient criterion for that zone will suffer from a high degree of statistical uncertainty. And large numbers of zones will likely lead to excessive regulatory complexity. Deciding on an appropriate number of zones is a matter of regulatory judgment which must balance the need for accuracy in region-specific nutrient criteria *vs.* issues of sample size and regulatory complexity. One mechanism to reduce the number of nutrient zones in an extant sub-stratification methodology is to single out only those zones that are empirically determined to have different average nutrient concentrations from their parent stratification. For example, a methodology based on substratifying level IV ecoregions within level III ecoregions would require the formation of 34 nutrient zones in the Middle Rockies ecoregion since there are 34 level IVs nested within the Middle Rockies ecoregion in Montana. Multiple comparisons (based on procedures to compare nutrient means or medians, such as the t-test, for instance), however, could be used to compare the concentrations in each level IV ecoregion to the combined concentration in the remaining 33 ecoregions of the Middle Rockies to see if they are significantly different. Such comparisons may reveal that only particular level IV ecoregions have sufficiently distinct concentrations from the average level in the Middle Rockies to warrant separating them out. The analysis just described were carried out, and a number of level IV ecoregions were found to be unique relative to their parent ecoregion[61]. These results will be incorporated with other considerations relating to level IV ecoregions, and discussed again in Section 7.4.

4.6.2 Nutrient-Specific Zones

Nutrient zones applicable for a single nutrient group (e.g., TP) may not match spatially with nutrient zones derived for another nutrient group (e.g., TN). Regulators then have to decide whether they wish to delineate separate nutrient zones for different nutrient groups, as has been proposed elsewhere[70], or use nutrient zones that apply to multiple nutrients. The advantage of nutrient specific zones is that they are more likely to produce the most desirable ecological outcomes. The disadvantage of nutrient specific zones is increased regulatory complexity and potential public resistance.

4.6.3 Seasonal Considerations

Ecoregions are likely to show distinct inter-seasonal nutrient concentrations and, therefore, stratifying the nutrient data by different seasons would further improve the characterization of regional nutrient concentrations. Flowing waters often demonstrate distinct seasonal nutrient concentration patterns[72]. For example, P is frequently associated with total suspended sediment[73] and during spring runoff in streams both TSS and TP can be orders-of-magnitude higher than at other times[74]. Seasonal variation in stream nutrient concentrations is not only influenced by abiotic factors such as runoff patterns, but also by biological uptake and release by organisms such as aquatic plants. Aquatic plant growth — including algal growth — is influenced by (among other things) light availability and temperature, which are themselves climatically driven. Therefore, stratifying the nutrient data seasonally to better characterize nutrient concentrations across time had to consider not only hydrologic patterns, but also climatic factors such as the onset of cold winter temperatures.

Giving consideration to the factors above, an analysis was completed and three seasons were defined for each ecoregion: a growing season, which would roughly correspond to the summer months; winter, which would follow the growing season; and runoff, which would terminate the winter period and comprise the yearly high flow period[60]. Some minor changes to the start and end dates were carried out since the publication of the aforementioned work, and these are shown in Table 4.1. Table 4.1 provides the current recommendations for start and end dates for the winter, runoff and growing season for each level III ecoregion.

Table 4.1 Start and Ending Dates for Three Seasons (Winter, Runoff and Growing), by Level III Ecoregion.

Ecoregion Name	Start of Winter	End of Winter	Start of Runoff	End of Runoff	Start of Growing Season	End of Growing Season
Canadian Rockies	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Northern Rockies	Oct.1	March 31	April 1	June 30	July 1	Sept. 30
Idaho Batholith	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Middle Rockies	Oct.1	April 14	April 15	June 30	July 1	Sept. 30
Northwestern Glaciated Plains	Oct.1	March 14	March 15	June 15	June 16	Sept. 30
Northwestern Great Plains	Oct.1	Feb. 29	March 1	June 30	July 1	Sept. 30
Wyoming Basin	Oct.1	April 14	April 15	June 30	July 1	Sept. 30

4.7 Results of the Empirical Analysis

As part of the process of creating regional nutrient zones, the proposed extant stratification systems were subjected to various statistical tests (described in Section 4.5) using the historical nutrient database of stream monitoring data. The results of these tests are summarized below:

- Of the various coarse scale stratification systems tested (level III ecoregions, Strahler stream order, and geology), level III ecoregions produced strata that differed significantly from one another in terms of their median nutrient concentrations, for the reference data, for all nutrient groups (TKN, TN, NO_{2+3}, SRP, TP) except total dissolved P (TDP), which has a very small dataset. The efficacy of stratification by level III ecoregions is most apparent in the growing season, and for year-round data.

- Post-ANOVA Wald tests were used to verify the statistical significance of various sub-stratification methodologies of the coarse-scale strata, for the reference population, on a limited selection of nutrients. For year-round data, sub-stratification by level IV ecoregions was consistently an improvement over stratification by level III ecoregions. The other sub-stratification methods did not show statistically significant results. However, sample size was limited at this level of stratification for the reference population and the power of these tests is likely to be low. (Low statistical power results in an inability to declare a stratifying parameter as significant even though in reality it may be significant. Low power can result from inadequate sample size.)

- Analysis of the measures of intra-zone variance (MCV and COE) indicates that the statistically significant stratification methodologies are more successful in explaining variance for nitrogen-group nutrients than for phosphorus groups. The stratification methodologies have the most explanatory power in the winter season. The growing season appears to be the most noisy.

- The measures of intra-zone variance indicate a considerable improvement in the measures of variance with increasing sub-stratification. However, as explained earlier, this improvement may partly be the result of a fewer number of observations contributing to each stratum and should be considered with caution.

- When sub-stratifying level III ecoregions by level IV ecoregions, multiple comparison tests indicate that only certain level IV ecoregions are statistically different from their parent level III ecoregions. It is thus possible to reduce the overall number of nutrient zones by creating nutrient zones only for those level IV ecoregions that are in fact distinct from their parent level III ecoregion.

- Seasonal analysis of the stream data indicate that seasonal differences in background nutrient concentrations are significant when seasons are defined per the methodology outlined in Section 4.6.3.

If the reader would like further detail on these topics, further explanation is available in both documents by Varghese and Cleland[61,69].

4.8 Conclusions about the Geographic Stratification Systems

Based on the results described above, the following conclusions were drawn about geographically stratified nutrient zones in Montana:

- Level III ecoregions and level IV ecoregions constitute statistically significant systems for stratifying the state for most nutrients in most seasons for both the general- and reference-data population.

- Nutrient zones based on sub-stratifying level III ecoregions by level IV ecoregions may be regarded as the best performing stratifying methodology examined in our analysis, based on tests of statistical significance, measures of variation, and *a priori* theoretical considerations (i.e., the underlying theoretical basis of the maps themselves).

- Most of the stratifying methodologies considered in this analysis perform better for nitrogen than for phosphorus. It may be possible to generate more complex and efficient stratifying methods specifically for the phosphorus-group nutrients. However, for simplicity, it is advisable to create common nutrient zones for N *and* P together, rather than different zones for each nutrient.

- When using level IV ecoregions as a sub-stratifying methodology within level III ecoregions, it is advisable to reduce the overall number of nutrient zones by identifying and creating separate zones only for those level IV ecoregions that are significantly different from their parent level III ecoregion. This will be discussed further in Section 7.4.

Section 5.0
Identifying Eutrophication Impacts on Sensitive Beneficial Uses

One of the most important aspects of setting criteria is determining when beneficial uses begin to become harmed. An example of harm to a beneficial use was given back in Section 1.4. In this section, we will address eutrophication-specific harmful effects on uses.

As noted in Section 2.2, heavy algae growth, especially by filamentous forms, is a very common effect of eutrophication and can be seen every summer in many Montana streams. It has been generally observed that as the level of benthic algae increases in a stream, the suitability of the waterbody for public recreation decreases[2,58,75,76,76]. To verify this observation, in 2006 DEQ and the University of Montana carried out a statistically rigorous statewide public perception survey concerning benthic river and stream algae[77]. (The study will shortly be published in the *Journal of the American Water Resources Association.* A copy of the article can be requested from M. Suplee[c]. A web-available summary is at http://www.umt.edu/watershedclinic/algaesurveypix.htm .) Photographs of varying levels of stream-bottom algae, as seen in typical western Montana gravel-bottomed streams, were shown to Montana citizens and recreators on Montana rivers and streams across the state. Stream algae levels were quantified as chlorophyll *a* (Chl *a*) and ash free dry weight (AFDW) per square meter of stream bottom (this information was not provided to survey participants; just the pictures). Participants were asked how the algae level shown in each photograph would affect their recreational use of the stream or river, whatever that recreation might be (e.g., swimming, fishing, boating, etc.). The results were remarkably clear. 70% or more of the public felt that algae levels less than or equal to 150 mg Chl a/m^2 (≤ 36 g AFDW/m^2) were acceptable for recreation. But then a sharp threshold occurred, and only 30% or less of the public considered algae levels at or above the next level up — 200 mg Chl a /m^2 (95 g AFDW/m^2) — to be acceptable for recreation. The more elevated algae levels in the survey were clearly viewed as undesirable aquatic life by the public majority. And, the sharp change in public majority opinion concerning the acceptability of benthic algae levels can be used to define the threshold where the recreation use becomes harmed. (As a frame of reference for the reader, the algae level in Figure 2.1 is 300 mg Chl a/m^2, whereas the algae level in Figure 7.1 is < 8 mg Chl a/m^2.)

Salmonid fishes are common in most western Montana streams. How does the algae level found to be a recreation impact threshold (150 mg Chl a /m^2) relate to the ecology of these fish? Studies are few, but there is an excellent and applicable study from British Columbia in which N and P are added to a small, low-nutrient river during summer in order to observe the effects on benthic algae and salmonid fish[14]. The study shows that with increased N and P concentrations algae reach maximum average values of 150 mg Chl a /m^2, filamentous algae become much more dominant, and juvenile salmonids show significant weight gain. Fish grew better probably because of increased macroinvertebrate abundance (i.e., more food). Thus, what is a harm threshold for one use (algae level of 150 mg Chl a /m^2; recreation) equates to an enhancement of another use (salmonid fishes). Because fish benefit from some enrichment, does it make sense to allow eutrophication to proceed further, growing larger fish and, consequently, growing more algae? Not really. The initial benefits from nutrient enrichment are subsequently lost when too

[c] E-mail: msuplee@mt.gov Phone: (406) 444-0831

much enrichment occurs (note in Figure 2.2, for example, how the number of salmonid fish per unit stream length declines again where TP > 0.06 mg/L)[28,78]. Furthermore, it does not make sense to allow streams and rivers to be strongly eutrophied in order to grow more and larger fish if that action clearly results in an impact to stream recreation which, of course, Montana's world-renowned fishing is a part of. That is, it might be possible to grow bigger trout at algae levels greater than 150 mg Chl*a* /m^2, but only a minority of the public (30% or less) would be interested in recreating at such streams.

Algae levels held to 150 mg Chl *a* /m^2 or less should also better protect the agriculture use. Less filamentous algae means irrigation systems, which are often operating along eutrophied streams and rivers in the summer in Montana, will become clogged much less often. Keeping these irrigation systems clean of algae is inefficient and costly.

The public perception survey did not directly address eastern Montana prairie streams which are quite different in appearance from their western counterparts. Montana prairie streams often become intermittent, are generally low gradient, typically have mud bottoms and are turbid, frequently have substantial macrophyte populations, and support fishes such as bullhead, walleye, chubs, bass and other fish preferring summer temperature 18^{o} C or greater[41,79]. It is not uncommon in these streams to see macrophytes intermixed with filamentous algae and floating masses of green algae; these types of conditions are common even in prairie streams minimally impacted by people (i.e., prairie reference streams). Because prairie streams are fundamentally different in many ways from western Montana trout streams[41], the results from the public perception algae survey should probably not be directly applied to them. Prairie streams nevertheless have important and sensitive beneficial uses that need to be protected. Prairie streams have a diverse array of fish and aquatic life that can be harmed by eutrophication. Harm-to-sensitive use thresholds for prairie streams should therefore be defined by those existing water quality criteria for DO, pH, and total dissolved gas (TDG) already adopted as standards[8] intended to protect fish and aquatic life. Since these effect criteria have been linked to nutrients in prairie streams (e.g., Appendix A), numeric nutrient criteria can be determined.

Section 6.0
Stressor-Response Studies and Reference Site Data as Complementary Components in Determining Numeric Nutrient Criteria

6.1 Stressor-Response Studies

Stressor-response studies examine the relationship between a variable that has the potential to cause a water quality problem (stressor) and the specific effect that it manifests (response). In this work, the stressors of interest are nutrients and the responses are the measurable impacts, i.e. harm, to a stream or river beneficial use. Several stressor-response studies have already been discussed in Section 2.0. When it comes to developing stream and river numeric nutrient criteria, the most useful studies tend to be those that have been carried out in the field (i.e., not in laboratories). Field-based nutrient stressor-response studies vary in the degree of control the researcher has over the study and can be broadly categorized (from most to least controlled) as: artificial stream studies (e.g.,[62,80]); whole-stream fertilization experiments (e.g.,[13,14]); and “mensurative experiments”[81]. Mensurative experiments or studies are those in which the researcher seeks to define a quantitative relationship between an ecological response variable (e.g., stream trout density) and a gradient of an environmental condition (e.g., total P[28]). All three of these study types were considered when developing Montana’s numeric nutrient criteria and there are many, many such studies that have been carried out worldwide. For developing Montana’s numeric nutrient criteria, however, we focused on studies that could be used to relate stream nutrient concentrations to beneficial uses, and that have some regional relevance[14,28,43,57,78,80,82-84]. The most important of these are shown in Table 6.1. How the studies were used to help with nutrient criteria development will be discussed in Section 6.3.

Table 6.1 Studies Addressing Nutrients and Eutrophication that Were Useful for Developing Montana's Numeric Criteria.

Scientific Study	Nutrient(s) Concerned	Study Summary	Type of Study
Perrin *et al.* (1987)	N, P	Nitrogen and P are added to a small river in British Columbia, Canada, to observe the effect on benthic algal biomass & algal population structure, as well as salmonid fish production.	Whole-Stream Fertilization Study
Welch *et al.* (1989)	P	Results from artifical stream channels are used to help model benthic algal growth in the Spokane River, Washington. Modifications to the model are made to mesh with the river's specific conditions. Emphasis is on soluble P.	Artificial Stream Study, Adapted to a River
Bothwell, M.L. (1989)	P	Artificial channels built alongside and using water from the S. Thompson River in British Columbia, Canada. The channels are dosed with P to determine the effect on peak areal biomass of benthic diatom algae.	Artifical Stream Study
Watson *et al.* (1990)	N, P	Water from the Clark Fork River, Montana is pumped at a location where nutrient concentrations are low (due to the influence of two tributaries) into artificial stream channels. The artificial channels are then dosed with soluble N and P and the benthic algal biomass changes are measured.	Artificial Stream Study
Miltner & Rankin (1998)	N, P	Large scale study of Ohio stream & river sites, ongoing since 1982. Nutrient concentrations correlated and compared to fish and macroinvertebrate biometrics.	Mensurative Study
Chételat *et al.* (1999)	P	Benthic algal biomass and species composition compared to nutrient concentrations in 13 Canadian rivers. Correlations between algae and nutrients given.	Mensurative Study
Sosiak, A. (2002)	N, P	A 16 year study on the Bow River (Alberta, Canada) quantifying the reduction in biomass of benthic algae and aquatic macrophytes resulting from reduced N and P concentrations in the discharge of two municipal wastewater treatment plants.	Mensurative Study
Dodds *et al.* (2006)*	N, P	A large database containing benthic algae and N and P data from hundreds of temperate streams wordwide is used to define regression-equation relationships between total N and P concentrations and benthic algae levels.	Mensurative Study
Wang *et al.* (2007)	N, P	240 wadeable streams in Wisconsin are systematically sampled for N & P, macroinvertebrates, and fish. A series of correlations between the nutrients and the biological assemblages are presented.	Mensurative Study
Suplee *et al.* (2008) Unpublished data, Appendix A, this document	N	Relationships between diatom algae and stream environmental characteristics (including nutrients) are presented for a group of NE Montana prairie streams. Diatom population characteristics (metrics) are then used to infer DO concentrations in the streams.	Mensurative Study

*This study is an update and correction to two earlier studies. The first study recommended nutrient criteria specifically for the Clark Fork River in the Middle Rockies ecoregion. These studies were: Dodds, W.K., V.H. Smith, and B. Zander, 1997. Developing Nutrient Targets to Control Benthic Chlorophyll Levels in Streams: A Case Study of the Clark Fork River. *Water Research* 31: 1738-1750; and, Dodds, W.K., V.H. Smith, and K. Lohman, 2002. Nitrogen and Phosphorus Relationships to Benthic Algal Biomass in Temperate Streams. *Can. J. Fish. Aquat. Sci.* 59: 865-874.

6.2 Reference Sites

DEQ has been working for nearly 20 years to locate and characterize wadeable streams which have little or no human disturbance. Some work was completed in the early 1990s and involved the collection of water quality and biological data at stream sites considered by regional land managers to be minimally disturbed[85]. Little was done through the remainder of the 1990s, but the work was recommenced in 2000 and continues to this day in an updated guise using better defined and much more rigorous screening methods compared to the earlier undertaking[42]. Over 130 reference stream sites have so far been identified around Montana (see Figure 4.1, page 20). Reference sites represent our best approximation of stream condition in the absence of substantial human disturbance or alternation[86], although they are not all pristine[d]. In the selection of reference sites, human activities are considered an integral part of the landscape as long as those activities do not negatively harm the various uses of the water (drinking, aquatic life, fisheries, recreation, etc.). DEQ assesses each candidate site and those that pass (i.e., are considered final reference sites) are ranked as either tier 1 or 2 in accordance with how well they fit one of the following definitions:

> *Tier 1 — Natural Condition***:** The characteristics of a waterbody that is unaltered from its natural state, or there are no detectable human-caused changes in the completeness of the structure and function of the biotic community and the associated physical, chemical, and habitat conditions. All numeric water quality standards must be met and all beneficial uses must be fully supported unless impacts are clearly linked to a natural source. The natural condition is the highest attainable biological, chemical, physical, and riparian condition for waterbodies.
>
> *Tier 2 — Minimally Impacted Condition*: The characteristics of a waterbody in which human activities have made small changes that do not affect the completeness of the biotic community structure and function and the associated physical, chemical, and habitat conditions, and all numeric water quality standards are met and all beneficial uses are fully supported unless measured impacts are clearly linked to a natural source. Minimally impacted conditions can be used to describe attainable biological, chemical, physical, and riparian habitat conditions for waterbodies with similar watershed characteristics within similar geographic regions and represent the waterbody's best potential condition.

Montana reference sites represent an array of stream sizes, having Strahler stream orders[4] that range from 1st through 6th (Figure 6.1); most are 3rd order. They occur in all of Montana's level III ecoregions, except for in the Wyoming Basin which has only a small extent in SE Montana (Figure 4.1).

[d] Pristine is, in and of itself, a difficult concept to pin down, given the ubiquitous activities of man both modern and ancient. It is beyond the purpose of this document to address the range of thinking associated with this concept. However, related definitions are presented by Suplee *et al*.[42].

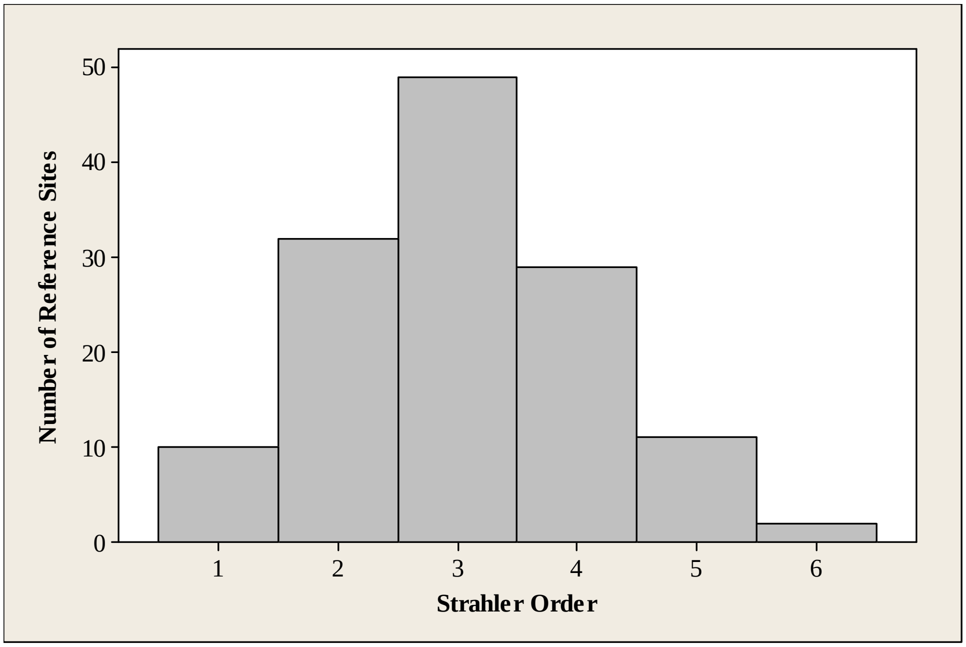

Figure 6.1 Distribution of Strahler Stream Orders for Montana Reference Sites.

6.2.1 The Reference Site Nutrient Database

DEQ has assembled a database containing all nutrient data collected from Montana reference sites. This database has been rigorously screened to assure data quality[60]. It was noted during an early phase of the database's development that some sites among the network of reference sites contributed — for a variety of reasons — a disproportionate amount of nutrient data to the whole than did others. For example, one site may have been sampled 20 times while other sites were only sampled once or twice. This usually occurred because a few sites had a long history of nutrient sampling, while others had only been identified and sampled in recent years. Equitable representativeness among the sites was important for proper reference characterization of each ecoregion. So, in 2007, reference stream sites were sampled in a targeted manner for a suite of nutrients (TN, TP, TKN, NO_{2+3}, SRP, and ammonia) with the intent of making each site a significant contributor to the aggregate nutrient dataset. We used the Brillouin evenness index (J)[87], calculated on an ecoregion-by-ecoregion basis, to measure our success. Very uneven datasets have J values near zero (e.g., 0.2), while a dataset with a J value of 1.0 means each site contributes exactly the same number of samples to the total[87]. Our goal was to achieve index values of ≥ 0.8 (80% evenness) for each level III ecoregion. The work was very successful (Table 6.2), and gives DEQ confidence that the 2008 database has good dispersion of sampling effort among the reference sites and good overall representation of the range of nutrient concentrations found across all reference sites.

Table 6.2 Nutrient Sampling Indices for Montana Reference Sites, Before & After 2007 Sampling. Data are Presented by Level III Ecoregion.

		Status as of November 2006				Subsequent to 2007 Sampling			
Ecoregion (Level III)	Season*	Brillouin Evenness Index (J)	Proportion of Sites Providing Zero Samples	Number Reference Sites	Total Nutrient Samples	Brillouin Evenness Index (J)	Proportion of Sites Providing Zero Samples	Number Reference Sites	Total Nutrient Samples
Middle Rockies	Growing	0.65	14%	42	693	0.81	2%	42	924
Northern Rockies	Growing	0.62	0%	13	230	0.82	0%	13	332
Canadian Rockies	Growing	0.38	31%	13	165	0.72	15%	13	261
Idaho Batholith	Growing	0.72	0%	2	14	0.92	0%	6	80
NW Glaciated Plains	Growing	0.72	29%	21	351	0.80	14%	21	417
NW Great Plains	Growing	0.72	36%	28	185	0.90	8%	36	500

6.3 Integrating Information from Stressor-Response Studies and Reference Sites

Stressor-response studies provide information on the effect nutrients have in streams. Stream reference sites confer an understanding of what nutrient concentrations are like in the absence of substantial human disturbance. For the purpose of setting criteria over a large and diverse landscape, however, each of these individual pieces of information is, in a sense, incomplete. Stressor-response studies provide the scientific understanding as to how eutrophication is manifested in streams, but each study has its own statistical uncertainties (e.g., these studies often have only a moderate correlation coefficient, or r^2, between nutrient and response variable[88]), they are usually limited in scope (specific to a particular region or individual stream), and there are usually only a few studies available in any given ecoregion. In contrast, reference sites — if there are enough of them — provide good landscape coverage for an array of un-impacted regional stream types, but they do not by themselves tell us about thresholds of harm to beneficial uses. But when these two types of information are brought together, a very powerful tool is created that affords good confidence about steam ecology, eutrophication effects, and when beneficial uses become harmed. This section will demonstrate how reference and stressor-response data were integrated in order to derive Montana's numeric nutrient criteria.

6.3.1 Nutrient Concentrations from Stressor-Response Studies Compared to Concentrations from Reference Sites (*or*, Integrating the Stressor-Response and Reference Approaches)

Figure 6.2 is a conceptual diagram showing how nutrient concentrations from reference sites and nutrient concentrations that harm uses (derived from stressor-response studies) might be related to one another. Because reference streams are, by definition, minimally impacted by people and support their beneficial water uses, it is logical that the majority of the nutrient concentration data collected from reference sites would be acceptable[58]. Intuitively, nutrient concentrations that harm stream uses should only be among the highest concentrations observed at reference sites or, perhaps, be even higher than any concentration observed in reference sites (Figure 6.2). By comparing harm-to-use nutrient concentrations derived from stressor-response studies to applicable reference site distributions, one can gain more confidence that the stressor-response study values are correct. For example, if a particular stressor-response study suggests that a total

P concentration of 1.0 µg/L is needed to protect beneficial stream uses, but that concentration falls at the 25th percentile of the applicable distribution of reference sites (i.e., on the far left side of the histogram in Figure 6.2), it would lead one to suspect that either there was a problem with the stressor-response study results or the quality of the reference sites. In this manner, stressor-response results and reference data are used to complement and cross-check one another.

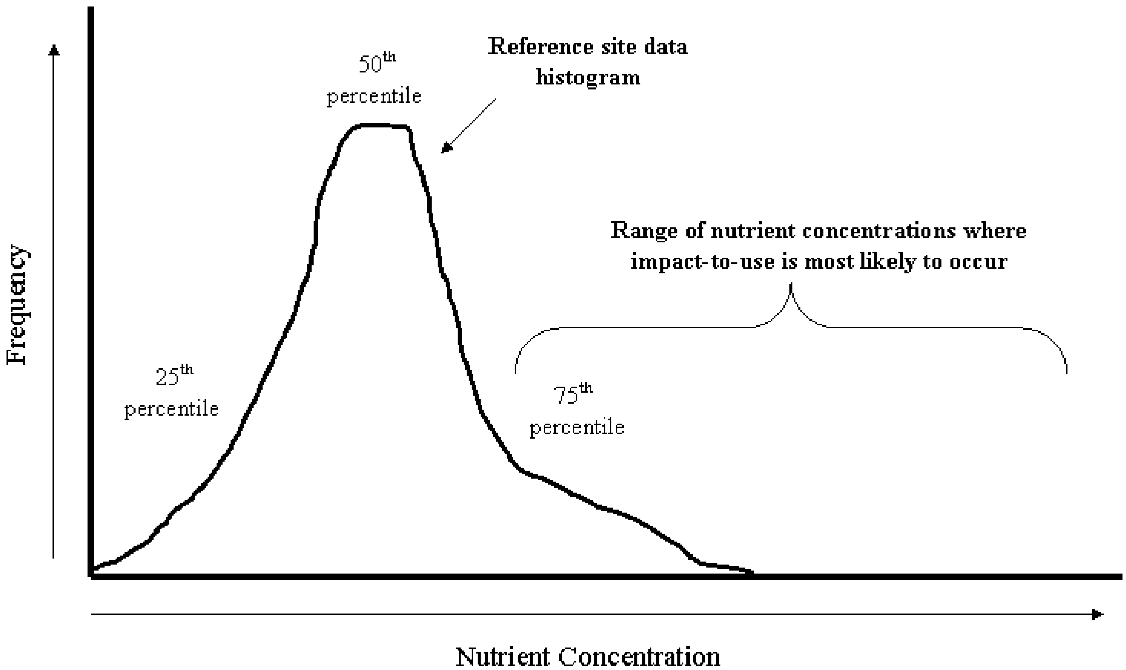

Figure 6.2 Conceptual Diagram Showing a Nutrient Concentration Histogram for Reference Sites. The figure shows where along the x-axis, relative to the reference-data histogram, nutrient concentrations likely to harm beneficial water uses would be expected to be found.

The cross-comparison analysis described above was undertaken several years ago for Montana and showed that, on average, nutrient concentrations at the 86th percentile of the reference distribution were equivalent to harm-to-beneficial use thresholds[60]. Referring again to Figure 6.2, empirical data analysis indicates that among reference sites harm-to-use nutrient concentrations are not greater than (beyond) the aggregate reference-site distribution, but instead are among the very highest concentrations measured in reference sites. It is not surprising that some reference sites have some nutrient samples whose concentrations are higher than the harm-to-use threshold identified using stressor-response studies. In any population of data there are always low and high values that differ considerably from the population's central tendency; the important point is that nutrient concentrations in reference sites that are greater than the harm-to-use threshold occur infrequently, e.g. due to an atypical high-flow event in summer. It is when the harm-to-use concentrations occur *commonly* in a stream that eutrophication problems occur (e.g., see Section 4.2.3 in Appendix H of Varghese and Cleland[61]).

Since the work described above[60] was published, DEQ has made improvements to the reference site nutrient database (see Section 6.2.1), there has been a new stressor-response study completed

(Appendix A), and our understanding of harm-to-use thresholds is improved due to the nuisance algae public perception study[77]; these changes warrant another iteration of the analysis. Table 6.3 shows key descriptive statistics for the current reference-site data (2008 database). Table 6.4 shows the relationship between regional stressor-response studies and regional reference datasets. In Table 6.4, for each stressor-response study, the harm-to-use nutrient concentration threshold was derived as (1) that concentration that would maintain benthic algae levels ≤ 150 mg Chl a/m^2 or (2) that nutrient concentration that would maintain DO concentrations at state standards. Each of the stressor-response studies shown was carried out in and specific to an ecoregion that occurs in Montana. The nutrient concentration at the harm-to-use threshold from each study was matched to the equivalent concentration in its corresponding reference-site nutrient distribution. By "corresponding" reference site nutrient distribution, we mean all data in our database from reference sites located in the same ecoregion where the stressor-response study took place, collected during the same time of year (growing season, i.e., summertime).

Table 6.3 Descriptive Statistics for Nutrients from Montana Reference Sites, 2008 Database. Data are Presented by Level III Ecoregion.

	Level III Ecoregion					
Descriptive Statistics, by Nutrient	Northern Rockies	Canadian Rockies	Middle Rockies	Idaho Batholith	Northwestern Glaciated Plains	Northwestern Great Plains
TOTAL P						
n	88	68	182	15	96	107
minimum (mg/L)	0.001	<0.001	0.001	0.001	0.005	0.001
25th percentile (mg/L)	0.003	0.001	0.007	0.005	0.039	0.030
50th percentile (mg/L)	0.003	0.002	0.010	0.006	0.070	0.069
75th percentile (mg/L)	0.007	0.004	0.018	0.009	0.123	0.124
maximum (mg/L)	0.018	0.035	0.840	0.011	1.350	9.911
TOTAL N						
n	38	20	74	15	59	85
minimum (mg/L)	0.012	0.005	0.018	0.027	0.107	0.050
25th percentile (mg/L)	0.031	0.050	0.070	0.045	0.635	0.442
50th percentile (mg/L)	0.050	0.085	0.110	0.053	0.790	0.735
75th percentile (mg/L)	0.116	0.174	0.179	0.101	1.311	1.358
maximum (mg/L)	0.360	0.235	9.580	0.225	6.303	9.900
NO_{2+3}						
n	89	69	188	15	81	104
minimum (mg/L)	<0.001	0.001	0.001	0.001	<0.001	<0.001
25th percentile (mg/L)	0.005	0.004	0.005	0.014	0.001	0.001
50th percentile (mg/L)	0.009	0.006	0.010	0.020	0.005	0.007
75th percentile (mg/L)	0.011	0.010	0.033	0.038	0.020	0.076
maximum (mg/L)	0.150	0.150	0.350	0.077	4.610	1.864

Note: ***n*** represents all samples from all reference sites combined together.

Table 6.4 Regional Stressor-Response Study Nutrient Concentrations and their Corresponding Percentile Values in Corresponding Reference-Site Nutrient Frequency Distributions.

				Reference Stream Sites				
Stressor-response Study	Nutrient	Notes on Study	Stressor-response Study Nutrient Concentration (mg/L)	Season of Application*	Level III Ecoregion	# Samples in Distribution During Growing Season	Percentile in Reference Distribution Matching Stressor-response Study Concentration	Benefical Use the Nutrient Concentration Threshold Applies To
Welch *et al*. (1989)	SRP	The SRP concentration would constrain the distance the Spokane River has algal biomass of 150 mg Chl *a*/m^2 to about 16 km.	0.01	Growing	Northern Rockies	75	**94th**	Recreation
Watson *et al.* (1990)	SRP	The SRP concentration corresponding to algal standing crop of 150 mg Chl *a*/m^2.	0.011	Growing	Middle Rockies	211	**87th**	Recreation
Sosiak, A. (2002)	TP	Based on a nutrient *vs.* benthic-algae regression equation, TP concentration would maintain algal standing crop ≤ 150 mg Chl a/m^2 on the Bow River near Calgary, Alberta, Canada†.	0.018	Growing	Canadian Rockies	68	**97th**	Recreation
Suplee, M.W. (2008) Appendix A, this document	TN	TN concentration would prevent dissolved oxygen (DO) from dropping below state standards in prairie streams. Quantitative relationships (correlation, changepoint analysis) between diatom-inferred DO and TN concentrations were used to derive the TN concentration.	1.12	Growing	Northwestern Glaciated Plains	59	**70th**	Fish & Aquatic Life
						Mean:	**87th**	
						Median:	**91st**	
						CV (%):	**14**	

* See Table 4.1, this report, for the start and end dates of the Growing Season.

†The Bow River at Calgary sits downstream of the boundary of the level III ecoregion 'Canadian Rockies'. Stressor-response study TP recommendation was assigned to the Canadian Rockies ecoregion since the majority of the river's drainage area upstream of Calgary is in the Canadian Rockies.

The results in Table 6.4 show that harm-to-use-threshold nutrient concentrations equal concentrations at the 87th (mean) and 91st (median) percentile of reference. If only the mountainous ecoregions are considered (Northern, Canadian, and Middle Rockies), harm-to-use-threshold nutrient concentrations correspond to the 93rd (mean) and 94th (median) percentile of reference, with a low CV of 5.5%. The single study from low-gradient, warm-water prairie streams (Appendix A) has a notably lower reference percentile match (70th of reference) compared to the other, mountainous studies (87th-97th of reference)(Table 6.4). This may have resulted because (1) the prairie stream study is looking at a different cause of harm (minimum DO *vs.* nuisance benthic algae levels) than the other studies, (2) the empirical relationship between reference site nutrient data and stressor-response derived nutrient concentrations is inherently different in prairie streams, (3) all the reference sites in the prairie ecoregions fit the Tier 2 definition (some human impacts noted) whereas in the mountain ecoregions there is a mix of Tier 1 and Tier 2 sites, or (4) this is an unusually low reference-to-stressor response match, something of an outlier, and other studies we might carry out in prairie streams in the future may show results more like the other studies in Table 6.4.

Overall, nutrient concentrations suggested by the individual stressor-response studies become more convincing when they are viewed through their collective relationship to reference. It is very unlikely that the pattern seen in Table 6.4, wherein nutrient concentrations from four regionally-applicable stressor-response studies are clustered among the upper percentiles of four different reference distributions from four different ecoregions, would occur by chance[e]. Thus, it can be reasonably concluded that (1) stressor-response derived harm-to-use nutrient concentrations are consistently found among the upper percentiles of applicable reference site distributions[60], and (2) concentrations in the upper percentiles of reference site nutrient distributions can act as surrogates for harm-to-use concentrations. On a statewide basis, the data suggest the 90th percentile of reference (midrange between the mean and median of the case study-to-reference matches) is a good starting point for determining appropriate statewide criteria.

6.3.2 Other Information Reviewed to Cross-check the Criteria

The criteria were cross-checked using the Redfield ratio[89,90], which has long been used to assess which nutrient is likely to limit algal growth. Optimal nutrient ratios (by weight) in benthic stream algae are about 54:8:1 (carbon:nitrogen:phosphorus)[91,92], similar to the widely-accepted phytoplankton algal ratio of 47:7:1. “Optimal” means that, from an alga’s perspective, all three macronutrients are sufficiently available in the environment to allow maximum growth (i.e., none of the elements is in short supply). Although benthic stream algae have optimal Redfield N:P ratios of about 8, N:P ratios ranging from 6 to 10 mean neither element is strongly limiting[92].

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[e] A simple way to calculate the probability of this outcome is by considering each case study as an independent event. If it is assumed that the probability that the nutrient concentration derived from any given stressor-response study has, by random chance, a 0.75 probability of falling somewhere between the 1st and 75th percentile of the corresponding reference distribution, and a 0.25 probability of falling between the 76th and 100th percentile, then the likelihood of the pattern seen in Table 6.4 becomes $0.25 \cdot 0.25 \cdot 0.25 \cdot 0.75 = 0.01$ (1% chance).

Ideally, nutrient criteria should be set so that N:P concentration ratios are near optimal or, if one nutrient tends to be limiting in a region, the criteria should lean towards controlling the limiting nutrient. To accomplish the latter, the N:P ratio should generally be higher than Redfield ratio to control P and lower than Redfield to control N. We examined the N:P ratio of nutrient concentrations at the 90th percentile of reference to see if we would maintain appropriate nutrient ratios, relative to the Redfield ratio. We constrained this analysis to the four mountainous level III ecoregions (Northern, Canadian, and Middle Rockies, and the Idaho Batholith) where control of nuisance benthic algae is the goal. In these ecoregions, TN: TP ratios of the concentrations at the 90th percentile of reference were either within the optimal range (between 6 and 10) or were high, meaning the criteria would tend to control algae via P limitation. The high ratios occurred in the Northern Rockies, Canadian Rockies, and Idaho Batholith, where the TN:TP ratios ranged from 12-35. For these ecoregions this situation is acceptable, even desirable, as several regional studies show that P limitation is common here and benthic algae respond quickly to small increases in P[43,80,82,83].

Another check of the criteria was made by assessing whether or not total P concentrations at the 90th of reference would maintain soluble P concentrations low enough to assure an effect on algae. Studies, many carried out in this region, show soluble reactive phosphate (SRP) should be kept under 5 µg/L[23,43,82,83], no greater than 11 µg/L[80], or perhaps as high as 22 µg/L (if grazers are present)[93] in order to maintain benthic algae levels (including *Cladophora*) at or below 150 mg Chl *a*/m^2 of stream bottom. We again focused on the four western Montana ecoregions (Northern, Canadian, and Middle Rockies, and Idaho Batholith) as their streams most resemble those in the cited studies. SRP:TP ratios in rivers and streams worldwide range from about 0.1 to 0.7[73,74,94,95]. In Montana, long-term monitoring at river and stream sites show SRP:TP ratios typically range from 0.26 to 0.5. We used an SRP:TP ratio of 0.35, which we considered to be a good regional average. Multiplying 0.35 by the ecoregional TP concentration at the 90th percentile of reference (during the growing season, i.e., summertime) resulted in calculated SRP concentrations of: 2 µg/L (Canadian Rockies); 4 µg/L (Idaho Batholith); 4 µg/L (Northern Rockies), and 17 µg/L (Middle Rockies). All fell below the more conservative SRP benchmark (5 µg SRP/L), except the Middle Rockies, which was much higher. Because it seemed elevated, the SRP concentration (17 µg/L) calculated for the Middle Rockies was further evaluated relative to a highly-applicable artificial stream study (the study was carried out *in* the Middle Rockies ecoregion *in* Montana[80]). That study shows 17 µg SRP/L might still keep algae below 150 mg Chl *a*/m^2, as 17 µg SRP/L falls within the 95% confidence interval of the study's benthic chlorophyll measurements[80].

We also reviewed studies that did not occur specifically in a Montana ecoregion but were carried out in northern temperate rivers and streams and provide good comparative information. We compared the TN and TP concentrations at the 90th percentile of reference for the Middle Rockies ecoregion (0.32 mg/L and 0.048 mg/L, respectively) to results from other temperate-stream studies (Table 6.5). These studies occurred streams roughly comparable to those found in the Middle Rockies. (The Middle Rockies ecoregion is the largest ecoregion in western Montana and has more land area than the Northern Rockies, Canadian Rockies, and Idaho Batholith combined.)

Table 6.5 Nutrient Concentrations from Studies Carried out in Northern Temperate Rivers & Streams Compared to Nutrient Concentrations at the 90th Percentile of Reference Sites from Montana's Proportion of the Middle Rockies Ecoregion.

Study	Where Study Took Place	Notes on Study	Nutrient (mg/L) Total N	Total P
This Document	Middle Rockies Ecoregion, Montana	90th percentile of reference	**0.320**	**0.048**
Perrin *et al.* (1987)	British Columbia, Canada	Total N and total P concentrations quantitatively added to a small, low-nutrient river and resulted in peak benthic algae of 150 mg Chl *a* /m^2 and a shift towards dominance by filamentous algae.	0.4	0.02
Miltner & Rankin (1998)	Ohio	Nutrient concentration threshold beyond which deleterious effects on fish communuties are observed.	n/a	0.06
Chételat *et al.* (1999)	Ontario & Quebec, Canada	Benthic algal biomass and nutrient concentrations examined in 13 rivers. Moderately strong relationship (r^2 = 0.56) found between total P and benthic Chl *a* levels. TP concentration shown would maintain algae at 150 mg Chl *a* /m^2.	n/a	0.07
Dodds *et al.* (2006)	Data from North American, Austrailian, New Zealand and European temperate streams	Based on a nutrient *vs.* benthic-algae regression equation, TN and TP concentrations would maintain benthic algae ≤ 150 mg Chl *a* /m^2 (maximum). Concentrations determined using the 9th listed equation of the literature dataset.	0.578	0.080
Wang *et al.* (2007)	Wisconsin	Total N and total P concentration thresholds where the largest change in biometrics occur and beyond which fish and macroinvertebrate assemblages are likely to be degraded.	0.99	0.073

From each of the studies in Table 6.5, nutrient concentrations that represent a biological impact threshold for fish or macroinvertebrates were used, or, alternatively, nutrient concentrations that would result in benthic algae of no more than 150 mg Chl *a* /m^2 (see "Notes on Study" in the table). Overall, the Middle Rockies TP concentration at the 90th of reference (0.048 mg/L) falls within the range of the other studies, while the TN concentration (0.32 mg/L) is at the low end. In all cases, total N or total P concentrations are within the same order of magnitude.

Comparable studies specific to prairie streams are few. The study in Appendix A (this report) suggests 1.12 mg TN/L as a threshold concentration that should prevent nighttime DO minima from dropping below state standards. Other prairie stream studies recommend similar values. One study analyzes a subset of the data in Appendix A (from the Milk/Lower Missouri Basin), and also data from the Sheyenne River basin in North Dakota, and suggests TN should be held to about 1 mg/L[96]. The 1 mg TN/L threshold was determined based on (1) a sharp decline, at 1.03 mg TN/L, in a correlation between TN and pollution-sensitive diatoms[97,98] from the Milk/Lower Missouri basin, and (2) due to the decline in macroinvertebrate EPT taxa with increasing TN concentrations in the Sheyenne River basin. Another study uses a weight-of-evidence approach, compiling literature values, median reference stream concentrations, etc., and recommends 0.96 mg TN/L as the concentration most likely to protect use and integrity of prairie streams[99].

Finally, we considered other states' approaches. The Tennessee Department of Environment and Conservation finds that nutrient concentrations at the 90th percentile of stream reference sites correspond well to harm-to-use thresholds for their wadeable streams. Like DEQ, they also stratify their regional nutrient expectations using ecoregions. Their work shows that once nutrient concentrations exceed the 90th of reference, streams generally show aquatic life impairment based on their macroinvertebrate biointegrity metrics[100].

From this series of comparisons, it can be reasonably concluded that N and P concentrations found among the upper percentiles of reference site nutrient distributions can act as surrogates for harm thresholds of sensitive beneficial uses. As discussed in Section 6.3.1, comparing stressor-response data to their corresponding reference data adds strength to the conclusions drawn from the individual stressor-response studies. Further, using concentrations that are linked to the reference distribution assures that localized, regional landscape effects on background nutrient concentrations will be reflected in the criteria, assuring that the criteria will not be overly stringent or insufficiently protective.

Section 7.0
Criteria Specifications for Montana's Ecoregions

Previous sections detailed how we identified a rational geospatial frame for segregating nutrient concentrations (Section 4.0), how we identified eutrophication's harm to beneficial uses (Section 5.0), and how we zeroed in on appropriate nutrient criteria concentrations (Section 6.0). This section will specify criteria expectations for different regions of Montana, and includes recommended effect variables (e.g., nuisance algae limits) that should accompany the nutrient criteria in some regions.

Table 7.1 below shows the criteria recommendations for wadeable streams of the state, including one example from a level IV ecoregion. Note that benthic algae criteria are also provided; these should accompany the nutrient criteria to assure use protection. The details of how these values were arrived at are discussed in the following sub-sections. As for all water quality criteria, numeric nutrient criteria will undergo periodic revision and update as more stressor-response studies are completed and more reference data is collected.

Table 7.1 Example Criteria for Different Ecoregions in Montana. The Numeric Nutrient Criteria Values May Change Slightly Due to Ongoing Data Collection in Reference Sites.

		Nutrient Criteria				
Ecoregion	Period When Criteria Apply	Reference Percentile Criteria Are Linked to	TP (mg/L)	TN (mg/L)	NO_{2+3} (mg/L)	Benthic Algae Criteria
Level III Ecoregions						
Northern Rockies	July 1 -Sept. 30	90^{th}	0.012	0.233	0.081	150 mg Chl *a* /m^2 (36 g AFDW/m^2)
Canadian Rockies	July 1 -Sept. 30	90^{th}	0.006	0.209	0.020	150 mg Chl *a* /m^2 (36 g AFDW/m^2)
Middle Rockies	July 1 -Sept. 30	90^{th}	0.048	0.320	0.100	150 mg Chl *a* /m^2 (36 g AFDW/m^2)
Idaho Batholith	July 1 -Sept. 30	90^{th}	0.011	0.130	0.049	150 mg Chl *a* /m^2 (36 g AFDW/m^2)
Northwestern Glaciated Plains	June 16-Sept. 30	75^{th}	0.123	1.311	0.020	n/a
Northwestern Great Plains, Wyoming Basin	July 1 -Sept. 30	75^{th}	0.124	1.358	0.076	n/a
Level IV Ecoregions						
Non-calcareous Foothill Grassland *(Parent Level III: Northwestern Great Plains)*	July 1 -Sept. 30	80^{th}	0.040	0.132	Does not pass screening criteria	150 mg Chl *a* /m^2 (36 g AFDW/m^2)

7.1 Streams in Montana's Mountainous Level III Ecoregions

Wadeable streams in the largely mountainous level-III ecoregions Canadian Rockies, Northern Rockies, Middle Rockies, and Idaho Batholith have recreation and fisheries among their beneficial uses. They generally support salmonid fishes, commonly have moderate gradient and gravel bottoms (Figure 7.1), and are used for recreation of all kinds (fishing, swimming, boating, etc.). Preventing harm to the recreation use and the fish and aquatic life use in this region is very important. In Section 6.0 we presented a series of arguments that support the idea that nutrient concentrations at the 90^{th} percentile of reference are, overall, equivalent to harm-to-use thresholds. The 90^{th} percentile is also near the low end of the error bar (CV; ± 5.5%) around the mean reference-to-stressor response match (93^{rd} percentile) specific to the four mountainous ecoregions in Montana (see page 38). Therefore, we recommend that nutrient concentrations in the Canadian Rockies, Northern Rockies, Middle Rockies, and Idaho Batholith be set at the 90^{th} percentile of each ecoregion's reference dataset (see Table 7.1). Nutrient concentrations held to the 90^{th} of reference should assure that benthic algae levels do not exceed 150 mg Chl a/m^2. In addition to the TP, TN, and NO_{2+3} criteria, a benthic algae criterion of 150 mg Chl a $/m^2$ should be adopted for these streams to prevent nuisance growth, per DEQ's algae perception survey[77]. Benthic algae should be measured and reported using DEQ Standard Operating Procedures. All criteria (TP, TN, NO_{2+3}, and benthic algae levels) should apply during the "Growing Season" of each ecoregion (Table 4.1), as that is when the use-impact (nuisance benthic algae) is typically manifested.

Figure 7.1 Willow Creek. A reference stream site in the level III ecoregion Middle Rockies.

7.2 Prairie Streams in Eastern Montana Level III Ecoregions

Wadeable prairie streams are found in eastern Montana's level III ecoregions Northwestern Glaciated Plains and Northwestern Great Plains. Due to its very small size in Montana (Figure 4.1), limited data, and lack of reference sites, the level III ecoregion Wyoming Basin should receive the same criteria as the adjacent Northwestern Great Plains ecoregion. Streams in this overarching region have recreation and fisheries among their beneficial uses. Many become intermittent, are generally low gradient, typically have mud bottoms and are turbid (Figure 7.2), and often have substantial macrophyte populations[41]. The streams are mostly classified as marginal- or non-salmonid fisheries (e.g., C-3 classification; ARM 17.30.629[1]) meaning, in general, they should support warm-water fish and associated aquatic life. It is not uncommon in these streams to see macrophytes intermixed with filamentous algae and floating masses of green algae; these types of conditions are common even in prairie reference streams. Because prairie streams are fundamentally different in many ways from western Montana trout streams[41], the results from the public perception algae survey should probably not be directly applied to them.

The single, applicable stressor-response study we have (Appendix A) shows that a harm-to-use threshold, based on assuring that DO concentrations remain above state minima in order to protect fish and aquatic life, occurs at the 70th percentile of the regional reference TN distribution (Table 6.4). Because the nutrient concentration derived from this stressor-response study has a lower match to reference (70th) compared to the other studies (87th-97th), nutrient concentrations at the 90th of reference are almost certainly too high to protect this region's beneficial uses. For example, TN at the 70th percentile of reference in the Northwestern Glaciated Plains equals 1.12 mg/L, but the TN concentration at the 90th of reference is 1.91 mg/L — in all likelihood much too high to protect fish and aquatic life. Concentrations at the 75th percentile of reference, on the other hand, appear to be appropriate for prairie streams, for three reasons. First, EPA recommends the 75th percentile of reference as generally appropriate for setting numeric nutrient criteria[58]. Second, the TN concentration at the 75th percentile of reference for the Northwestern Glaciated Plains (1.31 mg/L) falls within the 90% confidence interval (0.78-1.48 mg TN/L) around the harm-to-use threshold (1.12 mg TN/L) identified using changepoint analysis[101] in DEQ's study (Appendix A). Third, if one considers the overall statewide pattern, it is clear that other stressor-response studies applicable specifically to Montana indicate higher percentile matches to reference (87th-97th).

Given EPA's general recommendation (the 75th), the statistical uncertainties associated with a single regional study, and the overall statewide stressor response-to-reference pattern, it seems reasonable to use for the prairie streams a somewhat higher value than the 70th. We concluded that nutrient concentrations at the 75th of reference (e.g., TN = 1.311 mg/L, Northwestern Glaciated Plains) should protect fish and aquatic life uses in eastern Montana's prairie streams. The nutrient criteria (TN, TP, and NO_{2+3}) should apply during the Growing Season (Table 4.1) when aquatic plant growth is heaviest and is most likely to impact DO concentrations. One more con-sideration: the other prairie stream studies we reviewed (Section 6.3.2), few that there are, suggest TN should be held to around 1 mg/L, which is lower than our final recommendation. Thus, we also recommend that another prairie stream stressor-response study be carried out in this region.

Figure 7.2 Rock Creek, a Reference Prairie Stream Site in the Level III Ecoregion Northwestern Glaciated Plains.

7.3 Mountain-to-Prairie Transitional Streams

Streams that originate in the mountains and flow out to the eastern prairies of Montana (i.e., east of the Rocky Mountain Front) present a special situation for criteria setting. Some streams originate in mountainous level III ecoregions (e.g., Middle Rockies) and then immediately cross into either the Northwestern Glaciated Plains or Northwestern Great Plains (each level IIIs), but the ecology and water quality of the streams within a mountain-to-prairie transitional area still largely reflect mountain-like conditions (Figures 7.3, 7.4). With distance from their mountain source, the streams gradually become more prairie-like. Level III ecoregions are too coarse to capture the transitional nature of these streams, however specific level IV ecoregions do. Level IV ecoregions that contain mountain-to-prairie transitional stream reaches are: Pryor-Bighorn Foothills; Limy Foothill Grassland; Rocky Mountain Front Foothill Potholes; Non-calcareous Foothill Grassland; and Foothill Grassland[64] (Figure 7.5). Because transitional stream reaches in the level IV ecoregions listed above are highly mountain influenced, criteria should reflect their mountain-like qualities. The algae criterion of 150 mg Chl a/m^2 should apply to streams within these specific level IV ecoregions. At present, nutrient data for these regions are scarce and reference sites are few. DEQ continues to target these regions as a high priority for reference site identification and nutrient sampling. The determination of specific nutrient criteria for these and other level IV ecoregions are discussed next.

Figure 7.3 Rock Creek, a Reference Stream Site in the Transitional Level IV Ecoregion Non-calcareous Foothill Grassland (Part of the Northwestern Great Plains).

Figure 7.4 Sweet Grass Creek, a Reference Stream Site in the Transitional Level IV Ecoregion Non-calcareous Foothill Grassland (Part of the Northwestern Great Plains).

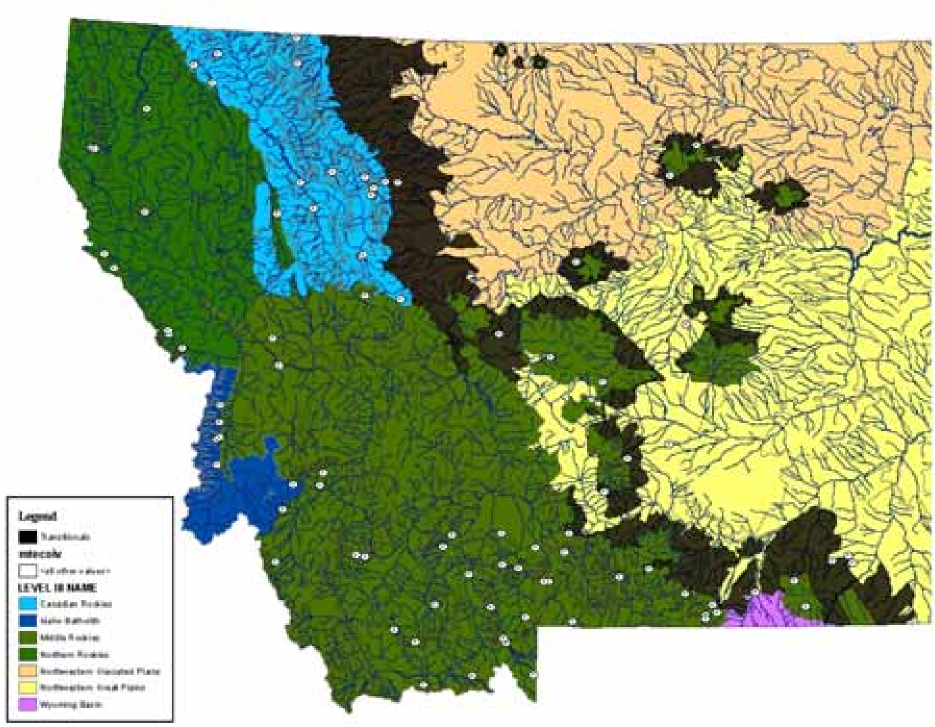

Figure 7.5 Mountain-to-Prairie Transitional Level IV Ecoregions. The level IV transitional ecoregions are shown collectively in dark brown. Reference sites are white dots.

7.4 Identifying Criteria for Unique Level IV Ecoregions

As discussed in Section 4.0, level IV ecoregions have nutrient concentrations that are often demonstrably different from the larger level III ecoregion of which they are a part[61]. When warranted, unique level IV ecoregions will be separated from their level III parent ecoregion. Nutrient concentrations in level IV ecoregions should be set at the 75th or 90th percentile of their particular reference sites depending upon whether the parent level III ecoregion uses the 75th or 90th (see Sections 7.1 and 7.2), *except for* mountain-to-prairie transitional ecoregions (Section 7.3). For the latter, we recommend the 80th of reference as it accounts for these ecoregions' transitional nature between mountain (where the 90th percentile is used) and prairie (where the 75th percentile is used). Criteria for all level IV ecoregions should apply during the Growing Season, using the same dates shown in Table 4.1 for their parent level III ecoregion.

Segregating nutrient criteria at the level IV scale will be done carefully. The much smaller spatial scale of level IV ecoregions leads to issues of small sample size and limited statistical power and, in addition, the stressor-response studies presented in Table 6.4 all occurred at the level III scale. In order for nutrient criteria to be broken out at the level IV scale, the following conditions must be satisfied:

- Within the level IV ecoregion there are at least 12 samples for each nutrient group (12 TP samples, 12 TN samples, 12 NO_{2+3} samples). For details on this sample minimum, see Appendix H in Varghese and Cleland[61].

- There is a statistically significant difference (90% confidence level) between the level IV ecoregion reference-site nutrient data and those of the parent level III ecoregion. (This analysis has already been completed[61].)

- The manner in which the level IV nutrient concentrations are different (higher, lower) compared to the parent level III ecoregion makes sense, given the geology, vegetation, soils, climate, and hydrology of the level IV ecoregion.

- If in doubt, e.g. the nutrient concentration for a particular level IV ecoregion doesn't make sense relative to what would be expected, lump rather than split (i.e., continue to use the level III concentrations rather than split out the level IVs).

- If the level IV ecoregion in question is a mountain-to-prairie transitional one (see list in Section 7.3), the nutrient criteria should be established at the 80th percentile of reference.

An example of a level IV ecoregion meeting the conditions above is the Non-calcareous Foothill Grassland. It is a sub-ecoregion of the Northwestern Great Plains ecoregion. During the growing season (June 16th –September 30th), there are 16 TP samples from reference sites whose concentration at the 80th percentile of reference is 0.04 mg/L (Table 7.1). TP concentrations in the Non-calcareous Foothill Grassland are significantly lower than those of the Northwestern Great Plains[61], which makes sense given this mountain-to-prairie transitional ecoregion's position downstream of the mountains.

In the near future, DEQ will assemble a water quality circular containing all the specific nutrient concentrations and their associated regions and times of application (level III or IV ecoregion, season of application, etc.). In assembling the circular, the level IV ecoregion screening process bulleted above will be carried out. Only level IV ecoregions that meet the specified conditions will have level IV ecoregion criteria in the circular. Note also that not all level IV ecoregions have yet been subjected to the full evaluation process outlined above, therefore only one level IV example was provided in Table 7.1.

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Section 8.0
Review and Recommendations

8.1 Review

Beneficial uses are valuable characteristics of a stream or river resource that, directly or indirectly, contribute to human welfare. Examples of beneficial uses include water supply, fish and aquatic life, and recreation. Beneficial uses are established in law and reflect the societal values embodied in those laws. The intent of water quality criteria, in turn, is to assure a level of water quality that will protect the beneficial uses. Some beneficial uses are more sensitive to harm than others; water quality criteria are required by law to protect the most sensitive use from harm.

DEQ has been carefully developing the nutrient criteria discussed in this document over the past eight years; they are applicable to wadeable streams. They do not apply to large rivers, lakes, or wetlands. The criteria are intended to protect wadeable streams from the detrimental and undesirable effects of eutrophication (nutrient enrichment by nitrogen and phosphorus compounds). Nitrogen and phosphorus concentrations in wadeable streams vary a great deal in accordance with regional geology, soils, climate, and vegetation. To address this fact, DEQ has developed and tested a mapping system that will assure that appropriate nutrient criteria are established in different parts of the state only after taking into consideration natural regional landscape variation.

Fundamentally, the nutrient criteria are based on stressor-response scientific studies in which harm to a beneficial use is shown. All regionally applicable stressor-response studies (nitrogen or phosphorus as stressor, beneficial use impact as response) that could be located were reviewed. The results of these studies were then compared to corresponding Montana reference sites to assure their results made sense (i.e., the concentrations derived from the studies did not seem unrealistic given the nutrient concentrations observed in the reference sites). Further, we were able to establish specific linkages between the stressor-response results and the reference site nutrient-data distributions; these linkages help overcome statistical uncertainties in any given stressor-response study, and assure that natural, regional effects on background nutrient concentrations are reflected in the criteria.

In some parts of the state, mainly in the west, the most sensitive beneficial use is recreation. Public opinion analysis shows that the public majority does not want to see excessive algae growth in the gravel-bottomed, clear running, trout-fishery streams common in western Montana. For these types of streams, nutrient criteria were developed that should prevent nuisance algal levels (as defined by the Montana public perception study) from developing and should, therefore, protect the recreation use. The criteria will also protect the fishery, which typically comprises trout, char, and whitefish, from the negative effects of excessive nutrient enrichment (e.g., low dissolved oxygen concentrations). The criteria should also better protect the agricultural use by reducing elevated algae levels that clog irrigation systems.

In other parts of the state, low gradient prairie streams are common. Wadeable prairie streams in Montana often become intermittent, commonly have mud bottoms, are turbid, frequently have substantial macrophyte populations, usually have filamentous algae but sometimes have only phytoplankton algae, and support catfish, bullhead, walleye, chubs, bass, and other warm water fishes. Because prairie streams are fundamentally different in many ways from western Montana trout streams, the results from the public perception algae survey should probably not be directly applied to them. Prairie streams nevertheless have important and sensitive beneficial uses that need protection, like the diverse species of fish mentioned above. For these types of streams, therefore, the nutrient criteria have been set so that they will maintain dissolved oxygen concentrations that will protect regional fish and aquatic life. These dissolved oxygen concentrations are already established in state law[8]. At this time the most sensitive use in prairie streams is considered to be fish and aquatic life, and the nutrient criteria are set to protect them. But because studies in prairie streams are so few, future prairie stream stressor-response studies (which should be carried out) may lead to revised criteria which could be based on sensitive-use endpoints other than dissolved oxygen.

Some wadeable streams are transitional between the mountainous region, which is found mainly in the western part of the state, and the prairie region, which is found in the eastern part of the state. The mapping system mentioned above accounts for these transitional streams and DEQ will, where warranted, set criteria for this specific group. These transitional streams typically have characteristics in common with mountain streams and therefore nuisance algae levels (as determined by the public perception study) should be controlled in them as well. Thus, the most sensitive use for these transitional streams is recreation.

8.2 Recommendations

- Omernik ecoregions[63] should be used as the basis for applying the criteria across Montana. Ecoregions at the level III scale should at this time be used as the principal means of applying the criteria. However, level IV (fine-scale) ecoregions should be used where warranted. This is particularly true for streams in the mountain-to-prairie transitional areas of the state. Before level IV ecoregions are selected for application of specific nutrient criteria, they should be subjected to the series of screening evaluations in Section 7.4 to assure that the segregation is warranted.

- Criteria should be established for total nitrogen (TN), total phosphorus (TP), and nitrate + nitrite (NO_{2+3}). In some ecoregions (detailed below), stream-bottom algae levels should also be set as criteria based on the results of the benthic algae public-perception survey[77]. Using appropriate statistical evaluation methods and sufficiently-sized datasets (minimum of 12 for each nutrient is suggested), compliance with nutrient criteria should be undertaken using a 20% allowable exceedence rate. Details on these statistical assessment methods are provided in appendix H and I of another document[61]. Stream benthic algae levels should be sampled and analyzed using DEQ Standard Operating Procedures.

- Nutrient criteria should be established as a function of the regionally applicable reference streams. Analysis shows that nutrient concentrations among the upper percentiles of reference-stream nutrient frequency distributions correspond to concentrations that scientific studies show impact water quality and beneficial uses. The advantage of linking the stressor-response derived concentrations to the regional reference distribution is that it helps overcome statistical uncertainties in any given stressor-response study, and assures that inherent, regional landscape effects on background nutrient concentrations are reflected in the criteria. This helps to assure that the criteria are not overly stringent or insufficiently protective.

- Across all parts of the state, the criteria (nutrients and benthic algae levels) should apply during the Growing Season (i.e., generally during the summer). The start and end dates of the Growing Season vary somewhat by ecoregion; see Table 4.1.

- Streams in the mountainous level III ecoregions (Northern Rockies, Canadian Rockies, Middle Rockies, and Idaho Batholith) should have TN, TP, and NO_{2+3} criteria established at the 90th percentile of each ecoregion's reference stream nutrient-concentration distribution. Benthic (i.e., stream-bottom) algae level criteria should be set at ≤ 150 mg Chl *a*/m^2 (≤36 g AFDW/m^2). Level IV ecoregions that are separated out from any of the mountainous level III ecoregions listed here should also have the same benthic algae criteria, and nutrient criteria set at the 90th percentile of the level IV ecoregion's reference-stream nutrient distribution.

- Streams in the level III prairie ecoregions (Northwestern Glaciated Plains, Northwestern Great Plains, and Wyoming Basin) should have TN, TP, and NO_{2+3} criteria established as the 75th percentile of each ecoregion's reference nutrient-concentration distribution. The Wyoming Basin has only a tiny extent in Montana, and has no reference sites, therefore it should have the same criteria as the adjacent Northwestern Great Plains. The criteria should maintain dissolved oxygen levels at state standards. Level IV ecoregions that are separated out from these level III ecoregions should also have nutrient criteria concentrations set at the 75th percentile of the applicable level IV ecoregion's nutrient reference distribution, unless they are a mountain-to-prairie transitional ecoregion (discussed in the next bullet).

- Mountain-to-prairie transitional streams found in the level IV ecoregions Pryor-Bighorn Foothills, Limy Foothill Grassland, Rocky Mountain Front Foothill Potholes, Non-calcareous Foothill Grassland, and Foothill Grassland should have TN, TP, and NO_{2+3} as numeric nutrient criteria. Benthic (i.e., stream-bottom) algae level criteria for these ecoregions should be set at ≤150 mg Chl *a*/m^2 (≤36 g AFDW/m^2). The nutrient criteria should be established as the 80th percentile of the applicable level IV ecoregion reference distribution *ONLY IF* the screening conditions listed in Section 7.4 are met. If not, the nutrient and algae criteria for these level IV ecoregions should continue to be the same as for their parent level III ecoregion.

- DEQ should continue sampling reference streams in order to better characterize regional reference conditions and, in turn, refine the nutrient criteria. In particular, more nutrient samples are needed in the transitional mountain-to-prairie ecoregions. It is also recommended that another stressor-response study be carried out in the prairie regions of Montana, since there is only one completed at this time.

Appendix A

Development of Numeric Nutrient Criteria for Montana Prairie Streams (Northwestern Glaciated Plains Ecoregion)

Abstract

Twenty-four sites on twenty-two prairie streams of the Northwestern Glaciated Plains ecoregion in Montana were sampled from 2001-2004 in order to help establish regional nutrient criteria. Two approaches to deriving nutrient criteria were integrated: the stressor-response approach (nutrient as stressor, impact to beneficial water use as response), and the reference approach. Stream sites manifesting a range of conditions from excellent to poor were chosen to establish a human-disturbance gradient. Short stream reaches were sampled multiple times from late May to late September for a suite of parameters including riparian habitat condition, channel morphology, nutrient concentrations, aquatic plant biomass, and diatom-algae communities. Eight sites were determined to be in a reference condition. Data were used to determine the key parameters affecting the streams' aquatic plant communities (i.e., we identified key ecological drivers). Several diatom metrics were calculated and subjected to five screening conditions before they would be considered for use in nutrient criteria development. These were (1) the metric had to be linkable to a numeric Montana water quality standard, (2) there was a significant correlation between the metric and a nutrient concentration, (3) the metric changed (increased, decreased) in the direction expected *a priori*, (4) reference site data points in the scatterplot between the metric and a nutrient were located in the region of the plot where they were expected, and (5) the metric was fairly insensitive to other important variables measured in the streams. Vulnerability to scouring flows and TSS concentrations were found to be major driving variables in the streams and influenced the aquatic plant communities observed. One diatom metric, the Oxygen Tolerance Index (OTI), met all 5 screening conditions and was used to derive a nutrient criterion. Three key characteristics of the metric were (1) it was insensitive to changes in TSS and measures of stream-scour potential, those variables shown to have great influence in the streams, (2) it did not respond to changes in total benthic plant biomass (benthic algae & macrophytes), or streambed macrophyte cover, and (3) it *was* responsive to nutrient concentration gradients after the elimination of the confounding influence of organic pollution present in some streams. The OTI metric was used to infer the streams' nighttime DO concentrations, which were in turn compared to state minimum DO standards. The OTI was positively correlated to TN concentrations and a significant changepoint ($p = 0.026$) occurred at 1.12 mg TN/L. Streams having TN concentrations below the changepoint were, on average, in compliance with the applicable DO standard (5 mg/L), while streams with concentrations greater than 1.12 mg TN/L were not. All of the reference sites in the diatom OTI *vs*. TN concentration scatterplot were located to the left of the changepoint where one would expect them to be.

Section 1.0
Introduction

The over enrichment of waterbodies by nutrients (usually nitrogen [N] and phosphorus [P]) can increase nuisance algal growth, alter aquatic communities, and result in undesirable water-quality changes that impair beneficial water uses such as fisheries & aquatic life, irrigation, and water-supply[23,32,33,55,102]. The U.S. Environmental Protection Agency (EPA) has published a series of documents containing ecoregion-specific nutrient criteria recommendations that are intended to control nutrient over-enrichment problems in streams and lakes (e.g., recommendations for streams in nutrient ecoregion IV[103]) and, in turn, protect beneficial uses. The EPA has indicated that these criteria are preliminary, however, and work remains to develop more region-specific, scientifically-based nutrient criteria[52,58]. Two approaches recommended by EPA to develop numeric nutrient criteria to protect aquatic systems are the reference approach and the stressor-response approach[58].

The reference approach relies on the identification of relatively undisturbed examples of waterbodies (i.e., reference sites[86,104]), and proper geospatial classification of both the reference and non-reference waterbodies to assure “apple to apple” comparisons[105,106]. This knowledge establishes a baseline against which changes in stream characteristics can be compared[107,108]. More specific definitions such as ‘Minimally Disturbed Condition’, ‘Least Disturbed Condition’, etc. have been proposed to better classify waterbodies categorized as reference[86].

The stressor-response approach comprises two broad categories; studies carried out in laboratories, and those undertaken in the field. The most controlled stressor-response techniques are laboratory concentration-response studies between aquatic organisms and aqueous concentrations of a toxin[109,110], and EPA has well-developed protocols for these methods[111]. But the focus of this discussion is on the other category of stressor-response studies (those using field data), which seeks to define quantitative relationships between an ecological response parameter (e.g., a stream fish population) and a gradient of an environmental condition[112]. In the context of the present study, nutrient concentrations are the environmental condition of interest. This field-based approach has elsewhere been referred to as a mensurative experiment[81]. Algae are commonly used for developing these field-based, stressor-response relationships. For example, regression equations are developed between benthic algal biomass and stream nutrient concentrations[55,57,113]. Diatom algae in particular have been used for a long time as indicators of nutrient/organic enrichment in European rivers[97,114-116], and are also widely used in North America[117-121]. Work in Europe shows that, along a human-caused gradient of stream condition, diatoms have high discriminatory power along the gradient and are more strongly correlated to eutrophication than are macrophytes, macroinvertebrates, or fish[122,123]. Others have developed models relating diatom algae to particular nutrients such as total N and phosphate[120,124,125].

Reference and stressor-response approaches are discussed as separate (though complementary) techniques for developing nutrient criteria in EPA guidance[58]. However, to establish a “gradient of environmental condition”[112], it is usually necessary to locate stream sites from the minimally to the highly impacted ends of an environmental spectrum, and those at the minimally-impacted end often meet at least some general definition of reference; in effect, this integrates the reference and stressor-response approaches. Several studies specifically involving diatom algae integrate reference and stressor-response techniques because “reference” or “minimally impacted” sites are used to establish the high-quality end of the gradient[118,119,123,126]. But it is often the goal of these studies to develop general indices of stream health or biological integrity, and to develop nutrient criteria there still remains the dual tasks of (1) linking the stream-health indices or metrics to specific nutrients and (2) deciding where along the ecological condition gradient an impact to a beneficial water use has occurred. The first task is certainly feasible[28,78,124,125], but the latter can be difficult. If one were to use only the “pristine” (if available) end of the spectrum to establish nutrient criteria, criteria setting would be fairly easy. But setting criteria at pristine raises issues of cost, plausibility and public acceptance, and once one attempts to set criteria at something other than pristine difficult questions arise, such as “how much benthic algae *is* too much?” and “how much change in the macroinvertebrate community *is* acceptable?”. Such questions are not easily answered. Answering these questions requires value judgments as well as scientific understanding, and gets directly at what has been termed “valued ecological attributes”[3], defined as ecosystem characteristics that directly or indirectly contribute to human welfare.

Our purpose in preparing this appendix is to describe how we used water quality tolerances of diatom assemblages to help derive nutrient criteria for a group of prairie streams in Montana. To do this, we first characterized physical, chemical, and habitat parameters of the streams in order to better understand which parameters had the most influence on aquatic plant biomass and diatom assemblages (i.e., what was the basic ecology of the streams). We were then able to establish linkages between a diatom assemblage water-quality index and stream nutrient concentrations. We used information provided by the diatom assemblage to infer stream dissolved oxygen (DO) concentrations, which allowed us to relate the stream nutrient concentrations to established state DO standards. Minimum stream DO requirements in state law (intended to protect aquatic life), reference sites, and sharp changes in the diatom index *vs*. nutrient concentrations were all used together to identify a threshold for harm to uses (i.e., fish and aquatic life). Thus, the stressor-response and reference approaches were integrated, which optimized the results and provided more confident conclusions.

Section 2.0
Methods

2.1 Description of Study Area

The study was carried out in prairie streams of Montana's part of the Northwestern Glaciated Plains ecoregion[63,64] (Figure 2.1), from 2001 to 2004. Prior to European settlement, the region was a semi-arid mixed prairie[127], and is now used mainly for grazing and growing cereal grain crops. In 2001 parts of the region were experiencing extreme summer drought based on the Palmer drought severity index[128] housed at the NCDC (National Climate Data Center, historic monthly Palmer drought severity index records; a*vailable at* http://www.ncdc.noaa.gov). Drought had been ongoing in Montana since 1998. During the 2002-2004 period, the summertime Palmer index returned to normal or near-normal ratings as precipitation improved.

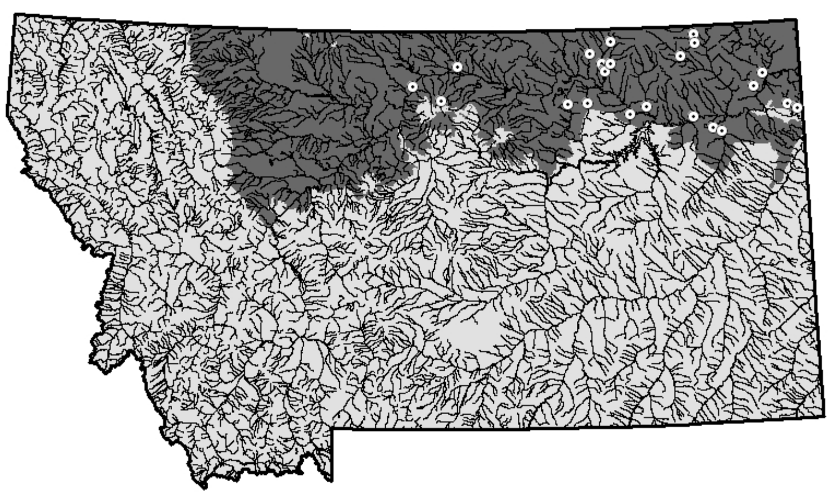

Figure 2.1. Sites Sampled in the Study. The Northwestern Glaciated Plains ecoregion is shaded.

2.2 Study Site Selection

Sites were selected along an estimated "human disturbance" gradient, from best condition (reference) to most impacted. Sampling sites were selected on the basis of land cover information, field observations during site visits, and best professional judgment. In 2001, prior to fieldwork, candidate sites were selected using a Geographic Information System (GIS) assisted approach. Fifth-field hydrologic unit code basins (HUCs[129]) were overlaid in GIS with Multi-Resolution Land Characteristics (MRLC) land-use data and Strahler stream order[4]. The proportion of agricultural land, forested land, urban land, and natural land cover were calculated for the 3rd and 4th order streams. A simple human-impact rating was assigned to each land type; for example, agricultural lands were rated as “high impact”, natural land cover was rated as “low impact”. Candidate stream sites where ranked (best to worst) based on the ratings and the proportion of each land type in their basins. Field reconnaissance in spring 2001 was undertaken to evaluate the overall condition of each candidate site. Best professional judgment was used to make qualitative evaluations of the condition of the riparian plant community, the degree of grazing impact, streambed stability, bank erosion, and human-caused dewatering. Sites to be sampled were chosen only after visiting many candidate sites in the ecoregion. In total, 24 sites on 22 streams were selected.

2.3 Final Site Rankings and Identification of Reference Stream Sites

After the 24 sampling sites were selected, detailed on-site data were collected which better enabled ranking of the sites’ conditions. (A description of on-site data collection procedures is presented later in Methods.) Each site was scored using three different stream condition assessment procedures: MT DEQ’s assessment form (1995-version), EPA’s rapid habitat assessment form[130], and EPA’s Environmental Monitoring and Assessment Program riparian disturbance metric (W1_HALL[131]). These scores were normalized to a 0-100 scale and then averaged (Table 2.1). Each procedure employs semi-quantitative visual estimates of various human impacts at each stream site, estimates that are ultimately converted to a quantitative score.

MT DEQ has screening criteria and procedures for identifying reference sites[42], and these were applied to study sites for the purpose of identifying those in a reference condition. About half of the 24 sites scored low enough on the composite stream-condition assessments discussed above that it was unlikely that they were in a reference condition. Therefore, the reference evaluation process was applied only to streams in the upper half of the score-ranked sites (Table 2.1). Eight sites passed all screening criteria, and are considered reference sites[42] (Table 2.1). The eight reference sites are not pristine, but better fit the Least Disturbed Condition definition[86].

Table 2.1. Site Names, Locations and Stream Condition Assessment Ratings for Each Site in the Study. Riparian Habitat Types are Also Shown. The Process Used to Determine Average Stream Habitat Score is Described in Section 2.3.

Stream Site Name	STORET Station ID*	Lat (dd)	Long (dd)	Mean Site Habitat Score (% of max)	Evaluated as Reference?†	Final Reference Site?	Riparian Type‡
Rock Cr (BLM)	M43ROCKC01	48.6569	-107.0389	93	Yes	**Yes**	Shrub & Herbaceous Riparian Complex
Clear Creek	REFCC	48.3061	-109.4906	90	Yes	**Yes**	Intermittent Riparian Coulee (Yellow Willow/Beaked Sage)
Bitter Cr	M43BITRC01	48.6489	-106.9025	90	Yes	**Yes**	Recent Riparian Complex (Great Plains Cottonwood/herbaceous)
Rock Creek (Site 1)	REFRC1	48.8758	-106.8967	83	Yes	**Yes**	Shrub & Herbaceous Riparian Complex
Sheep Cr	M48SHEPC01	47.9675	-105.3869	77	Yes	No	Shrub & Herbaceous Riparian Complex
Horse Tied Cr	M51HRSTC01	48.1175	-104.1053	77	Yes	No	Woody Draw (Green Ash/ Common Chokecherry)
Willow Cr (North)	M43WILOC01	48.5764	-106.9814	73	Yes	**Yes**	Recent Riparian Complex (Great Plains Cottonwood/herbaceous, & Sandbar Willow)
Porcupine Cr	M44PRCPC01	48.2081	-106.3814	73	Yes	No	Recent Riparian Complex (Great Plains Cottonwood/herbaceous)
Wolf Creek at Wolf Point	REFWC	48.0878	-105.6781	72	Yes	**Yes**	Recent Riparian Complex
Rock Creek (Site 2)	REFRC2	48.5903	-107.0011	69	Yes	**Yes**	Recent Riparian Complex (Great Plains Cottonwood/herbaceous)
West Fork Poplar River	REFWFPR	48.6969	-105.8319	68	Yes	**Yes**	Shrub & Herbaceous Riparian Complex
Shotgun Cr	M51SHGNC01	48.1608	-104.2467	67	Yes	No	Oxbow/Cattail Marsh
Middle Fork Poplar River	M47POPR02	48.9194	-105.6075	63	No	No	Shrub & Herbaceous Riparian Complex
Wolf Creek nr Medicine Lake	M50WOLFC01	48.4908	-104.6064	62	No	No	Shrub & Herbaceous Riparian Complex
Big Sandy Cr	M20BSNDC01	48.4517	-109.9189	61	No	No	Shrub & Herbaceous Riparian Complex (Foxtail Barley and Woods Rose)
Larb Cr	M41LARBC01	48.2664	-107.2650	61	No	No	Recent Riparian Complex (Western Snowberry)
Butte Cr	M47BUTEC01	48.8300	-105.6047	61	No	No	Shrub & Herbaceous Riparian Complex
Beaver Cr	M41BEVRC01	48.2511	-107.5722	55	No	No	Recent Riparian Complex (Boxelder/Common Chokecherry)
Frenchman Cr	M40FRMNC01	48.7564	-107.2114	54	No	No	Recent Riparian Complex (Western Snowberry)
Battle Cr	M36BATLC01	48.6506	-109.2303	53	No	No	Recent Riparian Complex (Great Plains Cottonwood and Western Snowberry)
Redwater River	M48RDWR01	47.9281	-105.2636	53	No	No	Recent Riparian Complex (Great Plains Cottonwood)
Smoke Cr	M50SMOKC01	48.3589	-104.7461	52	No	No	Shrub & Herbaceous Riparian Complex
Little Muddy Cr	M51LMDYC01	48.1303	-104.1128	52	No	No	Shrub & Herbaceous Riparian Complex
Willow Cr (South)	M45WILOC01	48.1403	-106.6267	25	No	No	Recent Riparian Complex (Sandbar Willow)

* STORET is the EPA's national database for water quality data. *Available at* http://www.epa.gov/storet

†Per methods in Suplee *et al.* [42].

‡Per Hansen *et al.* [157].

2.4 Reach Layout, Stream Habitat Assessment, and Sampling Frequency

Each site was laid out as a short reach determined as 40 times the mean wetted width taken at the initial visit, or a minimum of 150 m of stream length, and assessed using EPA's Environmental Monitoring and Assessment Program (EMAP) physical habitat characterization protocols[67]. Other stream condition and human impacts were assessed as well (see Section 2.0 of the Northwestern Glaciated Plains technical report by Suplee[41]. Each stream site was sampled from 2 to 4 times per year during a restricted time frame between late May and late September. This period will be referred to hereafter as the "spring-summer period". Repeat visits to each site during the spring-summer period were spaced so that they occurred approximately thirty days apart. The study was carried out over a four-year period, not all sites were sampled in each year, and some sites were sampled multiple years (Table 2.2). Reference sites were sampled most frequently, as we wanted to characterize them the most thoroughly given our available resources. At sites sampled over multiple years, physical habitat characterizations were done only in the first year since stream channel/riparian conditions did not substantially change from 2001 to 2004.

Tabel 2.2 Inventory of Repeat Measures for Parameters Measured at Each Site, 2001-2004.

Stream Site Name	Year(s) Sampled	Reference Site (y/n)	Nutrient Samples	Common ions, TSS, pH, etc.	Flow	Total Benthic Plant Biomass*	Diatom Assemblage
Rock Cr (BLM)	2004	**Yes**	3	3	3	22	4
Clear Creek	2001, 2003	**Yes**	6	5	3	21	6
Bitter Cr	2004	**Yes**	3	3	3	31	6
Rock Creek (Site 1)	2003	**Yes**	2	2	1	0	4
Willow Cr (North)	2001, 2002	**Yes**	8	7	5	36	5
Wolf Creek at Wolf Point	2002, 2003, 2004	**Yes**	9	9	4	49	15
Rock Creek (Site 2)	2001, 2002, 2003, 2004	**Yes**	12	12	9	66	17
West Fork Poplar River	2002, 2003, 2004	**Yes**	9	10	7	67	16
Sheep Cr	2002	No	4	4	2	12	4
Horse Tied Cr	2002	No	4	4	2	16	2
Porcupine Cr	2001, 2002, 2004	No	11	10	7	70	12
Shotgun Cr	2002	No	4	4	2	22	3
Middle Fork Poplar River	2002, 2004	No	7	8	5	54	10
Wolf Creek nr Medicine Lake	2002	No	4	4	2	22	3
Big Sandy Cr	2001, 2004	No	7	6	6	55	8
Larb Cr	2001	No	3	3	2	19	1
Butte Cr	2002, 2004	No	7	8	5	53	13
Beaver Cr	2001	No	4	3	3	22	2
Frenchman Cr	2001	No	4	3	3	22	2
Battle Cr	2001	No	5	3	3	18	2
Redwater River	2002	No	4	4	2	21	4
Smoke Cr	2002	No	4	4	2	21	5
Little Muddy Cr	2002	No	4	4	2	21	3
Willow Cr (South)	2001, 2004	No	6	6	5	53	6

* Each sample collected at each transect is seperately inventoried here. See text for details on reach layout and sample collection.

2.5 Water Quality Measurements

See Section 2.3.2 and 2.3.3 in Suplee[41] for details on water quality sampling of nutrients, common ions, etc. Flow was measured during site visits using the velocity-area method[132]. Real-time water quality measurements were taken during each field visit including pH, specific conductance (µS/cm @ 25^{o} C), temperature, and dissolved oxygen (DO). To provide a cross-check of temperature measurements taken during field visits, continuous temperature measurements were collected using Optic Stow Away® loggers placed in three sites (Rock Cr [Site 2], Porcupine Cr and Wolf Cr at Wolf Point) during summer 2004. Mean summer-long temperatures derived from the loggers placed in these three streams were 21, 19 and 21^{o} C, respectively. Stream water DO at saturation was calculated using the mean elevation of the sites (713.5 m; 1 standard error = 25 m) and the mean spring-summer water temperature (from field measurements) for all sites (22^{o} C).

2.6 Algal and Macrophyte Biomass Sampling (Quantitative)

See Section 2.3.4 of Suplee[41] for details on macrophyte, filamentous algae, and phytoplankton sampling and laboratory analysis. A change that occurred for 2004 was the separate collection of macrophytes and filamentous algae. Whereas in the first years of the project all aquatic plant material in a given hoop sample (macrophytes, filamentous algae) was analyzed together for reporting of Chl *a* and AFDW, starting in 2004, individual hoop samples were processed in the field such that the macrophyte component and the filamentous algae component were physically separated so they could later be quantified individually. In this appendix the term "total benthic plant biomass" refers to the aggregate biomass, in mg Chl *a*/m^2, of floating filamentous algae, attached benthic algae, and macrophytes. It does not include phytoplankton biomass, which were separately measured and are expressed as µg Chl *a* /L.

2.7 Determination of Algal Taxonomical Composition

Composite samples from each site during each visit were collected and preserved (Lugol's solution) for identification of soft-bodied algae and diatoms. In 2001 and 2002, composite samples of three algae habitat types were collected: "plant" type, "rock" type and "sediment" type. After collection of the quantitative Chl *a* samples at each transect, a qualitative sample of the same representative material was also collected. For the "sediment" type samples, only material from the very surface of the stream bottom was collected. For each reach, all material from a common sample type (e.g., all "plant" type subsamples) was composited, resulting in up to three composite habitat-type samples per stream site, per visit. In 2003 and 2004, multi-habitat composite samples were collected from each site and comprised all three habitat types described above[133].

Samples were submitted to *Hannaea* in Helena, MT for identification of soft-bodied algae and diatoms. Relative abundance and ordinal rank by biovolume of diatoms and genera of soft (non-diatom) algae were determined per methods by Bahls[98]. Algae of the division Cyanophyta (cyanobacteria) were also identified. Soft algae were identified using standard taxonomic texts[134-138]. A subsample of each sample was then cleaned of organic matter using potassium dichromate, sulfuric acid, and hydrogen peroxide. Permanent diatom slides were prepared using Naphrax following standard methods[139]. Approximately 300 diatom cells (600 valves) were randomly counted and identified to species using established taxonomic references[140-143]. Diatom naming conventions followed those adopted by the Academy of Natural Sciences for USGS NAWQA samples[144].

Published diatom indexes (i.e., metrics) were reviewed to see if they could be related to Montana's water quality standards[97,115]. By metric we mean a group of diatom attributes used together to assess a water quality condition, for example to tell us about pH. Metrics considered were (1) a metric for pH and (2) an oxygen tolerance index (OTI)[115]. The OTI classes range from 1 to 5 and —counter intuitively — higher class values indicate *decreasing* DO concentrations. That is, the oxygen tolerance index is assessing tolerance to low DO conditions[115]. For each sample, a weighted-average OTI score was calculated by multiplying each DO tolerance value or class (i.e. 1, 2, 3, 4, or 5) by the proportion of sample taxa in that class, summing these results, and then dividing by the proportion of OTI-classified taxa in the entire sample[117]. If, for example, 25% of the taxa in a sample were not classified into one of the five OTI classes, the denominator would be 0.75.

We also wanted to make distinctions between DO saturation-deficit resulting from eutrophication *vs.* DO saturation-deficit attributable mainly to organic pollution (i.e., carbonaceous or nitrogenous BOD). For each sample, the proportion of organic pollution tolerant taxa in each sample was determined per Kelly and Whitton[124] and Kelly[145]. Sites with >

40% organic pollution tolerant taxa are likely to have significant organic pollution[124], and such sites were removed from the dataset in order to carry out analyses (e.g., correlation with nutrients) without the confounding influence of organic pollution. Table 2.3 provides the diatom taxa considered to be organic pollution tolerant.

Table 2.3. List of Organic Pollution Tolerent Diatoms*, Per Kelly and Whiton[124] and Kelly[145].

Gomphonema parvulum
Navicula gregaria
Navicula incertata
Navicula lanceolata
Navicula minima
Navicula pelliculosa
Navicula saprophila
Navicula subminuscula
Navicula tenelloides
Nitzschia spp.
Sellaphora seminulum

*List was crossed-check with L. Bahls for applicability to Montana taxa autecology.

Biological metrics are generally designed around population-level changes, and the dominance of a single or a few unclassified taxa in a sample tends to compromise the sample's metric results[120] (also W. Bollman, personal communication, April 2008; D. Charles, personal communication, April 2008). "Unclassified" taxa are those that do not play a role in the metric in question. Nine samples (of 152) were eliminated from the OTI metric dataset because a solid majority (> 55%) of each of the sample's diatom counts was given as "unclassified"; that is, the majority of diatom species encountered did not fall into any of the five OTI tolerance classes. Taxa counts from each of these nine samples were reviewed, and in all nine samples diatom abundance was dominated by a few or even a single taxa. Further, the total number of taxa in most of these samples was greatly reduced relative to what was seen in samples from the rest of the study. The usability of the nine samples was likely compromised by a reduced overall population and/or the dominance of a few unclassified taxa, and therefore were not used in the analyses.

2.8 Data Analyses, Statistical Tools, and Data Compilation

To understand differences between reference and non-reference steams and to assess the influence of different stream parameters on aquatic plant measurements, we carried out statistical tests of difference and correlation. We examined parameters that have already been shown to affect aquatic plant growth in streams (for overview see Wetzel[9] and Stevenson *et al*.[36]) and parameters that helped us evaluate the usefulness of the diatom metrics (more on this, next subsection). Parameters important to stream aquatic plants include light[146-148], nutrients[56,62,83,148,149], stream scour/TSS[45,150-154], substrate[155,155], and total dissolved salts[156,157].

Each study site was considered to be independent. Most sites were located on different streams, but those on the same stream (i.e., the three Rock Creek sites) were considered independent because (1) there was at least 1.6 km of stream distance between them and (2) at least one tributary confluence was found between the sites. Sampling frequency varied greatly in this study. Between 2001 and 2004 some sites were sampled only in one year, while others were sampled for two, three or even four years (Table 2.2). During any given spring-summer period, each site was repeatedly sampled from 2 to 4 times. To preclude potential temporal pseudoreplication issues[81], data collected at each site were reduced to means. For example, if a site was sampled 4 times during the spring-summer period but only in 2002, the 4 repeat measures of parameter *X* were used to calculate the mean; if another site was sampled 3 times in 2001 and 4 times in 2004, all 7 repeat measures of *X* were used to calculate the site mean. This resulted in an *n* of 24 (8 reference, 16 non-reference sites) for each parameter, one value per site, with some sites being better characterized over the study period than others.

Statistical analyses were made using Minitab® (Release 15). Statistical differences were considered significant when p-values were < 0.10. For all parameters, non-detects were replaced with values equal to 50% of the reported detection limit[158] prior to use in statistical tests. One-sided tests were used in all situations where *a priori* relationships were expected. The Mann-Whitney test was used to determine significant differences between reference and non-reference sites. The Spearman Rank test[159] was used to examine the strength (rho) and significance (p-value) of correlations between parameters. Spearman Rank p-values were not taken from Minitab® (which Minitab® states are inaccurate), but were instead calculated as shown on page 317 of Conover[159]. In the correlations we did not apply a Bonferroni adjustment to the 0.1 significance threshold because (1) each smaller "study" (i.e., analysis) done in the context of the larger study was considered on its own merits, and (2) we did not want to further increase the chance of making type II errors (i.e., declaring truly significant relationships insignificant). Changepoint analysis[101] was used to identify the occurrence of thresholds in scatterplots between diatom metrics and nutrients. Language was written for Stata® (version 10) to carry out changepoint analysis via the deviance reduction approach[61,101,160]. The analysis included an approximate Chi-square-test to determine the significance of the changepoint. The Chi-square test assumes that the deviance reduction divided by the scale parameter is approximately Chi-square distributed[161].

2.9 Identifying a Harm-to-Beneficial Use Threshold

Five conditions were used to assess whether or not a diatom metric would be useful for helping establishing nutrient criteria. These were (1) the metric had to be meaningfully linked to a numeric Montana water quality standard, (2) there was a significant correlation between the metric and a nutrient concentration, (3) the metric changed (increased, decreased) in the direction expected *a priori*, (4) reference site data points in the scatterplot between the metric and a nutrient needed to be located in the region of the plot where they were expected, and (5) the metric was fairly insensitive to other important variables measured in the streams.

Montana water quality standards include numeric criteria for DO and pH[8], and diatom metrics that could be linked to these standards were described earlier (see Section 2.7, this Appendix). The pH metric was not further considered, however, because the streams all had pH values $\geq$ 7.6

(maximum 9.0), which only permitted the use of one or at most two (of six) pH tolerance-values in van Dam *et al.*[115]. In effect, the attribute could not be evaluated by condition No. 3 above. For the diatom oxygen tolerance index (OTI), van Dam *et al.*[115] provide diatom-assemblage tolerance values (classes) that correspond with DO saturation deficits (e.g., diatoms with a tolerance value of 3 tolerate DO between 75% and 50% of saturation, while tolerance-value 2 diatom taxa are found where DO is between 100% and 75% of saturation). A simple linear regression was made between van Dam *et al.'s* five tolerance values (x axis) and their associated % DO saturation requirements (y axis) (Figure 2.2). The linear regression could then be used to convert metric scores to inferred % DO saturation.

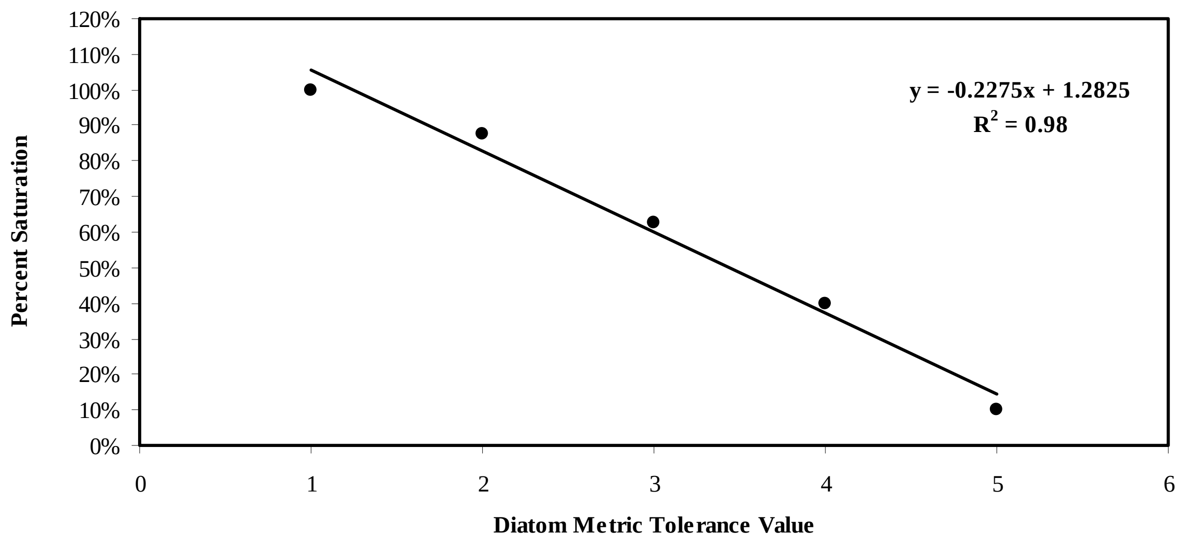

Figure 2.2. Relationship Between Diatom Metric Values and Inferred Percent DO Saturation of Stream Water (After van Dam *et al.*[115]).

DO concentrations were then estimated for each sample by multiplying the diatom-inferred % saturation by the stream's DO *at* saturation. Diatom-inferred DO concentration estimates were then compared to state DO standards. Montana DO standards vary by averaging period (e.g., 1 day, weekly, monthly, etc.[8]). Given the typical generation time of diatoms, the DO estimates from the aggregate diatom-community samples are best associated with the weekly DO standards[162] (also L. Bahls, personal communication, April 2006), as opposed to the monthly or daily standards. Therefore, we compared the diatom-inferred DO estimates to the state's 7 Day Mean Minimum standard. The 7 Day Mean Minimum is the average of each of the lowest daily DO values measured over a seven day period; these minima almost always occur at night.

Section 3.0
Results

3.1 Physical, Chemical, and Biological Characteristics of the Study Streams

Riparian habitats ranged from those dominated by sedges and rushes and having no trees and few shrubs, to deciduous forests of green ash, boxelder, and American elm (Table 2.1). The most common riparian habitats were the Shrub and Herbaceous Riparian complex and the Recent Riparian Complex[163]; this was true for both reference and non-reference sites. Flow was highly variable (note minimum and maximums; Table 3.1), and most sites ceased flowing in summer and became intermittent (i.e., a series of disconnected pools). Half the reference sites could be considered perennial and half intermittent, while 31% of non-reference sites were perennial and 69% intermittent. Individual TSS samples varied over four orders of magnitude across all sites, while TSS site means ranged from 3 (West Fork Poplar River) to 513 mg/L (Willow Cr [South]). TSS correlated significantly with Rosgen entrenchment ratio (rho = -0.311, p = 0.077). Higher Rosgen entrenchment ratios are found in streams having better-developed floodplains, inferring decreased potential for streambed scour[164]. Thus, the streams with higher scouring potential had higher TSS concentrations. The streams generally had high ionic concentrations, as mean specific conductivity for most streams was above 1000 μS/cm during the spring-summer period. The dominant cation and anion were Na^+ and SO_4^{2-}.

Table 3.1. Descriptive Statistics for Parameters Measured in Reference and Non-reference Sites, 2001-2004. All Statistics Were Based on an *n* of 8 and 16 for Reference and Non-reference Sites, Respectively, Unless Noted Otherwise.

Parameter	Reference Sites			Non-reference Sites		
	Mean ± 1 SE	Minimum	Maximum	Mean ± 1 SE	Minimum	Maximum
Water temperature (°C)	20.8 ± 0.6	19.0	24.0	23 ± 0.9	18.0	31.0
Flow (m^3/second)	0.05 ± 0.03	0.00	0.19	0.06 ± 0.02	0.00	0.30
TSS (mg/L)	64 ± 28	3	225	63 ± 31	4	513
Dissolved oxygen (mg/L)*	7.5 ± 0.6	4.0	10.0	8.9 ± 0.5	6.0	12.0
pH	8.3 ± 0.2	7.6	8.8	8.5 ± 0.1	8.1	9.0
Electrical conductivity (µS/cm @ 25 °C)	1365 ± 172	762	2333	2312 ± 301	532	4475
Total alkalinity (mg/L as $CaCO_3$)	331.0 ± 56.9	70.0	524.0	423.2 ± 57.8	71.0	794.0
Total hardness (mg equivalent $CaCO_3$/L)	253.4 ± 57.0	121.0	613.0	434.1 ± 56.0	103.0	891.0
Sodium (mg/L)	220.1 ± 30.2	46.0	353.0	395.8 ± 66.3	68.0	928.0
Chloride (mg/L)	10.3 ± 2.1	5.2	22.7	29.4 ± 6.9	2.7	105.0
Sulfate (mg/L)	430.0 ± 136.0	79.0	1263.0	921.0 ± 162.0	142.0	1918.0
Total phosphorus (µg/L)	135.4 ± 46.0	39.5	442.5	191.2 ± 62.1	29.1	900.7
Soluble reactive phosphorus (µg/L)†	8.8 ± 3.1	3.5	23.5	28.3 ± 11.3	1.8	129.3
Total nitrogen (µg/L)	863 ± 58	580	1120	1241 ± 498	580	2230
Nitrite + nitrate (µg/L)	10.0 ± 5.2	1.3	44.1	46.7 ± 43.6	0.3	699.7
Total benthic plant biomass (mg Chl *a* /m^2)‡	58 ± 20	8	133	104 ± 22	3	311
Stream bottom macrophye cover (%)	14 ± 9	0	74	30 ± 7	0	74
Phytoplankton density (µg Chl *a* /L)	9.9 ± 2.6	1.6	19.4	39.9 ± 29.2	1.4	475.7
Diatom Oxygen Tolerance Index (OTI) (all sites)	2.53 ± 0.10	2.04	2.83	2.82 ± 0.07	2.19	3.18
Diatom OTI (site <40% organic pollution tolerant taxa)[a]	2.44 ± 0.11	2.04	2.82	2.69 ± 0.08	2.19	3.10

* All DO measurments were taken during daylight hours.

† Soluble reactive phosphorus was measured from 2002 onward, and the *n* for reference and non-reference sites equals 6 and 12, respectively.

‡ Samples not collected from one reference site, therefore *n* for reference and non-reference sites equals 7 and 16, respectively.

[a] After removing sites with >40% organic pollution tolerant taxa, there were 6 reference and 10 non-reference sites remaining.

TN concentrations were significantly lower in reference sites than in non-reference sites (p-value = 0.03). Reference sites had a mean TN of 863 µg/L and 1 standard error of the mean equal to 58 µg/L (mean ± 1SE), while non-reference sites had a mean of 1241 µg/L (1 SE = 498 µg/L; Table 3.1). Among the 24 sites, mean NO_{2+3} concentrations varied over three orders of magnitude (0.3 to 699.7µg/L) and were more variable in the non-reference sites than the reference sites (Table 3.1); however, no significant difference between the groups was detected (test assumption that reference NO_{2+3} < non-reference was not fulfilled). TP concentrations in reference sites were not significantly lower than non-reference sites (p-value = 0.46). Similarly, mean SRP in the reference sites was not significantly lower (p = 0.24) than in the non-reference sites (Table 3.1).

Total benthic plant biomass in the reference sites averaged 58 mg Chl *a*/m^2 (1 SE = 20), and in the non-reference sites 104 mg Chl *a*/m^2 (1 SE = 22; Table 3.1). There was no significant difference in total benthic plant biomass between the two groups (p = 0.3, two-sided test). The % stream bottom macrophyte cover averaged 14% (1 SE = 9%) in the reference sites and 30% (1 SE = 7%) in the non-reference sites, and there was no significant difference between the two groups (p = 0.11, two-sided test). Finally, phytoplankton density averaged 9.9 µg/L (1 SE = 3 µg/L) in the reference sites and 39.9 µg/L (1 SE = 29 µg/L) in the non-reference sites, and there was no significant difference between the reference and non-reference sites (p = 0.91, two-sided test).

3.2 Response of Non-diatom Aquatic Plant Measurements to Environmental Parameters

Total benthic plant biomass was negatively correlated with TSS (rho = -0.58), and positively correlated with Rosgen entrenchment ratio (rho = 0.39; Table 3.2) — note that higher entrenchment ratios are associated with channels that are better able to access wide floodplains. Thus, more total benthic plant biomass was found in sites that had a reduced likelihood of scouring flows. Riparian canopy density, stream substrate size, and electrical conductivity were not significantly correlated with total benthic plant biomass. TP was negatively correlated with total benthic plant biomass (rho = -0.48), however TP was also significantly correlated (positively) with TSS (rho = 0.53), suggesting that TP may have been acting as a TSS surrogate in these streams. Total N was not significantly correlated with TSS (Table 3.2). The % streambed cover by macrophytes correlated negatively with TSS and TP (rho = -0.63 and -0.39, respectively; Table 3.2) but not with any other parameters. Surprisingly, phytoplankton density — quantified as µg Chl *a*/L — correlated positively with TSS (rho = 0.59). Phytoplankton correlated with several nutrients as well (TN, TP, and NO_{2+3}).

Table 3.2. Spearman Rank Correlations (rho) and p-values for Correlations Between Driving and Response Variables. Significant Relationships ($p < 0.1$) Shown in Bold*.

Variables Tested: Response Variable	Driving Variable	Expected Relationship (+, -, not sure is 2 tail)	Spearman's rho	n	Estimated p-value, from Table A1 of Conover (1999)
Total benthic plant biomass (mg Chl a/m^2)†	Bank canopy density	negative	0.041	23	0.424
Total benthic plant biomass (mg Chl a/m^2)†	Midchannel canopy density	negative	0.115	23	0.295
Total benthic plant biomass (mg Chl a/m^2)†	Stream substrate D50	not sure	-0.073	23	0.732
Total benthic plant biomass (mg Chl a/m^2)†	**Log10 mean TSS conc. (mg/L)**	**negative**	**-0.577**	**23**	**0.004**
Total benthic plant biomass (mg Chl a/m^2)†	**Rosgen entrenchment ratio**	**positive**	**0.386**	**22**	**0.038**
Total benthic plant biomass (mg Chl a/m^2)†	EC (uS/cm)	negative	0.212	23	0.160
Total benthic plant biomass (mg Chl a/m^2)†	TN conc. (mg/L)	positive	0.266	23	0.106
Total benthic plant biomass (mg Chl a/m^2)†	NO_{2+3} conc. (ug/L)	positive	-0.202	23	0.172
Total benthic plant biomass (mg Chl a/m^2)†	**TP conc. (ug/L)**	**not sure**	**-0.475**	**23**	**0.026**
Total benthic plant biomass (mg Chl a/m^2)†	SRP conc. (ug/L)	positive	-0.031	18	0.449
Streambed macrophyte cover (%)	Midchannel canopy density	negative	0.27	24	0.100
Streambed macrophyte cover (%)	Stream substrate D50	not sure	-0.24	24	0.250
Streambed macrophyte cover (%)	**Log10 mean TSS conc. (mg/L)**	**negative**	**-0.634**	**24**	**<0.001**
Streambed macrophyte cover (%)	Rosgen entrenchment ratio	positive	0.14	22	0.261
Streambed macrophyte cover (%)	EC (uS/cm)	negative	0.332	24	0.056
Streambed macrophyte cover (%)	TN conc. (mg/L)	not sure	0.2	24	0.338
Streambed macrophyte cover (%)	NO_{2+3} conc. (ug/L)	positive	-0.379	24	0.070
Streambed macrophyte cover (%)	**TP conc. (ug/L)**	**not sure**	**-0.389**	**24**	**0.062**
Streambed macrophyte cover (%)	SRP conc. (ug/L)	positive	0.022	18	0.464
Phytoplankton concentration (ug/L)	Midchannel canopy density	negative	0.161	22	0.230
Phytoplankton concentration (ug/L)	Stream substrate D50	not sure	-0.335	22	0.124
Phytoplankton concentration (ug/L)	**Log10 mean TSS conc. (mg/L)**	**positive**	**0.558**	**22**	**0.005**
Phytoplankton concentration (ug/L)	Rosgen entrenchment ratio	negative	-0.179	21	0.212
Phytoplankton concentration (ug/L)	EC (uS/cm)	negative	-0.031	22	0.443
Phytoplankton concentration (ug/L)	**TN conc. (mg/L)**	**positive**	**0.352**	**22**	**0.053**
Phytoplankton concentration (ug/L)	**NO_{2+3} conc. (ug/L)**	**positive**	**0.346**	**22**	**0.056**
Phytoplankton concentration (ug/L)	**TP conc. (ug/L)**	**positive**	**0.436**	**22**	**0.023**
Phytoplankton concentration (ug/L)	SRP conc. (ug/L)	positive	0.121	18	0.309
Diatom OTI	Total benthic plant biomass (mg Chl a/m^2)	positive	-0.021	23	0.461
Diatom OTI	**Phytoplankton Concentration (ug/L)**	**positive**	**0.369**	**22**	**0.046**
Diatom OTI	Streambed macrophyte cover (%)	positive	0.103	24	0.311
Diatom OTI	Log10 mean TSS conc. (mg/L)	not sure	0.188	24	0.183
Diatom OTI	Rosgen Entrenchment Ratio	not sure	-0.352	22	0.108
Diatom OTI	**EC (uS/cm)**	**not sure**	**0.422**	**24**	**0.021**
Diatom OTI	**TN conc. (mg/L)**	**positive**	**0.431**	**24**	**0.019**
Diatom OTI	NO_{2+3} conc. (ug/L)	positive	0.13	24	0.266
Diatom OTI	**TP conc. (ug/L)**	**positive**	**0.446**	**24**	**0.016**
Diatom OTI	**SRP conc. (ug/L)**	**positive**	**0.41**	**18**	**0.046**
Diatom OTI, sites with <40% organic pollution tolerant taxa	**TN conc. (mg/L)**	**positive**	**0.363**	**16**	**0.080**
Diatom OTI, sites with <40% organic pollution tolerant taxa	NO2+3 conc. (ug/L)	positive	0.167	16	0.259
Diatom OTI, sites with <40% organic pollution tolerant taxa	**TP conc. (ug/L)**	**positive**	**0.396**	**16**	**0.063**
Diatom OTI, sites with <40% organic pollution tolerant taxa	SRP conc. (ug/L)	positive	0.24	13	0.203
TP conc. (ug/L)	**Log10 mean TSS conc. (mg/L)**	**positive**	**0.526**	**24**	**0.006**
TN conc. (mg/L)	Log10 mean TSS conc. (mg/L)	not sure	-0.023	24	0.912
Log10 mean TSS conc. (mg/L)	**Rosgen entrenchment ratio**	**negative**	**-0.311**	**22**	**0.077**

*Significant relationships that responded opposite of the expected relationship are not shown in bold.

†Combined biomass of benthic algae, macrophyes, and floating filamentous algae.

3.3 The Diatom OTI in Relation to Environmental Parameters

Using the complete dataset (24 sites), it was found that the diatom OTI did not correlate to Rosgen entrenchment ratio, TSS, total benthic plant biomass, or % macrophyte cover, but did positively correlate with phytoplankton density (rho = 0.369). The diatom OTI correlated positively with several nutrients, including TN (Table 3.2). The scatterplot between the diatom OTI and TN is shown in Figure 3.1. Note the position of the reference sites on the left side of the scatterplot.

In the analyses up to this point, the diatom OTI had passed all 5 screening conditions. The OTI could (1) be linked to Montana water quality standards (DO), (2) it correlated significantly to several nutrients and (3) in the manner expected (Table 3.2), (4) the reference sites are generally where they would be expected to be (Figure 3.1), and (5) it was insensitive to other important non-nutrient factors in the streams (paragraph above). To further assess condition 5 (i.e., the metric is fairly insensitive to other environmental variables measured in the streams), only sites having <40% organic pollution tolerant taxa (n =16 sites) were carried to the next step. The scatterplot for this diatom OTI *vs*. mean TN relationship is shown in Figure 3.2. A significant changepoint (p = 0.026) was detected at 1.12 mg TN/L where the mean diatom OTI increased from 2.5 (group of sites to the left of 1.12 mg TN/L) to 2.9 (sites to the right of 1.12 mg TN/L) (Figure 3.2). The 90% confidence interval for this changepoint is 0.78-1.48 mg TN/L. (Changepoint analysis was also run on the full [n=24] dataset. A significant changepoint [p <0.01] was also detected at 1.12 mg TN/L, with a 90% confidence interval from 0.84 to 1.3 mg TN/L.) Given that mean spring-summer temperature in the streams during the study was 22^{o}C and the average site elevation was 713.5 m with little variance (1 SE = 25 m), stream DO at saturation was estimated to be 8 mg/L. Electrical conductivity (i.e., dissolved salts) was not high enough in the streams to affect DO saturation meaningfully.

Recall that diatom-inferred DO saturation equals: -0.2275·(OTI score) + 1.2825 (Figure 2.2). Hence, the group of streams with TN ≤ 1.12 mg TN/L (and a mean OTI score of 2.5) had a mean inferred DO of about 5.7 mg/L (-0.2275·2.5 + 1.2825 = 71%; 8 mg/L·71% = 5.7 mg DO/L). The group with TN > 1.12 mg TN/L had a mean inferred DO slightly less than 5.0 mg/L (-0.2275·2.9 + 1.2825 = 62%; 8 mg/L·62% = < 5.0 mg DO/L). None of the reference sites had a mean TN concentration greater than 1.12 mg TN/L, nor did any of them have an inferred DO concentration < 5 mg/L (Figures 3.1, 3.2).

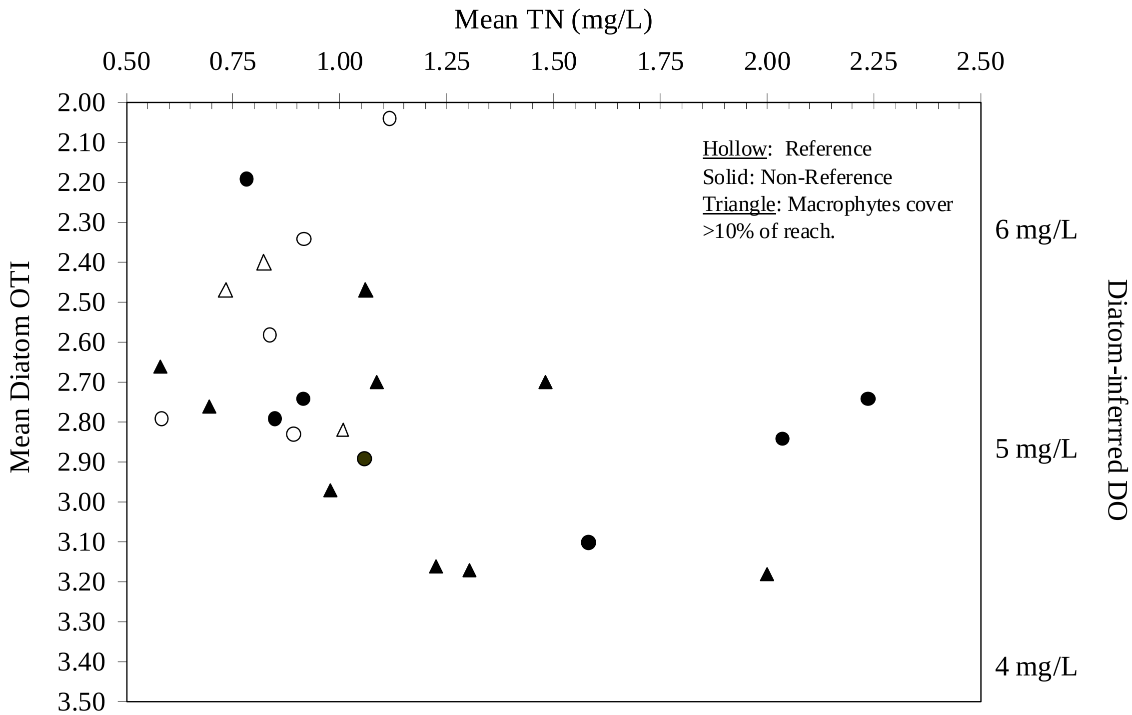

Figure 3.1. Scatterplot of Diatom OTI *vs.* Total N Concentrations, All Sites. Diatom-inferred dissolved oxygen (DO) was calculated based on a DO at saturation of 8 mg/L.

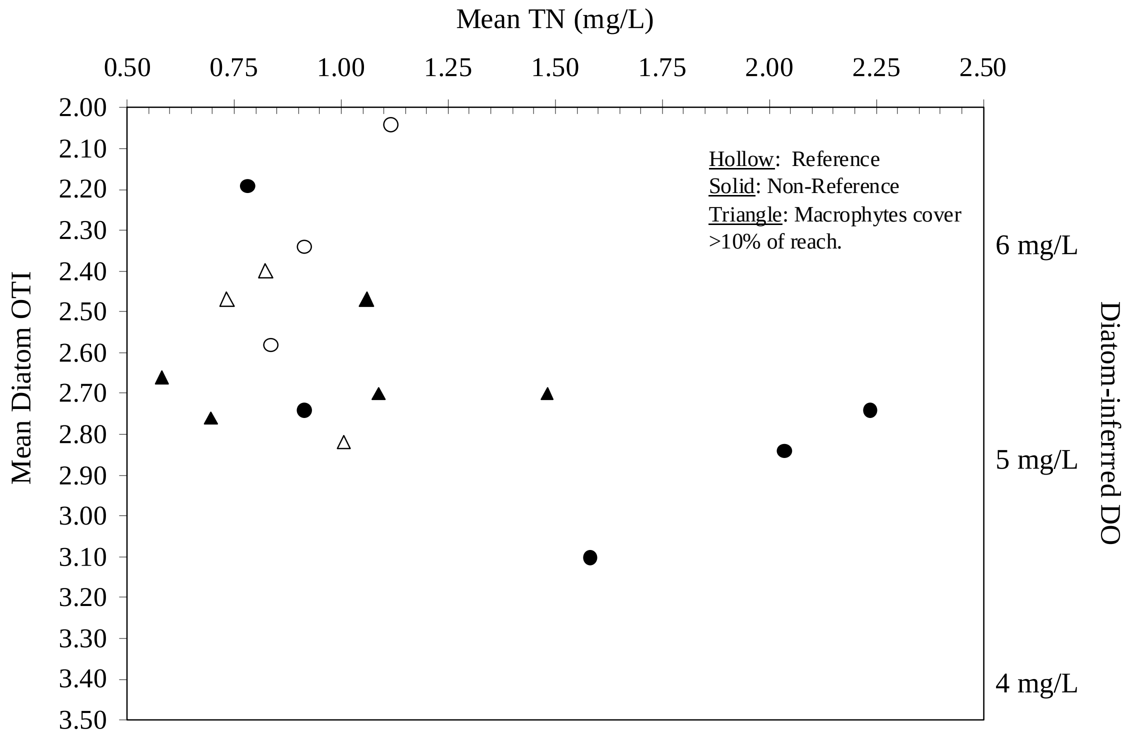

Figure 3.2. Scatterplot of Diatom OTI *vs*. Total N Concentrations, Only Sites Where Organic Pollution Tolerant Taxa Were < 40% of the Population. Diatom inferred dissolved oxygen (DO) was calculated based on a DO at saturation of 8 mg/L.

Section 4.0
Discussion

4.1 Important Driving Variables Affecting the Prairie Streams

Stressor-response relationships between nutrients and aquatic plants (including diatoms) are not easily studied in streams where multiple environmental variables are operating simultaneously[56,78]. So, to the extent practical, we quantified stream nutrient concentrations as well as other, non-nutrient parameters and correlated these to measures of the aquatic plant community to better understand the trend and magnitude of their effects (Table 3.2). A complex but identifiable pattern begins to emerge when these data are examined together. Streams with the highest potential for bottom scour during high flows (i.e., those with low Rosgen entrenchment ratios) had higher TSS as well as lower total benthic plant biomass (Table 3.2). Fine sediments are common in prairie streams and are easily entrained during high flows, especially in the more entrenched channels. Thus, it would be expected that TSS and entrenchment relate to one another. Phytoplankton were *positively* correlated with TSS, which may at first glance be counter-intuitive as one might expect that high TSS would reduce light and, therefore, the phytoplankton standing crop. But light (measured as riparian canopy density) was apparently not a strong regulator of phytoplankton biomass, nor of benthic plant biomass or macrophyte cover (Table 3.2). Phytoplankton responded positively to TSS increases probably because elevated TSS was more common in scour-prone streams and scoured streams had fewer of the phytoplankton's resource competitors (i.e., benthic plants).

In these prairie streams high-flow scour acted as a key driving variable and, depending on each stream's propensity to become scoured, tended to result in two very different general endpoints (with intermediate forms also present); (1) relatively un-scoured streams where plant biomass was dominated by benthic forms, and (2) more heavily scoured streams where phytoplankton made up a major proportion of the algal biomass (see Section 6.0 and 8.0, and especially Figure 8.7, in Suplee[41]). High flows are known to be an important regulator of benthic aquatic plant communities[150,152,153]. And artificial stream studies show that an elevation in water velocity beyond that to which benthic algae are adapted leads to reduced biomass, whereas TSS additions in the same artificial streams do not have a marked effect on algal biomass[45]. Flashy high flows are common in the region due to sporadic, heavy summer thunderstorms, and so the influence of scouring flow was likely to be manifested at most sites some time during the summer.

TP was negatively correlated with total benthic plant biomass and macrophytes and, following the same pattern as TSS, was positively correlated with phytoplankton. This most surely resulted because TP was acting as a TSS surrogate. The TP-TSS correlation is not unusual, as 90% of TP carried by rivers to the sea is associated with suspended solids[73]. Due to the tight coupling of TSS and TP (Table 3.2) the role of TP as a plant nutrient *vs*. TP as a TSS surrogate is difficult to tease apart; therefore, TP will not be further addressed in this appendix as a plant nutrient.

4.2 Effects of Non-Nutrient Environmental Factors on the Diatom OTI

We saw a fairly clear pattern between the diatom OTI and TN concentrations (Table 3.2; Figures 3.1, 3.2). Because we would like to use the diatom OTI metric to infer DO concentrations and derive regional nutrient criteria, it is important to evaluate non-nutrient factors that might influence the metric given the complex nature of these streams. Macrophytes, for one, can influence diel DO concentrations but may not necessarily correlate to instream nutrients since they can get nutrients from the sediments via their roots[165]. But neither macrophyte coverage nor total benthic plant biomass correlated with nutrients *or* the OTI metric in our study (Table 3.2), and note in Figure 3.1 that streams with substantial macrophyte cover (> 10%) show a wide range of diatom OTI values (i.e., macrophyte cover is not driving the diatom OTI values). Macrophytes are present in many of these streams, often in large quantities, but their role relative to the diatom OTI is more that of a random variable. The diatom OTI is not responding to TSS either, which is important since TSS has been shown to be a key driving variable in these streams. However, the OTI was responsive to phytoplankton, which were themselves closely linked to instream nutrients (TN and NO_{2+3})(Table 3.2). Phytoplankton, every bit as much as benthic plants, can strongly affect diel DO concentrations oscillations[40]. Lastly, the diatom OTI *vs.* TN relationship remains essentially unchanged after the sites likely contaminated with organic pollution are excluded (Figures 3.1, 3.2). Organic pollution and nutrient pollution often go hand in hand[7,19], but regardless, Figure 3.2 can be viewed as the best representation of the diatom OTI's ability to respond to purely nutrient-caused DO variations in these streams. Overall, from the preceding discussion, it can be concluded that the diatom OTI is relatively insensitive to important non-nutrient environmental parameters in these streams (i.e., it meets condition No. 5; it also meets conditions 1 through 4), and is therefore useful for developing nutrient criteria.

4.3 The OTI relationship to Nutrient Concentrations, and the Dissolved Oxygen Estimates Derived from the Diatom OTI

Diatoms are known to be good indicators of stream TN concentrations[125] and eutrophication in general[123]. Other researchers find that individual tolerance classes of the diatom OTI metric correlate well to nutrient gradients in streams throughout North American, and in this particular region[118,120]. Porter *et al.*[120] report that among 35 different diatom attributes, including the 5 OTI classes of van Dam *et al.*[115], class 4 of the OTI was one of the very best indicators of nutrient enrichment based on a U.S. national dataset from the USGS.

Important for the present work is an understanding of the nature of the DO estimates provided by the diatom OTI. The available data indicate that the DO values provided by the diatom OTI reflect nighttime minima. Six case studies (five in Montana, one from the literature) were available in which diatom samples were collected during the summer and instrument-measured 24 hr DO data were also available. Three case studies were from 2007 at two sites along the Yellowstone River (7th order large river), two were from Cottonwood Creek[f] (a 2-3nd order

[f] At Cottonwood Creek, DO data were measured 24 hr/day across two days at two sites in July 2003, while the diatom data were collected at the same two sites but in August 2001. Unfortunately, simultaneous diatom and DO instrument data were not collected. Multiple-year data from the sites suggest that stream conditions were generally comparable during the two time periods, therefore the 2003 DO data were compared to the 2001 diatom data.

wadeable stream), and the last was from The Grand River near Toronto, Canada; the latter is published in Rott *et al.*[117]. Rott *et al.* use the saprobic index[115] instead of the OTI. The saprobic index and the OTI provide very similar information about stream DO. Each index has five tolerance values, with analogous tolerance values in each index indicate essentially identical DO conditions[115]. In our database, 69% of the diatom species in analogous tolerance values of each index are identical. During summer, downstream sites in the Grand River have elevated N and P concentrations, instrument-measured DO deficits of 51% of saturation, and instrument-measured nighttime DO concentrations < 4 mg/L. The elevation and summertime temperatures of the Grand River's downstream sites equate to a DO concentration at saturation of around 8 mg/L. Most diatom samples from the Grand River's downstream sites trend towards saprobic index class III, corresponding to 50% DO saturation-deficit. Thus, diatom-inferred DO concentrations on the lower Grand River would equal about 4 mg/L, which corresponds well with the instrument-measured nighttime data.

We compared nighttime DO instrument data from each of the six case studies mentioned above to their corresponding diatom-inferred DO, in order to compute the diatom OTI's overall bias and relative error. The analysis showed that the diatom OTI has a slight low bias (it provides values that are 0.18 mg DO/L lower than "true" instrument-measured nighttime DO minima), with a standard error of the mean (SEM) around the DO estimate of ± 0.58 mg/L. DEQ is continuing to collect diatom and instrument DO data to further refine these measures of the OTI's bias and error.

4.4 Identifying the Threshold of Harm to the Beneficial Uses

Weekly DO standards vary in the study region depending upon classifications in state law. Most streams in the ecoregion are classified B-3 or C-3, meaning a DO standard of 4.0 mg/L. But there are also many B-1 and B-2 streams, which have a DO standard of 5.0 mg/L[8]. In the diatom OTI vs. TN relationship (Figure 3.2), a clearly visible (and significant; p-value = 0.026) changepoint occurred at 1.12 mg TN/L. Streams with TN ≤ 1.12 mg TN/L had a mean inferred DO of about 5.7 mg/L, and streams with TN > 1.12 mg TN/L had a mean inferred DO of just under 5.0 mg/L. None of the reference sites had a mean TN concentration greater than 1.12 mg TN/L nor did any have an inferred DO concentration < 5 mg/L (Figure 3.2). To assure general protection of all regional streams, including those with DO limits of 5 mg/L, the TN concentration at the observed changepoint (1.12 mg TN/L) appears to be a reasonable harm-to-use threshold. The concentration of 1.12 mg TN/L can be considered ecologically meaningful because (1) the reference sites fall to the left of this concentration and (2) it separates the dataset into streams that, on average, always meet the higher regional DO standard (5 mg/L) from streams that do not. It also makes sense that nitrogen correlates well to the diatom OTI as these streams are likely nitrogen limited[41].

4.5 Conclusion

Stream-bottom scour and TSS are apparently among the most important variables influencing aquatic plant populations in streams of the Northwestern Glaciated Plains ecoregion of Montana. In spite of the overarching effect of TSS/bottom scour in the streams, we were able to detect the influence of stream nutrient concentration gradients on a diatom metric (diatom Oxygen

Tolerance Index, or OTI; van Dam *et al.*[115]). The usefulness of the metric for identifying a harm-to-beneficial use threshold was evaluated by comparing it to five specified conditions that assured it responded to the pollutant of concern (nutrients) and that it was fairly insensitive to other environmental parameters (physical and biological) operating simultaneously in the streams. The diatom OTI passed all 5 conditions, including a close examination of the metric's ability to discern DO minima caused by nutrient-only *vs.* DO minima caused by nutrient + organic pollution. The metric was significantly correlated to TN concentrations, and we were able to use reference sites, changepoint analysis, and stream DO standards[8] to identify a nutrient concentration that is ecologically meaningful and should protect regional aquatic life and fish. The final TN nutrient criteria recommended from the analysis is 1.12 mg TN/L.

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Using a Computer Water Quality Model to Derive Numeric Nutrient Criteria

LOWER YELLOWSTONE RIVER, MT

Photo: Montana DEQ; Mark Greytak, MontanaPictures.Net

MAY 2013

Prepared by:
Kyle Flynn, P.H. and Michael W. Suplee, Ph. D.
Montana Department of Environmental Quality
Water Quality Planning Bureau
1520 E. Sixth Avenue
P.O. Box 200901
Helena, MT 59620-0901

WQPBDMSTECH-22

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EXECUTIVE SUMMARY

The development of numeric nutrient criteria for nitrogen (N) and phosphorus (P) is one of many tasks that the Montana Department of Environmental Quality (DEQ) is working on to support its statewide water quality management objectives. The intent of these criteria is to protect waterbodies and their associated beneficial uses from eutrophication. Eutrophication, or the enrichment of waters by nutrients, causes a variety of water quality problems in flowing systems including nuisance algal growth, altered aquatic communities, and undesirable water quality changes that impair beneficial uses.

In the mid 2000s DEQ concluded that successful technical approaches for developing numeric nutrient criteria for wadeable streams and small rivers would not be transferable to large rivers. This was due to a number of reasons including: (1) a lack of reference watersheds (i.e., those with little human influence) that could be used to help derive water quality benchmarks, (2) differences in the physical character of large rivers that make them different from wadeable streams (being deeper and more light limited), and (3) generally weak correlations between nutrients and eutrophication response in the scientific literature. Cross-correlations between ambient nutrient concentrations and a variety of different stressors were further considerations.

DEQ opted instead to develop criteria for large rivers using mechanistic water quality models. Such tools have been used for many decades in water quality management and environmental decision support and have shown great value in effluent loading studies, for example. Because water quality models are deterministic and use well-described mathematical relationships among nutrients, light availability, algal uptake, growth, and nutrient recycling, they can be used to proactively manage and understand a river's physical environment. More importantly they can assist in translating between ambient water column nutrient concentrations and Montana's existing water quality standards (e.g., dissolved oxygen, pH, algal biomass, etc.). Beneficial uses that DEQ is required to protect as part of existing state-wide water quality standards for large rivers are:
H

- Public water supplies
- Aquatic life, including fish
- Recreational uses
- Agricultural uses
- Industrial uses

Nutrients previously had been addressed in Montana using narrative criteria. These are qualitative statements that describe the desired condition of a waterbody. They are flexible in that they can be adapted to many potential situations (even unforeseen ones), however, because they lack specificity and are open to varied interpretations, their subjectivity is a concern. Adoption of numeric criteria will eliminate this fault and provide readily measurable limits that are easier to monitor, assess, and regulate. Consequently, the criteria outlined in this document closely reflect the spirit and intent of the narrative criteria, but also provide sufficient detail to be of practical value.

Upon embarking on this work DEQ found that very little had been done to advance the science of large river nutrient criteria in the United States. In fact, from our literature review, this is the first documented case where criteria were derived on a large river using a water quality model. As a result, DEQ determined that the model, as well as the data supporting the model, should be of research quality. Such a level of rigor would reduce the number of model assumptions and would enhance the defensibility of the proposed criteria determined through the model.

DEQ's first task was to select an appropriate water quality model and large river segment to model. Several tools were considered. After weighing the pros and cons of each, DEQ selected the enhanced river quality model QUAL2K (Q2K). Key advantages of Q2K included: (1) the ability to simulate the eutrophication variables of interest such as dissolved oxygen, pH, total organic carbon, bottom-attached algal growth, phytoplankton, etc., (2) widespread use and national familiarity with the model, (3) relatively modest data requirements, (4) simplicity in model application and development, (5) very good modeling documentation and user support, and (6) endorsement by the U.S. Environmental Protection Agency (EPA). Additionally, Q2K was found to have been used extensively for water quality regulation including permitting and compliance, wasteload allocations, and total maximum daily loads (TMDLs) throughout the U.S. and abroad.

The river we chose to model was the Yellowstone River. It was selected for three key reasons. First, it is unregulated which lends itself to less-complex modeling scenarios. Second, it is arguably one of the most important rivers in the state due to its proximity to a large proportion of Montana's population, the industrial base found along it, and the river's national and international recognition. Finally, it has transitional water quality characteristics (e.g., sharp changes in turbidity) that help us better understand lotic water quality mechanics. The specific study reach was in the lower part of the river between Forsyth to Glendive, MT. It is 232.9 km (144.7 mi) long and part of the Great Plains ecoregion.

In 2006, a reconnaissance was completed to confirm that a one-dimensional model such as Q2K was appropriate for use on the Yellowstone River. By evaluating vertical and lateral water quality gradients at several sites along the project reach, we determined that it was aptly sufficient. We also identified a suitable time-frame for data collection and modeling. A period of stability occurs from early August to late September when conditions are approximately steady-state (i.e., water temperature, light, and hydrology are fairly stable). Such assumptions and limitations are implicitly required in the use of the model.

We then launched a major data collection effort during the summer of 2007 to support development of the model. River surveys were completed throughout the summer and included continuous monitoring of dissolved oxygen, temperature, conductivity, pH, and chlorophyll-*a* (Chl*a*) (8 sites), water chemistry monitoring (2 times), measurement of bottom-attached (benthic) algae and free-floating algae (phytoplankton), characterization of quality and quantity of water from incoming tributaries and wastewater facilities, and much more. One sampling episode was completed in August to calibrate the model, and a second was undertaken in September for validation. DEQ also cooperated with the U.S. Geological Survey (USGS) on a 2008 dye-tracer time of travel study so as to provide information for the physical structure of the model. Locations were optimized through the monitoring to ensure that the requirements of Q2K were met.

To our fortune, the data collection took place during a relatively low-flow year. In fact, it was the 7th ranked seasonal low-flow on record, between a 10 to 20 year recurrence-interval. Hence conditions were very close to design requirements for nutrient criteria. Additionally, because eutrophication problems are exacerbated at low flows [such as those used in National Pollutant Discharge Elimination System (NPDES) permits] the timing of the data collection could not have been more ideal. Perhaps most interesting, though, was that despite low-flows in the river we saw no obvious signs of water quality impairment during 2007. It can therefore be inferred that nutrient concentrations observed in

2007 would have to be elevated even higher to drive nutrient impairment. In 2007 they were ≈500 µg total nitrogen (TN) L^{-1} and ≈50 µg total phosphorus (TP) L^{-1}. Assimilative capacity therefore still exists in regard to nutrient loads in the lower Yellowstone River.

We augmented our data collection program with information from other agencies. For example, climate, bathymetry, and atmospheric information were taken from the National Weather Service, the Yellowstone River Conservation District Council, and EPA. A great deal of related information was obtained from past water quality studies, algal growth experiments, and peer-reviewed literature. In examination of this material we determined that the Yellowstone River, despite being classified as a large river, would likely be strongly influenced by benthic algae. Hence we spent considerable time ensuring that model relationships related to benthic algae were consistent with prior research. We also collaborated on a new module, AlgaeTransect2K (AT2K), which assisted in our assessment of the river.

AT2K, unlike Q2K, has the ability to simulate lateral benthic algae growth and biomass accrual across a river transect. This gave DEQ the ability to assess the lateral effect of nutrients on large rivers by integrating depth, light, and near-shore channel geomorphology into river management. The importance of such a tool is highlighted by the fact that human use and perception is often inclined toward the near-shore or wadeable regions where beneficial use is first initiated. AT2K is suited best to simulating algal growth that is closely attached to the bottom, like diatoms and short filaments of green algae, whereas its ability to simulate long streamers of attached filamentous algae that exist in the three dimensions of the water column is more limited.

We then set about developing the Q2K model for the Yellowstone River. Standard scientific and engineering principles were used in construction, calibration, and confirmation of the model. Analysis was completed until acceptable agreement was found between observed and simulated state-variables. Of those variables available to us, we relied heavily on DO, pH, total nutrients, and benthic-algae. These were some of our best field measurements. Relative error and root mean squared error statistics were quantified to assess model prediction efficiency, and after rigorous testing, we were satisfied with the calibration. It met both the criteria specified in the project's 2006 quality assurance project plan as well as other criteria from the scientific literature. Upon validation however, we found that our calibrated model was not suitable for simulating late-season conditions (i.e., our September data collection event).

Consequently, we used two additional approaches to explore the differences between the two periods. First, we closely examined the river's biological conditions as indicated by the life history and ecological requirements of diatom algae which were collected in 2007 as part of the project. Life history and ecological requirements of diatoms have been extensively studied and provide an independent means of assessing river conditions. Analysis suggested that the river was different in September than in August for a number of possible reasons, including differences in diatom communities (a shift from more to less productive taxa), apparent changes of the benthic algae matrix (less *Cladophora* that provide a 3-dimensional environment for diatoms to colonize), and possible temperature and photoperiod-induced senescence. We were able to reproduce these changes in the model by adjustment of benthic algal related growth parameters.

We also completed a second independent validation of the original calibration to address any concerns with the initial validation. A data set collected by the USGS in August of 2000 (9^{th} lowest seasonal low-flow of the record) was used. Given that their data was from a different set of climatic and nutrient conditions (but similar low-flows), this was a robust test of the model to see if it could simulate conditions outside when the model was calibrated. The model was also extended to a much longer reach (586 km - Billings to Sidney, MT) to accommodate additional data. In this instance, the validation was successful and we believe it to be an even more rigorous test than the first given that it covers a much larger spatial area and nutrient conditions than previously attempted. Hence DEQ is satisfied with the quality of the final calibrated and corroborated model.

DEQ then set about the process of deriving N and P nutrient criteria with the model. This required several initial decisions including: (1) the hydrologic design flow to use, (2) climatic conditions associated with that design flow, and (3) what (if any) alterations to the model's headwater boundary conditions should be made to account for future changes in upstream water quality (i.e., as the river moves closer to the nutrient criteria over time). To determine the first constraint, we used algal growth rates as an indicator of the response time to reach nuisance algal levels. By assuming that a waterbody must respond biologically prior to any other adverse eutrophication-caused water quality conditions (such as DO or pH impairment), an appropriate design flow should be established that will constrain the concentration of nutrients over a duration that will limit such biologically-based excursions. The frequency of the occurrence must also allow for sufficient recovery time, as indicated in EPA guidance.

By using literature based first-order net specific growth rates, we concluded that benthic algae can reach nuisance levels in about 14 days under moderately enriched conditions. Subsequently, we recommend a design flow duration of 14-days for setting nutrient limits on large rivers. A slightly conservative frequency of once every five years (14Q5) was selected which corresponds with an excursion recovery every 3 years (as recommended for biological recovery by EPA) and is consistent with published USGS low-flow statistics making it easy to identify and apply in the future. Nutrient control policies must therefore achieve water quality conditions in agreement with this recommendation. It should be noted that this low flow differs from the 7Q10 flow commonly used by DEQ for permitting discharges of toxic compounds. The 7Q10 is intended to ensure non-exceedance of a chronic criterion concentration (which is derived as a 4-day average) so that it will not occur more than once every three years. Thus the design flow selected for nutrient criteria and the design flow for chronic toxic criteria are based on the same premise (allowing exceedances only once in three years), it is just that toxic compounds require a shorter averaging period which in turn leads to a different low-flow statistic.

DEQ then used a typical meteorological year (TMY) as the design climate. These data (developed by the National Renewable Energy Laboratory) provided an unbiased set of conditions for a given location over a long period of time, such as 30 years. We chose the most probable period during which the 14-day seasonal low-flow would occur, the third week of August. Since the TMY is an annual event (i.e., it could happen every year), it is well-suited for criteria development work as it does not alter the underlying probability of occurrence (i.e., still a 5-year event). In other words, DEQ did not select a low-flow and then couple it with the worst possible weather scenario. Rather, we selected an appropriate low-flow and then applied it to the expected annual mid-summer climate.

In parallel with the low-flow data analysis, we also evaluated historical water quality data for low-flow conditions. Data from the ten lowest flow years on record (1988, 1994, 2000, 2001, 2002, 2003, 2004, 2005, 2006, and 2007) were available thanks to USGS sampling over the years. Central tendencies of these data were used to estimate water quality conditions and associated loads for our scenario analysis. A similar procedure was done for the point loads (tributaries, WWTPs, etc.). From review of this information, nutrient concentrations in the river appear to be lower than would typically impair water quality. Consequently nutrient standards should be set higher (i.e., have a greater concentration than) existing conditions.

We then carried out a series of controlled nitrogen and phosphorus additions within the modeling tools to identify nutrient levels that would impair water quality (e.g., pH, DO, benthic algae levels, etc.). To do this, incremental increases in soluble nutrient supply were evaluated in the longitudinal model until a limiting response was achieved. Two types of model runs were considered, one where soluble nitrogen was limiting and soluble phosphorus was unlimited, and the other where phosphorus was limiting and nitrogen was unlimited. Each was necessary since only one nutrient can limit algal growth in the model at any time. Ten different model runs were carried out for each limiting nutrient under different degrees of nutrient limitation where the response for each state-variable of interest was recorded and compared with existing water quality standards.

Through these model runs it was realized that our upstream boundary condition would inevitably be altered over time as the river approaches the proposed criteria. We estimated changes at this boundary using published phytoplankton-nutrient relationships and resolved other related parameters such as algal detritus and dissolved organic carbon through iterative adjustment of boundary conditions until longitudinal stability was achieved near the upper end of the model. Total nutrient concentrations were of primary interest given their greater correlation with other water quality parameters, ease of monitoring, and EPA expectations. We then used biological uptake and advective transport in the model to relate nutrient supply at one location in the river to total nutrients recycled at another.

We evaluated simulation endpoints such as DO minima, pH flux, benthic algae biomass, total dissolved gas, total organic carbon, etc. in response to this increase in nutrient supply. The highest total N or P concentration (after recycle) that did not elicit a limiting water quality response was used to determine the effective nutrient criteria at a point just below where the harmful change occurred. In the upper portion of the study reach (between Forsyth and the Powder River confluence), pH was found to be most limiting with an induced change greater than Montana's allowable maximum water quality standard of 9.0 standard units (or a maximum allowable flux of 0.5 units of pH). The nutrient criteria were established at the threshold which should keep the river below a pH of 9.0, which should be protective of aquatic life including warm water fish.

In the lower river (Powder River confluence to Glendive) benthic algae biomasses were most limiting. Use impairment occurred when the mean biomass of the wadeable region reached a threshold of 150 mg Chl*a* m^{-2}, a value known to impact recreational use. Both AT2K and Q2K were needed to make this determination. Natural turbidity was the main factor in the change in nutrient sensitivity between the upper and lower river as water clarity declined longitudinally due to fine clay particles in suspension which were mainly input from the aptly-named Powder River.

DEQ is therefore recommending the following numeric nutrient criteria which extend somewhat up- and downstream of the modeled area dividing the river into practical units for water quality management:

- 655 µg TN L^{-1} and 55 µg TP L^{-1} from the Big Horn River confluence to Powder River confluence
- 815 µg TN L^{-1} and 95 µg TP L^{-1} from the Powder River confluence to the state-line

It should be noted that these are reach-specific estimates and are applicable only to the lower Yellowstone River (e.g., they should not be transferred elsewhere in the Yellowstone or other basins). Additionally, they apply only to late summer peak productivity. High flows, late fall conditions, or anything else outside this condition would preclude these recommendations. Readers should also note that two additional Yellowstone River criteria units were identified for future study. These extend from the Wyoming state line (headwaters) to the Laurel public water supply (PWS) and from the Laurel PWS to the Bighorn River. Field data collection was undertaken in summer 2012 for each of these units.

After determining the criteria, we quantified prediction error surrounding our estimates through an error propagation analysis. Monte Carlo simulation was used to evaluate the effect on parameter and load uncertainty, which was characterized at the 10^{th}, 25^{th}, 50^{th}, 75^{th} and 90^{th} percentiles. Uncertainty happened to be greater for pH and benthic algae than total nutrients. Nearly 75% of all model realizations were below the stated pH criteria thus we can be confident that the proposed criteria would support uses regardless of boundary conditions or parameter uncertainty. For benthic algae, uncertainty is quite large thus a 5-year monitoring program was proposed to identify algal trends as the river moves closer to the proposed criteria. The low output variance in total nutrients was attributed to several factors including our decision to not perturbate nutrient loads in the analysis (i.e., they had already been adjusted in the nutrient addition scenarios), the fact that rate uncertainties are less important to total nutrients than they are to specific soluble nutrient compounds, and finally that Bayesian inference techniques were used to narrow the range of allowable rate distributions from the broader literature array. Consequently, we believe the proposed criteria should remain unaltered.

Finally, we evaluated our nutrient criteria against other studies from the scientific literature. Overall they compared favorably and, if anything, inclined toward the higher reported concentrations. This was expected given the known extent of depth and light limitation in the river. Likewise, the modeled response (e.g., change in pH as a function of increasing P concentration) exhibited Monod-type non-linearity. Generally there was an initial phase where water quality changed linearly with each incremental increase in nutrients, an inflection point where this change subsided, and then a less responsive phase where additional nutrients altered water quality only slightly. Most of the criteria fell very near this change point. Consequently the river is at first quite sensitive to nutrient pollution and is then less sensitive thereafter. This knowledge will be helpful in proactive management of the river to maintain high quality waters.

Lastly, we conclude with a few remarks about the effectiveness of using models for criteria development. The greatest benefit encountered in this study was the added ability to directly quantify the relationship between nutrients and eutrophication response using the model. For example, we were able to evaluate multiple ecological endpoints of concern within a single simulation (e.g., DO, algal biomass, pH, etc.) which we would not have been able to do with statistical or data-based empirical approaches. Similarly, the complex interactions between light, algal assimilation and growth, and nutrient recycling were all much clearer after application of the model than before. Several noteworthy things were also identified specific to large rivers and models.

First, the eutrophication response is reach-specific and can be buffered through a number of mechanisms. In the Yellowstone River, longitudinal changes in turbidity and depth were the most important factors impeding nutrient response. This required multiple criteria to address localized conditions. Second, the lateral variation in biological response is important. Localized regions of high productivity necessitate that nutrient management plans protect not only the water column, but also specific regions of the river amenable to recreation or juvenile fish propagation. Finally, the use of models provides a way to gage the response between available nutrient supply and total nutrients (after recycle/mineralization) which is something that can only be done within a mechanistic framework. Consequently, we feel there is good merit for the use of modeling tools in the future and we recommend the approach as a suitable alternative for States or Tribes assessing numeric nutrient standards on large rivers elsewhere.

ACKNOWLEDGEMENTS

Recognition should be given to the citizens and taxpayers of Montana. Without their commitment to water quality, this endeavor would not have been possible. The Nutrient Work Group should also be thanked. They are a diverse group of stakeholders who afforded us a resonant sounding board for critical project review. Many DEQ staff contributed to the project. Included (in alphabetical order) were: Mark Bostrom, Bob Bukantis, Jonathan Drygas, Erik Makus, Al Nixon, Michael Pipp, Rosie Sada, Mike Stermitz, and Andy Welch. Their work is a credit to the agency. Special thanks must also go to Dr. Steve Chapra at Tufts University. His review of modeling work, hours of collaboration, and constructive criticism were a great benefit to the overall integrity of the project. In addition, two anonymous reviewers from the U.S. Environmental Protection Agency's Nutrient Scientific Technical Exchange Partnership and Support (N-STEPS) group provided critical and valued review. Finally, a number of other organizations also cooperated. They are listed below and their efforts were greatly appreciated:

- Army Corps of Engineers
- Buffalo Rapids Irrigation District
- Bureau of Reclamation
- City of Miles City
- City of Forsyth
- USDA Ft. Keogh Agricultural Experiment Station
- Montana Bureau of Mines and Geology
- Montana Department of Natural Resources and Conservation
- Academy of Natural Sciences of Philadelphia
- U.S. Geological Survey
- Yellowstone River Technical Advisory Committee of the Yellowstone River Conservation District Council

Work on this effort was competed cooperatively by the Water Quality Standards and Water Quality Modeling sections of DEQ in support of the Montana water quality standards program.

ACRONYMS

Acronym	Definition
7Q10	7-Day 10-Year Low-flow Condition
14Q5	14-Day 5-Year-Lowflow Condition
ACOE	Army Corps of Engineers
AFDM	Ash Free Dry Mass
APT	Airport
ARM	Administrative Rules of Montana
ASABE	American Society of Argriculture and Biological Engineers
ASTM	American Society of Testing and Materials
AT2K	Algae Transect2K
BAL	Benthic Algae
BMP	Best Management Practices
BOD	Biochemical Oxygen Demand
BOR	Bureau of Reclamation
BRGM	Bureau of Reclamation Buffalo Rapids Glendive AgriMet Station
BRID	Buffalo Rapids Irrigation District
BRTM	Bureau of Reclamation Buffalo Rapids Terry AgriMet station
CBOD	Carbonaceous Biochemical Oxygen Demand
CCC	Criteria Continuous Concentrations
CDF	Cumulative Density Functions
CFR	Code of Federal Regulations
CI	Confidence Interval
COV	Coefficient of Variation
C_T	Total Inorganic Carbon
CWA	Clean Water Act
DBP	Disinfection By-products
DEM	Digital Elevation Model
DEQ	Department of Environmental Quality (Montana)
DMR	Discharge Monitoring Report
DNRC	Department of Natural Resources & Conservation
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
DS	Downstream
DVT	Diversion
EPA	Environmental Protection Agency (US)
EQIP	Environmental Quality Initiatives Program
ET	Evapotranspiration
EWI	Equal width integrated
FAS	Fishing Access Site
FBOD	Fast CBOD
FWS	Fish & Wildlife Service (US)
GIS	Geographic Information System
GWIC	Groundwater Information Center
ICIC	Integrated Compliance Information System
ICIS	Integrated Compliance Information System

Acronym	Definition
IR	Integrated Report
ISS	Inorganic Suspended Solids
LIDAR	Light Detection and Ranging
MBMG	Montana Bureau of Mines and Geology
MCA	Montana Codes Annotated
MCS	Monte Carlo Simulation
MDOT	Montana Department of Transportation
MDT	Montana Department of Transportation
MPDES	Montana Pollutant Discharge Elimination System
MSU	Montana State University
NAIP	National Agriculture Imagery Program
NAWQA	National Water Quality Assessment Program
NB	Nuisance Biomass
NCDC	National Climatic Data Center
NLCD	National Land Cover Dataset
NPDES	National Pollutant Discharge Elimination System
NPS	Nonpoint Source
NRCS	National Resources Conservation Service
NREL	National Renewable Energy Laboratory
NRIS	Natural Resource Information System (Montana)
NTR	National Toxic Rule
NTU	Nephelometric Turbidity Units
NWIS	National Water Information System
NWS	National Weather Service
PANS	Philadelphia Academy of Natural Sciences
PB	Peak Biomass
PDF	Probability Density Function
PFD	Photon Flux Density
PHYT	Phytoplankton
POC	Particulate Organic Carbon
PORG	Organic Phosphorus
PWS	Public Water System (or Supply)
QA	Quality Assurance
QAPP	Quality Assurance Project Plan
RE	Relative Error
RMSE	Root Mean Squared Error
RWIS	Road Weather Information System
SAP	Sampling and Analysis Plan
SC	Sensitivity Coefficient
SCE	Shuffled-complex Evolution
SIN	Soluble Inorganic Nitrogen
SOD	Sediment Oxygen Demand
SRP	Soluble Reactive Phosphorus
SSC	Suspended Sediment Concentration
STORET	EPA STOrage and RETrieval database
SWSTAT	Surface Water Statistics Software
TDG	Total Dissolved Gas

Acronym	**Definition**
TDS	Total Dissolved Solids
TMDL	Total Maximum Daily Load
TMY	Typical Meteorological Year
TN	Total Nitrogen
T_{NB}	Time to Nuisance Biomass
TOC	Total Organic Carbon
TP	Total Phosphorus
T_{PB}	Time to Peak Biomass
TSS	Total Suspended Solids
USDA	United States Department of Agriculture
USGS	United States Geological Survey
VBA	Visual Basic for Applications
VNRP	Voluntary Nutrient Reduction Program
VSS	Volatile Suspended Solids
WDM	Watershed Data Management
WRS	Water Resource Surveys
WTP	Water Treatment Plant
WWTP	Waste Water Treatment Plant

CONVERSION FACTORS

Length

1 centimeter (cm)	= 0.394 inches (in)
1 meter (m)	= 3.2808 feet (ft)
1 mile (mi)	= 1.609 kilometer (km)

Area

1 square kilometer (km^2)	= 0.386 mi^2
1 hectare (ha)	=10,000 m^2

Volume

1 cubic meter (m^3)	= 35.313 cubic feet (ft^3)
1 cubic meter (m^3)	= 1,000 liters

Velocity

1 meter per second ($m\ s^{-1}$)	= 3.2808 feet per second ($ft\ s^{-1}$)

Mass

1 kilogram (kg)	= 2.2046 pounds (lb)

Concentration

1 $mg\ L^{-1}$	= 1,000 $\mu g\ L^{-1}$

Heat

1 langley per day ($ly\ d^{-1}$)	= 1 $cal\ cm^{-2}\ d^{-1}$

Temperature

Degrees Celsius (°C)	= 5/9 *(Fahrenheit -32)

1.0 INTRODUCTION

Detailed field studies and associated modeling were conducted on a 232.9 km (144.7 mile) segment of the lower Yellowstone River in eastern Montana, extending from Forsyth to Glendive, MT, to assess the feasibility of developing large river numeric nutrient criteria using a mechanistic water-quality model. Specifically, the one-dimensional QUAL2K model (Q2K) and a new model, AlgaeTransect2K (AT2K), were applied in conjunction with literature based approaches to derive nutrient concentrations capable of attaining and maintaining the river's beneficial uses. Goals and objectives of the study were as follows: (1) to assess whether numeric models are appropriate for numeric nutrient criteria development in large river settings, (2) to establish whether modeled criteria are consistent with other nutrient endpoint techniques, and (3) report our findings such that other States or Tribes can make informed decisions about these techniques for large rivers in their regions. Pending success, the methodology could then be transferred elsewhere.

This document describes the outcome of the above approach for the lower Yellowstone River between Forsyth and Glendive, MT (Waterbody IDs MT42K001_010 and MT42M001_012). Details on the project background, data compilation and assessment, materials and methods, model development, results and discussion, and critical low-flow simulations are described herein.

1.1 BACKGROUND OF NUMERIC NUTRIENT CRITERIA DEVELOPMENT IN MONTANA

Eutrophication (i.e., from excess nitrogen and phosphorus enrichment) has been a major water quality problem in the U.S. and abroad for many years (Smith et al., 1999; EPA, 2000b). This is well illustrated by the fact that the U.S. Environmental Protection Agency (EPA) initiated a national eutrophication survey of streams just shortly after its creation in the early 1970s (Omernik, 1977). Regulatory approaches for the control of water pollution had been in place since 1948 (through the Federal Water Pollution Control Act; Pub. L. No. 80-845, 62 Stat. 1155) (Andreen, 2004), however requirements for nutrients were only addressed later in 1972, through the Federal Water Pollution Control Act (33 U.S.C. §1251 et seq., 40 CFR). Better known as the Clean Water Act (CWA), legislative controls were finally provided to address eutrophication in our nation's waters and ensure that they remain fishable and swimmable (i.e., encompassing recreation and all other beneficial uses).

The Montana Department of Environmental Quality (DEQ) is the delegated federal authority required to implement and enforce CWA regulations within our state. While there are many CWA provisions (far beyond the scope of this document), this document specifically addresses Section 304(a). As required therein, states must identify ambient water quality criteria recommendations for their waters to limit impairments, including those from excess nutrients. DEQ currently uses narrative criteria which aim to limit nuisance conditions through codified statements that describe a desired condition. More recently, we have been requested to provide numeric quantification of these limits (EPA, 1998). Guidance was given by EPA to implement this mandate (EPA, 2000b), and flexibility was allowed to first outline a proposed approach and schedule. It was accompanied by regionally-based interim criteria (EPA, 2000a) that we feel are much too generalized (for large rivers) and simply are not defensible.

DEQ first submitted a nutrient criteria development plan to EPA in 2002. Since then, we have made good progress in developing numeric nutrient criteria for wadeable streams and small rivers by integrating stressor-response and reference-based approaches (Suplee et al., 2007). We are only one of 14 states to have done so (EPA, 2008a). Defensible approaches for large rivers are also necessary, but are not well established (Smith and Tran, 2010; Weigel and Robertson, 2007). Consequently, we propose a modeling approach that will benefit future efforts. Our intent is to explore the proposed methodology, develop criteria if appropriate, and better address the state-wide and national deficiency in large river numeric nutrient criteria development techniques.

1.2 Mechanistic Models & Montana's Proposed Large River Approach

Montana's proposed approach for large river numeric nutrient criteria development is shown in **Figure 1-1**. It includes sequentially: (1) identification and selection of an appropriate water quality modeling tool to use in large river settings, (2) data collection to support this tool, (3) application of the chosen model to a site-specific river reach, (4) subsequent evaluation of critical nutrient concentrations that impair beneficial uses in the model, (5) model reliability and uncertainty analysis, (6) literature and peer review of the findings, and (7) criteria establishment.

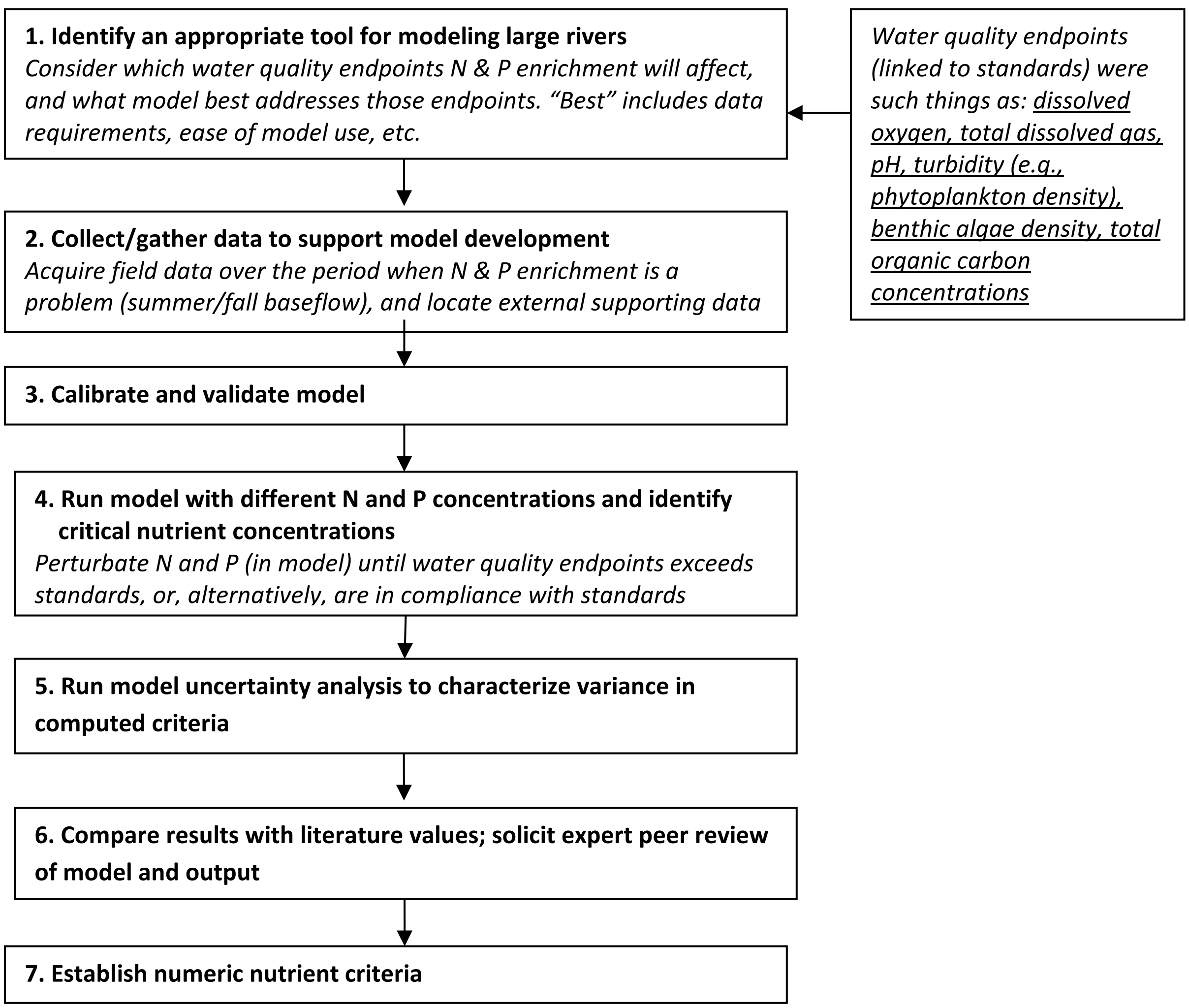

Figure 1-1. Montana's proposed approach for large river numeric nutrient criteria development.
Our large river approach is very similar to U.S. EPA (2000b), however, it relies heavily on modeling due to a number of limitations inherent in the EPA approach. These include, but are not limited to: lack of suitable populations required for establishing benchmarks via reference-based approaches, poor empirical correlations between ambient nutrient concentrations and algal responses, cross-correlations between different stressors, and limited information on algal and associated biological effects.

The rationale for model development is not to have a "black box" from which nutrient criteria are mysteriously manufactured. Rather it is to help us more thoroughly understand the linkages between nutrients (cause) and eutrophication (effect), and then relate those ecological responses to beneficial use attainment or non-attainment. Finally we wish to use this information to better manage our streams and rivers. The approach is therefore of good intent, robust, and absent of many of the criticisms of the EPA approach identified by others (Hall et al., 2009).

1.3 Why Use a Water Quality Model

One might ask why we are proposing a water quality model if other methods already exist to quantify nutrient limits (e.g., empirical statistical approaches). We have already addressed this to some extent, but to reinforce the Department position, other methods are too regionalized or rely too much on scarce reference-river datasets, historical or current impacts of anthropogenic stresses, or poorly transferrable empirical relationships between nutrient concentrations and biota to be practical for water quality management in Montana [similar to that pointed out by Weigel and Robertson (2007)]. Similarly, streamside mesocosm or other data-based approaches are not suitable for rivers which are primarily deep and turbid and have large underwater areas unsuitable for significant algae colonization.

Process-based models are a suitable alternative as they use well-established physical relationships between nutrient availability and algal uptake kinetics, and other site-specific dependencies such as light, streamflow, temperature, etc. to elicit tangible relationships between nutrient concentrations and biological or water quality responses (ecological endpoints). They are well suited to analytical determinations, and are particularly useful in large rivers where complex relationships might otherwise be difficult to ascertain due to confounding environmental factors. Mechanistic models also require less data collection than empirical methods because the field and laboratory work has already been done to establish the model theoretical construct. Finally, they can be used outside of the conditions for which the model was originally developed making them instructive for deterministic or predictive calculations.

Consequently, there are numerous advantages to our proposed methodology. The fact that many regulatory managers, permit writers, wasteload investigators, and total maximum daily load (TMDL) planners rely on models is further affirmation. Models have nearly 85 years of application in water quality management and environmental decision support (Chapra, 2003; Thomann, 1998) for example. More recently, the role of predictive models in criteria development has been detailed in the literature (Carleton et al., 2005; Carleton et al., 2009; Reckhow et al., 2005). The Montana approach is then of great benefit for both local and national audiences.

1.4 Large River Definition and Scope

DEQ defines a large river as one that is un-wadeable during the summer and early fall baseflow period. Essentially all rivers in the state meeting this definition will be considered for criteria development via modeling. Techniques to distinguish whether a river is wadeable or non-wadeable, as well as what constitutes the base flow period, have been outlined by Flynn and Suplee (2010). Eight rivers under management by DEQ are non-wadeable (or large) based on the relationship between their wadeability index and baseflow annual discharge[1]. These include the Bighorn, Clark Fork, Flathead, Kootenai, Madison, Missouri, South Fork of the Flathead, and Yellowstone rivers (**Table 1-1**).

Table 1-1. Large or non-wadeable rivers in Montana.

River Name	Segment Description
Big Horn River	Yellowtail Dam to mouth
Clark Fork River	Bitterroot River to state-line
Flathead River	Origin to mouth
Kootenai River	Libby Dam to state-line
Madison River	Madison Dam to mouth
Missouri River	Origin to state-line
S F Flathead River	Hungry Horse Dam to mouth
Yellowstone River	State-line to state-line

Since the Yellowstone is the most prominent river, being non-wadeable from state-line to state-line, it was a good candidate for our water quality model based criteria approach. Its length poses difficulties though as multiple criteria must be developed over its extent due to longitudinal changes in eutrophication response (e.g., from shifts in streamflow, temperature, light attenuation, etc.). Consequently, we chose to evaluate site-specific criteria on a segment (or case-by-case) basis. This will be done until either a sufficient understanding of behavioral response of nutrients in large rivers can be understood, or until available data can be pooled such that reasonable conclusions can be made. We consider two segments of the lower Yellowstone River in this work. Further detail on these segments can be found in **Section 4.0**.

1.5 Beneficial Uses of Montana Rivers and How Criteria Protect Them

Beneficial uses describe the societal or ecological characteristics that directly or indirectly contribute to human welfare (Biggs, 2000b; Stevenson et al., 1996). In Montana, such uses are defined by the Administrative Rules of Montana (ARM). For large rivers (use class B-1, B-2, or B-3) the following activities are included: (1) drinking, culinary, or food processing purposes (after conventional treatment); (2) bathing, swimming and recreation; (3) propagation of salmonid or non-salmonid fishes (depending on water use class) plus support of other aquatic life, waterfowl, and furbearers; and (4) agricultural and industrial water supply (17.30.601-17.30.646, 1999). Because rivers must be exploited for societal use (Benke and Cushing, 2005), nutrient criteria (or standards) are the regulatory limits that ensure the waterbody is not harmed beyond acceptable limits.

A hypothetical example of a nutrient criterion is presented in **Figure 1-2.** If total phosphorus (TP) concentrations of 0.03 mg L^{-1} are needed to protect recreational uses from nuisance algae (i.e., during the growing season), we see that a criteria exceedance (or excursion) was caused by a waste water treatment plant (WWTP) pulse load during the summer of 1993. The rest of the summer, criteria were met. Simply stated, anything below the criteria is protective of the use and anything above it is indicative of impairment.

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[1] Wadeability thresholds were identified from a compilation of 54 different rivers and 157 sites. A baseflow annual discharge of 1,500 ft^3 s^{-1} (42.5 m^3 s^{-1}), depth of 3.15 ft (0.96 m), or wadeability index of 7.24 ft^2 s^{-1} (0.67 m^2 s^{-1}) constitutes a non-wadeable river segment.

By default then, criteria are designed to measure beneficial use attainment. Consequently, they should be well articulated, good predictors of an anticipated water quality condition, and easy to measure (Reckhow et al., 2005). We intend on addressing each of these requirements for the criteria determination on the Yellowstone River.

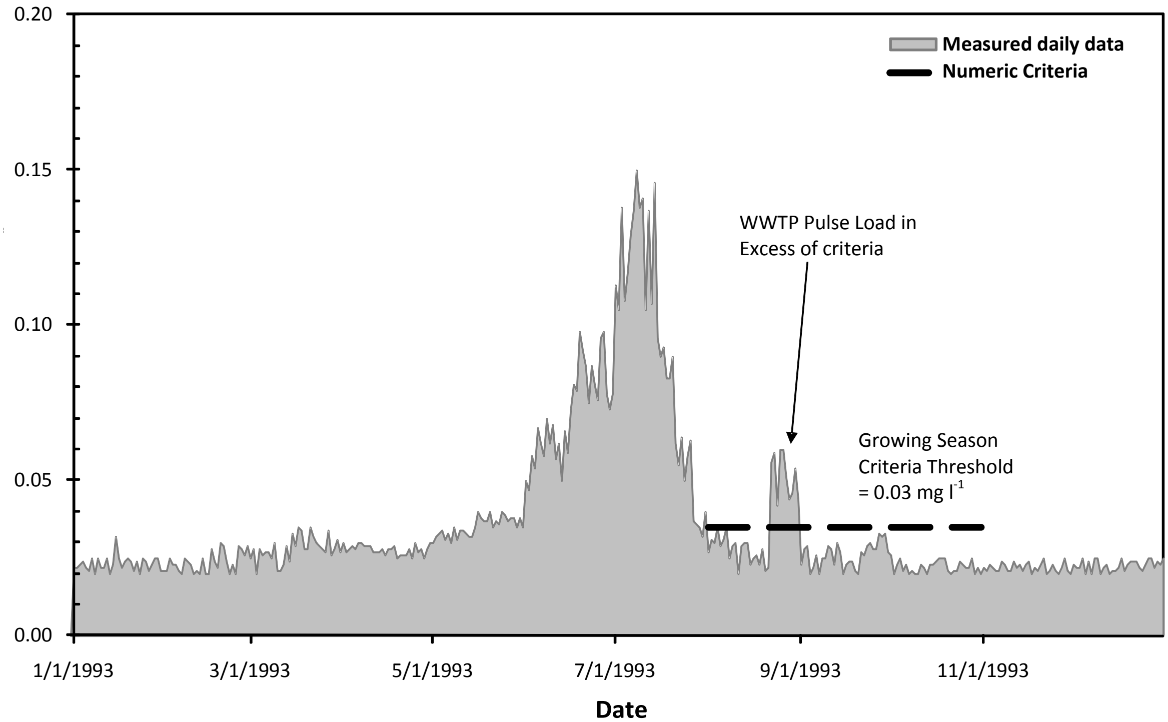

Figure 1-2. Hypothetical example of numeric nutrient criteria for total P in a Montana river. Nutrient levels in excess of the proposed criteria are indicative of beneficial use impairment. Concentrations below the criteria would support their intended uses. This hypothetical example illustrates probable impairment due to a WWTP pulse load during summer.

1.6 DOCUMENT OUTLINE

Throughout the remainder of this report, we build upon the basic tenants of this chapter. This includes a review of the science of eutrophication (including topics specific to large rivers) (**Section 2.0**), regulatory approaches for the control of eutrophication (**Section 3.0**), and then site-specific data compilation and modeling work specific to the Yellowstone River (**Sections 4.0-15.0**). Because the depth of some of these discussions are beyond the interest of some readers, specific topics applicable to each numbered box in **Figure 1-1** are provided below:

- Box 1 (identification of an appropriate model): **Section 5.0**
- Box 2 (data collection and literature review to support modeling): **Sections 6.0, 7.0**
- Box 3 (model calibration and validation): **Sections 8.0, 9.0, 10.0, 11.0**
- Box 4 (model nutrient-addition scenarios): **Sections 12.0, 13.0**
- Box 5 (uncertainty analysis): **Section 14.0**
- Box 6 (comparisons between the model results and other methods): **Section 15.0**
- Box 7 (establishment of numeric criteria): **Sections 12.0, 13.0, 14.0, 15.0**

In other words, for those interested only in criteria development and results, **Sections 12.0, 13.0, 14.0, and 15.0** will suffice. However, for those who prefer in-depth technical details about modeling, assumptions, background data and supporting files, and associated documentation, **Sections 5.0, 6.0, 7.0, 8.0, 9.0, 10.0, and 11.0** should be reviewed. The combined detail of the documentation is sufficient such that an independent reviewer, who wishes to either reproduce the findings, or conduct critical analysis or review of its contents and conclusions, can do so.

2.0 THE PROBLEM OF EUTROPHICATION

A basic understanding of eutrophication is fundamental to understanding criteria development. We recommend review of **Section 2.0** of Suplee et al., (2008) for a complete summary of eutrophication in Montana's wadeable streams and small rivers. A more focused review on large rivers is presented here. Other valuable references include Hynes (1966) and Laws (2000).

2.1 HOW EUTROPHICATION AFFECTS LARGE RIVERS

Eutrophication causes a variety of water quality problems in flowing waters such as nuisance algal growth, altered aquatic communities, and undesirable water-quality changes that impair beneficial uses (Dodds et al., 1997; Dodds, 2006; Freeman, 1986; Welch, 1992). Elevated or nuisance algal levels are most notorious (**Figure 2-1**), and the green algae *Cladophora spp.* in particular has benefited from excess nutrients in lotic systems worldwide (Dodds, 1991; Freeman, 1986; Robinson and Hawkes, 1986; Tomlinson et al., 2010; Whitton, 1970; Wong and Clark, 1975). Many other water quality problems are also associated with eutrophication. Those most commonly experienced in river environments are shown in **Table 2-1** (Smith et al., 1999). They are disruptive to both humans and aquatic inhabitants.

Table 2-1. Water quality problems associated with nutrient enrichment.

Human Impacts[1]	Aquatic impacts[1]
1. Taste and odor problems 2. Reduced water clarity 3. Blockage of intake screens and filters 4. Disruption of flocculation and chlorination processes at water treatment plants 5. Increased numbers of disinfection by-products (which are carcinogenic) 6. Restriction of swimming, boating, and other water-based recreation 7. Fouling of submerged lines and nets 8. Reduced property values and amenity 9. Tourism losses	1. Harmful diel fluctuations in pH and dissolved oxygen 2. Increased algal biomass 3. Changes in species composition of algae 4. Macrophyte over-abundance 5. Reduction in habitat for macroinvertebrates and fish especially in near-shore margins 6. Increased probability of fish kills 7. Toxic algae (more common with reservoir influence) 8. Commercial fishery losses

[1]From Smith et al., (1999) and Dodds et al., (2009).

2.1.1 Human and Societal Effects

The human and societal effects of eutrophication are notable. Common drinking water problems include taste and odor problems. Other health related concerns include elevated post-treatment disinfection-by-products (DBP), which are known or suspected carcinogens and result from increased organic material in the chlorinated drinking water treatment precursor pool (Palmstrom et al., 1988; Sadiq and Rodriguez, 2004), and greater accumulation of organochlorine pollutants (e.g., PCBs) in trout populations (Berglund et al., 1997; Berglund, 2003). Cyanobacterial blooms, which are rare, are also of great concern as they are toxic to both humans and animals (Vasconcelos, 2006). Given the prior concerns, the effects of eutrophication are not always trivial or simply a nuisance.

Nor is the impact of eutrophication constrained strictly to health or ecology. Dodds et al., (2009) estimate that societal damages from eutrophication (e.g., reduced property values; loss of recreational amenity; net economic losses for tourism and commercial use; and increased drinking water treatment) total $2.2 billion annually in the United States. Estimates are comparable to those made by Wilson and

Carpenter (1999) and Pretty et al., (2003). Costs associated with policy response, in-water preventative measures, or best management practices (BMP) are not included in these figures.

Figure 2-1. Example of nuisance *Cladophora spp.* growth in the Yellowstone River (August 2006).

2.1.2 Ecological Impacts

The ecological effects of eutrophication are also a concern. Changes in fish population density or size (Wang et al., 2007), shifts to less sensitive species (Hynes, 1966), and a plethora of other long-term chronic or acute ecological effects including loss of key or sensitive species or changed species composition have all been reported (Pretty et al., 2003). Altered diurnal DO and pH variation are the most common (Walling and Webb, 1992). If the impact is significant enough (i.e., fluctuations become too severe) fish kills can occur (Welch, 1992).

Aquatic insects or macroinvertebrates are also affected. Taxa shifts have frequently been reported in response to increasing enrichment. Sensitive macroinvertebrates such as mayflys (*Ephemeroptera*), stoneflys (*Plecoptera*) and caddisflys (*Trichoptera*) tend to prefer clean water with low nutrient concentrations (i.e., without extreme daily DO oscillations) while midge species (chironomids) tend to be abundant in heavy polluted water (Hilsenhoff, 1987; Hynes, 1966; Lenat and Penrose, 1996; Wang et al., 2007). In such systems, macroinvertebrate density and biomass tend to increase in relation to enrichment, yet sensitive species diminish (Gücker et al., 2006).

2.1.3 Other Considerations

Although not covered in previous discussions, it should not go unmentioned that eutrophication and small shifts in trophic status are not always harmful. For example, small increases of N and P have been shown to increase the productivity of fisheries by increasing fish biomass, fish abundance, and growth rates (deBruyn et al., 2003; Deegan and Peterson, 1992; Harvey et al., 1998; Perrin et al., 1987). This is exemplified in very nutrient poor watersheds. The Kootenai River in northwestern Montana is one such example where managers seek to increase productivity through nutrient additions (Holderman et al., 2009; Hoyle, 2003). Consequently, enrichment only becomes a problem when the effect of the increase in nutrient supply is undesirable.

2.2 Factors that Influence Eutrophication in Large Rivers

A number of environmental factors influence the eutrophication response of large rivers. They differ primarily from their wadeable counterparts in available light, water depth, and other physical features such as velocity and substrate. These differences are highlighted below.

2.2.1 Light

Light is a photosynthetic requirement that governs the rate at which algae grow (Hill, 1996). It is far less abundant in large rivers than wadeable streams, which is primarily due to increases in both turbidity and water depth. Factors that contribute to the influence of such things include terrestrial vegetation adjacent to the river (i.e. shading from riparian canopy cover), physical water depth, and adsorption and scattering properties of the medium. However, what sets wadeable and non-wadeable systems apart is the extent of light limitation. Larger rivers tend to be more light-limited than smaller waterbodies.

The amount of surface light reduction must be meaningful to accomplish any change in algal growth rate. Over 60% or more is suggested by some authors (Biggs, 2000a; Quinn et al., 1997). The extent to which this occurs in one of Montana's large rivers (the Yellowstone) is shown in **Figure 2-2** (top panel). Photosynthetically active radiation (PAR) diminishes quickly with depth and reaches a growth limiting threshold at approximately 0.5 meters. The spatial variation of attenuation is prominent for much of the channel transect (**Figure 2-2**, bottom panel) which leads us to conclude that light-limitation is a mechanism of primary interest for such systems. In this example, much of the bottom region is under strong light limitation whereas the near shore regions have ample light to stimulate algal growth. We define these regions as distinctly separate management zones. They are:

1. **The wadeable region** – which encompasses the shallow areas or margins of the river where the effect of eutrophication is most significant.
2. **The non-wadeable region** – consisting of the thalweg and deeper areas.

Management must be structured for each region thereby taking into account the lateral or spatial variation of light to characterize the response in the wadeable region where the overall water column response integrates the effect these two zones.

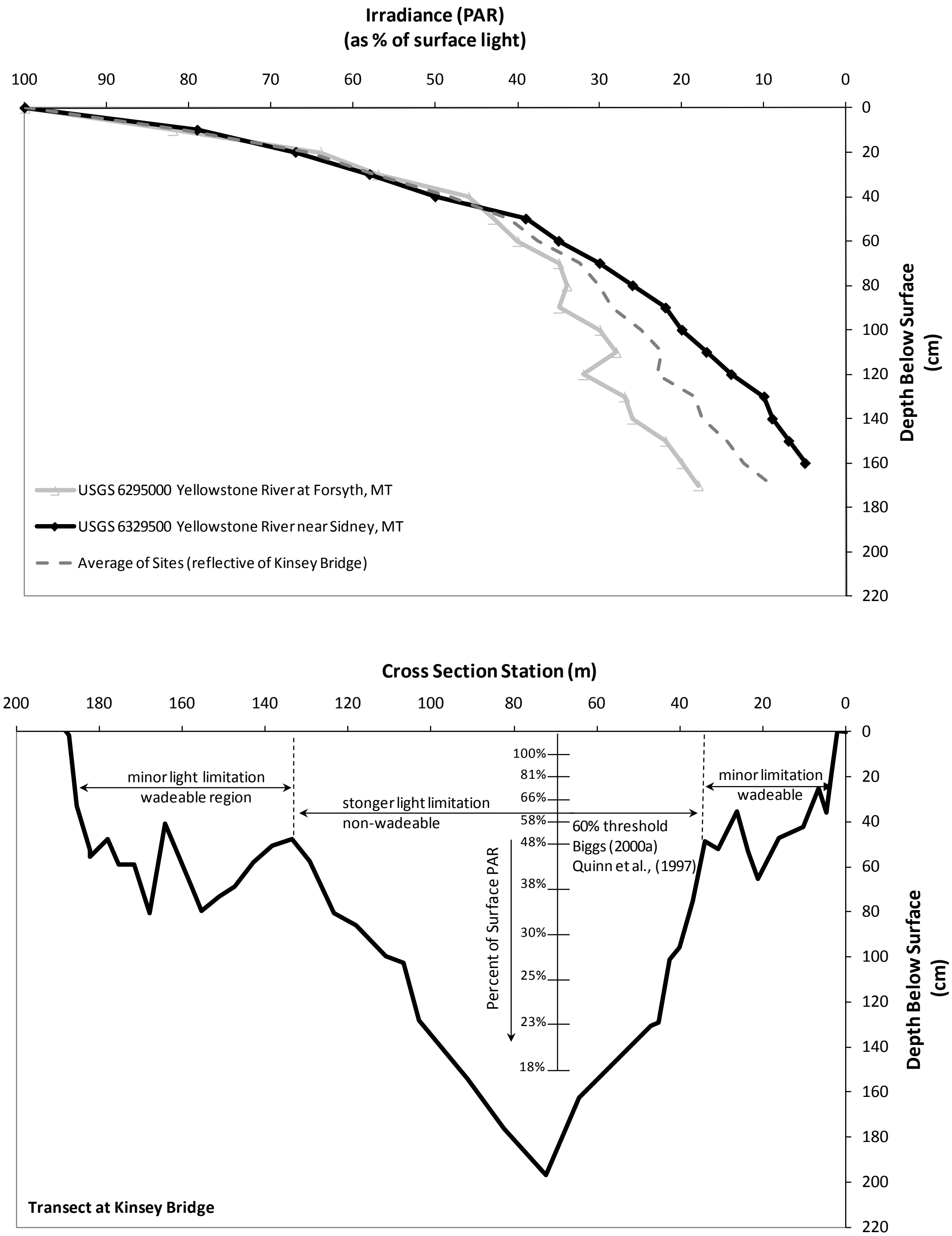

Figure 2-2. Light extinction in a Montana river and its lateral extent.
(Top panel) Light is quickly attenuated vertically in a Montana large river. In this example, only 50% of the surface light is available at 40 cm (15 inches) and drops to 25% at 120 cm (4 ft). Data from U.S. Geological survey (Peterson, 2009). (Bottom panel) A typical cross-section of that same river indicating the lateral extent of this variation. Cross-section shown for Kinsey Bridge near Terry, MT which is approximately the midpoint of the two irradiance stations (i.e., between Forsyth and Sidney).

The direct relationship between light and algal productivity for the same river is also apparent (**Figure 2-3**). In 2007, an influent tributary discharged highly turbid water and notably dampened productivity near one of our datasondes [days 1 through 3 as evidenced by the effect on the diurnal dissolved oxygen (DO) swing (delta, Δ)]. The river then returned to normal turbidity levels on day 4 and an upward shift in productivity ensued. Hence there is a good correlation between light and photosynthesis. It is important to note that high turbidities were needed to hasten the dampening effect in this instance (> 600 nephelometric turbidity units, NTUs). This exemplifies the notion that eutrophication response is muted when rivers become light limited (Hill and Harvey, 1990; Quinn et al., 1997; Rosemond, 1993).

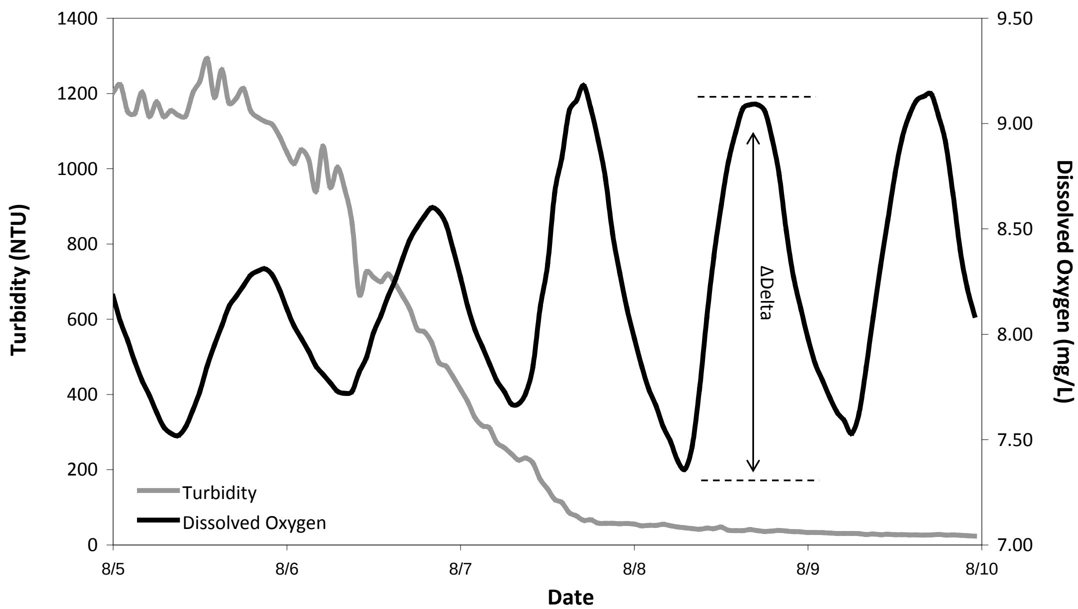

Figure 2-3. Influence of light attenuation on productivity in a Montana River.

Unusually high turbidity dampened DO changes over a three day period. During the 4th day, turbidity dropped, and the magnitude of the diel DO oscillations (i.e., productivity) markedly increased.

Light also influences the type of algal assemblage. According to Bayley et al., (2007), the dominant algae in the shallow lakes of the Canadian Boreal Plain switch from year to year according to environmental conditions. Phytoplankton tend to be dominant when the water is turbid whereas submerged aquatic vegetation proliferate when the water is clear. A similar phenomenon most likely occurs in large rivers, affirming the importance of light within the aquatic environment.

2.2.2 Velocity

Velocity is also important. A hyperbolic relationship exists between velocity and algal accumulation where incremental increases stimulate algal metabolism up to a point (by increasing nutrient transport to cells) and then ultimately causes decreases through drag and scour (Stevenson, 1996). Shear velocities increasing from 0 to 8 cm s^{-1} have been shown to allow larger mats and higher growth rates when compared to quiescent water because oncoming velocities force nutrients to reach algae cells at the base of the mat that might otherwise be starved of nutrients (Biggs, 2000a; Bothwell, 1989; Dodds, 1991). In contrast, higher velocities (50-70 cm s^{-1}) cause excessive drag and lead to reduced biomass via sloughing and scour (Biggs, 1996; Horner et al., 1990). Consequently, the range of 10-20 cm s^{-1} for diatoms, and 30-60 cm s^{-1} for filamentous algae seem to be most conducive for algal growth (Stevenson, 1996). Velocity can also influence early cell development and accumulation, which is slower in faster velocities. This effect is probably only minor compared to the other effects.

2.2.3 Substrate

Substrate is a final consideration. Roughness and texture influence biomass accumulation and several studies have shown that biomass concentrates more rapidly on rough surfaces such as rocks and bricks than on smooth surfaces such as tile (Cattaneou et al., 1997; Murdock and Dodds, 2007). Substrate motion and particle stability are also influential. Excessive movement can dislodge or damage algal cells upon impact (Peterson, 1996). Macroscopic algae are most susceptible to this kind of damage. Motile microalgae (i.e., those that can move) survive better than their sessile (fixed) counterparts in these settings (Burkholder, 1996). Finally, substrate size also affects growth dynamics. Large particles (i.e. boulders) increase algal settlement or emigration rates by slowing the velocity of the oncoming water whereas faster moving water (i.e., less roughness) has been shown to slow early cell development and accumulation. All of these are considerations affect large river algal accumulation. Accordingly, large rivers are probably most conducive to algae growth in shallow depositional zones where substrate stability is good and velocities are moderated by both substrate and river form.

2.3 Sources of Nitrogen and Phosphorus to Rivers

So far we have detailed only the environmental factors that influence eutrophication. The origin of nutrient supply should also be discussed. N and P enter aquatic systems in two ways, from: (1) the atmosphere and (2) the landscape. Natural sources (e.g., from rainfall, geochemical weathering erosion, etc.) can be exacerbated by human activity. Such anthropogenic sources are now believed to exceed natural sources on a global scale (Smith et al., 1999; Vitousek et al., 1997).

2.3.1 Atmospheric Sources

Atmospheric sources of nutrients are unavoidable and contribute a significant percentage to the N and P supply of aquatic systems. Nitrogen (as a gas) comprises approximately 78% of the atmosphere and thus its contributions are substantial. Atmospheric N requires reduction (e.g., to ammonium) by bacteria before it is biologically available (Stanier et al., 1986) however it can be directly deposited by both wet and dry deposition. P contributions also occur, but only from Aeolian (wind-based) transport. Nitrogen concentrations in rainfall approximate 400 µg L^{-1} in unpolluted regions of the world (Meybeck, 1982) while P depositional rates are approximately 0.05-0.1 gP m^{-2} yr^{-1} (Neff et al., 2008). Anthropogenic activity has increased the rate of accumulation of each. N accumulation is believed to be 10-100 times greater in urbanized settings than unpolluted regions (Vitousek et al., 1997) whereas P flux is 5 times higher than historical levels (Neff et al., 2008). Changes are believed to stem from a combination of activities including fossil fuel consumption, large-scale land disturbance, over-application of fertilizer, or use of nitrogen fixing crops (Smith et al., 1999).

2.3.2 Land-based Sources

Nutrients from the landscape can add to the rainfall input and consist of organic materials from forestland litter or duff accumulation (Triska et al., 1984), contributions from grassland or native ungulates (Frank and Groffman, 2010), or lateral accretion of organic material associated with streambank erosion. Geologic sources are important also. Dillon and Kirchner (1975) and Holloway et al. (1998) show that geochemical weathering can greatly contribute to N and P yields in some areas.

Minerals such as apatite (common in igneous rocks) contribute to orthophosphate while ammonia bearing mica and feldspars are readily oxidized to produce nitrate.

Human activity is probably the largest contributor of N and P to aquatic systems however. Point sources (e.g., waste water treatment plants) are the most conspicuous and have physically observable pipes that discharge directly to streams. However, nonpoint sources are widespread too, and can be equally, if not larger, sources of pollution than point sources. Urban runoff and sprawl, land clearing and conversion, agriculture, silviculture, riparian degradation, and streambank erosion are all examples (Hynes, 1969; Novotny and Olem, 1994; Porter, 1975; Smith et al., 1999). While such sources are most prevalent during runoff, they too can have year-round effects such as in the case of septic effluent migration or fertilizer leachate. Bioavailable forms are of greatest importance. The conversion of unreactive atmospheric N to ammonia salt to produce fertilizer through the Haber-Bosh industrial process is a major contributor (Smith et al., 1999; Vitousek et al., 1997). Advances in cheap energy and equipment technology have also greatly increased the disturbance trend in global nutrient cycling.

3.0 THE CONTROL OF EUTROPHICATION

Readers should refer to **Section 3.0** of Suplee, et al. (2008) for details on the state's past, current, and proposed approaches to eutrophication management in surface waters. To summarize, DEQ has regulated nutrients using narrative criteria. These require that, "*State surface waters must be free from substances attributable to municipal, industrial, agricultural practices or other discharges that will create concentrations or combinations of materials which are toxic or harmful to human, animal, plant, or aquatic life; or produce undesirable aquatic life*" [ARM 17.30.637(1)(d, e)].

To clarify, codified narrative statements such as above provide qualitative controls of harmful or undesirable conditions brought on by nutrients. However, because a narrative by definition lacks specificity and is open to interpretation, subjectivity is a potential concern. Adoption of numeric criteria will eliminate this fault and will provide readily measurable endpoints that are easier to monitor, regulate, and asses. Consequently, the criteria derived in this document closely reflect the spirit and intent of the narrative criterion, but also provide sufficient detail to make them of practical value.

3.1 TIME-PERIOD FOR NUTRIENT CONTROL

Past nutrient criteria development activities have focused on limiting the eutrophication response to a period when the impact would be most severe, such as during baseflow or the growing season (Dodds et al., 1997; Suplee et al., 2008). Our present work is no different, and required us to identify the critical time-period for nutrient control in large rivers within temperate regions. We considered the following:

- Water temperature, which needs to be warm enough for algae to grow.
- Light, which should be luminous enough that photosynthesis outpaces respiration.
- Streamflow, which needs to be at levels low enough that dilution and nutrient load assimilative capacity is greatly diminished.

From our analysis the growing season in Montana could potentially extend from April 1-October 31[2] (**Figure 3-1**) but is ostensibly shortened by a number of factors.

[2] This assumes a growth limiting threshold of 5˚C, similar to that of the nuisance algae genus *Cladophora* (Whitton, 1970). Data was evaluated from the following sites to make this conclusion: Missouri River at Toston, MT (06054500); Madison River below Ennis Lake near McAllister, MT (06041000); Yellowstone River near Livingston, MT (06192500); Clark Fork at Superior, MT (12353650); Flathead River at Columbia Falls, MT (12363000); and Flathead River at Perma, MT (12388700). Several rivers not actually classified as large rivers, but with similar character were also included to make the dataset more robust [e.g., Dearborn River near Craig, MT (06073500), Sun River near Vaughn, MT (06089000), Blackfoot River near Bonner, MT (12340000); and Bitteroot River near Missoula, MT (12352500)].

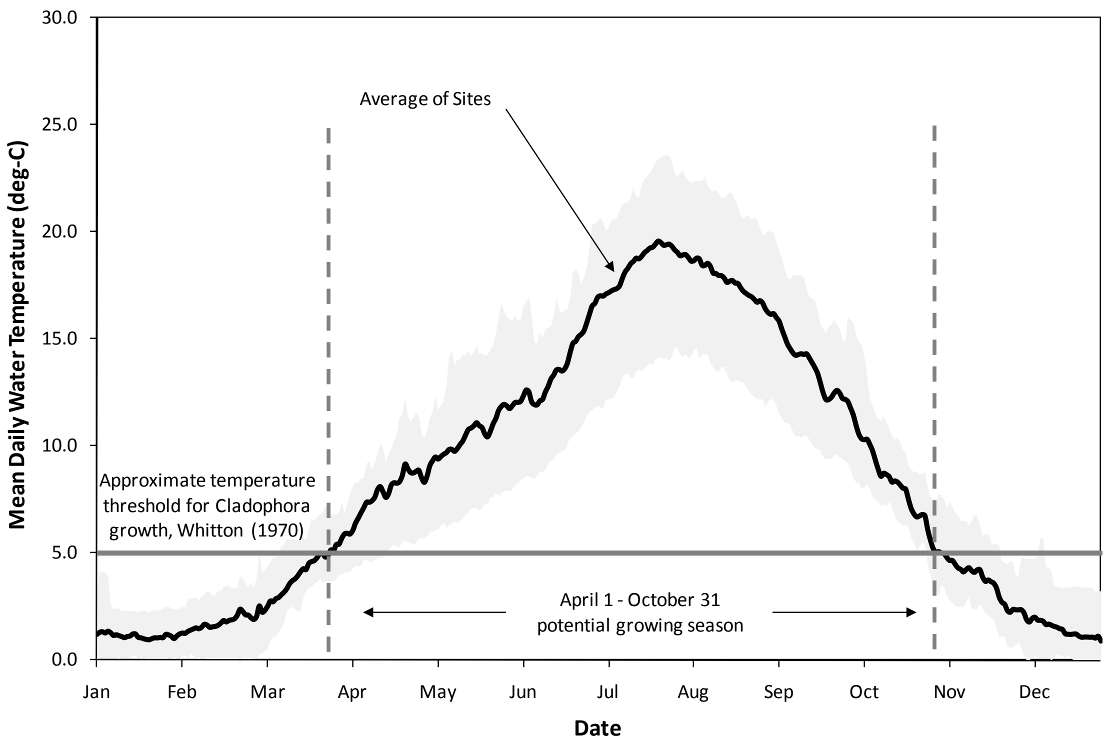

Figure 3-1. Plot of mean daily water temperature against algal growth limiting threshold.
Data includes a compilation of nine rivers in Montana with sufficient record to characterize long-term variation in water temperature. Sites include the Bitterroot, Blackfoot, Clark Fork, Dearborn, Flathead (2 sites), Missouri, Madison, Sun, and Yellowstone rivers. Temperatures above 5˚C from April to November are indicative of periods when algal growth could proliferate [as suggested in Whitton (1970)].

The actual growing season, however, is restricted by light-limitation during runoff. For example, the suspended sediment concentration in most Montana rivers during freshet is 100-200 mg L^{-1}. The effect on photosynthetic capacity can be approximated according to the Beer-Lambert law (**Equation 3-1**) where $PFD_{surface}$ and PDF_{depthz} = the photon flux density (PFD) at the surface[3] and bottom of the channel respectively, k_e = light extinction coefficient [m^{-1}, dependent on suspended sediment concentration[4] (SSC)], and where, z= mean hydraulic depth of the river[5] [m].

(Equation 3-1)

$$PFD_{depthz} = PFD_{surface} e^{(-k_e z)}$$

Optimal light conditions extend from approximately July 1-October 31 (**Figure 3-2**) assuming that intensities below the half-saturation constant would limit nuisance growth[6].

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[3] Surface irradiances taken from Lewistown, MT (Wilcox and Marion, 2008).
[4] Daily k_e calculated from daily SSC, where non-volatile solids were estimated from Ittekkot and Laane (1991) and partial extinction coefficients were as specified in Di Toro (1978).
[5] Mean daily hydraulic depth estimated using mean daily discharge and site rating curve.
[6] We used the midpoint of the range identified in Hill (1996) which was ~150 µmole quanta m^{-2} s^{-1}.

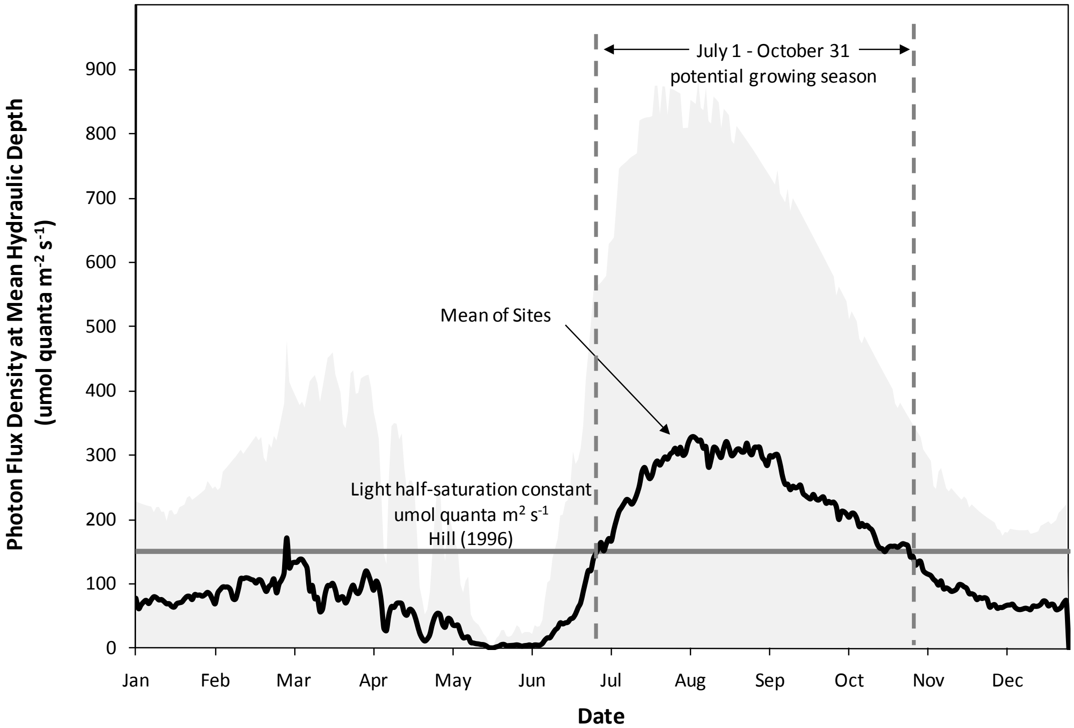

Figure 3-2. Plots of daily photon flux density against algal growth limiting threshold.
Based on a compilation of Montana Rivers[7].

Streamflow was the final consideration. From review of **Figure 3-3**, mean daily river flow in Montana reaches an inflection point on the falling limb of the hydrograph around August 1 which represents the transition from snowmelt to baseflow. A period of stability then follows which continues throughout the winter. Consequently, the critical period for nutrient control on the large rivers in Montana based on temperature, light, and streamflow constraints (**Figures 3-1**, **3-2**, and **3-3**) should occur over the period of August 1-October 31, when conditions are most apt to manifest nuisance responses. Monitoring, assessment, and modeling work should therefore target that period.

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[7] Only a handful of sites in the state had SSC data. These were the: Missouri River near Landusky, MT (06115200); Missouri River near Culbertson, MT (06185500); Yellowstone River at Billings, MT (06214500); and Yellowstone River at Forsyth, MT (06295000). To supplement this data, several other rivers were also included. These included: the Little Bighorn River near Hardin, MT (0629400); Clark Fork at Turah Bridge near Bonner, MT (12334550); Clark Fork above Missoula, MT (12340500); and the Blackfoot River near Bonner, MT (12340000).

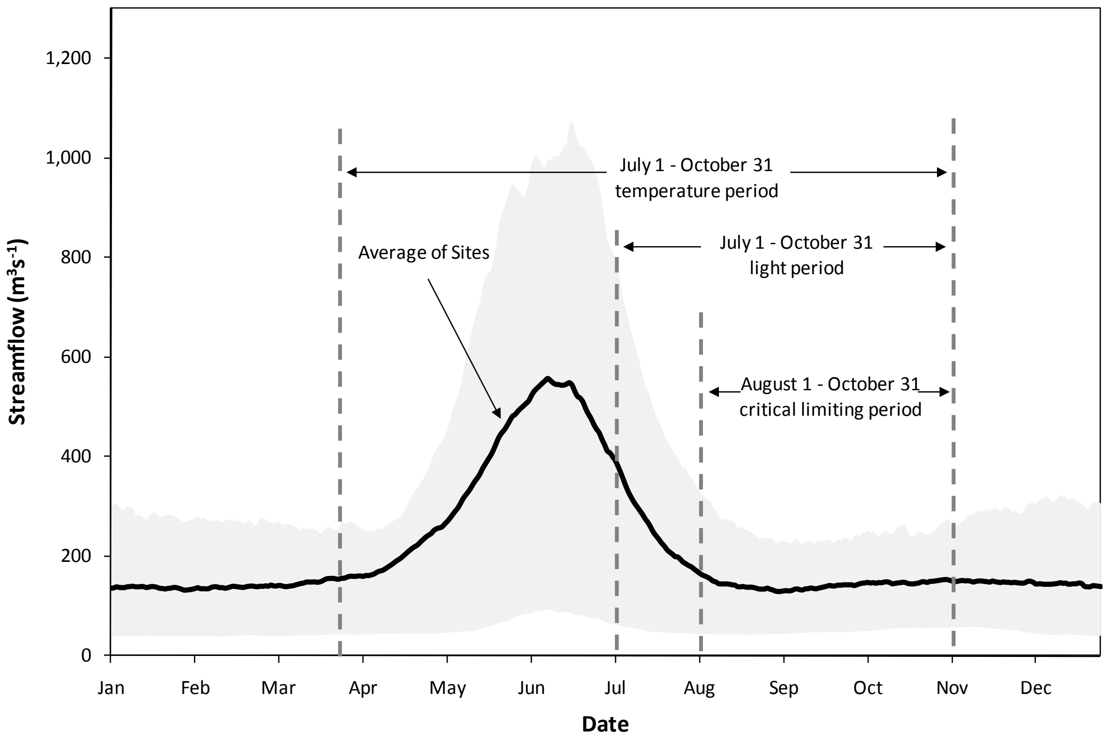


Figure 3-3. Streamflow hydrology and the critical period for large river criteria development. The most restrictive period relative to temperature, light, and streamflow is from August 1-October 31. Monitoring and assessment activities should target this timeframe for large river criteria development.

3.2 Total Nutrients as Recommended Criteria

Nutrient criteria necessitate that a specific target be achieved, for example what will be measured in the field to ensure compliance. Total nitrogen (TN) and total phosphorus (TP) are obvious choices as they have been shown to provide better overall correlations to eutrophication response than soluble nutrients (Dodds et al., 1997; Dodds et al., 2002; Dodds, 2006). They also coincide with the minimum acceptable nutrient criteria outlined by U.S. EPA (EPA, 2000b) and better lend themselves to ambient nutrient monitoring, permit compliance, and monitoring. Accordingly, DEQ will adopt these as targets. That said, water quality managers must use common sense when determining nutrient control strategies and permitted load limits. According to Liebig's law of the minimum, a single available resource (e.g., soluble N or P) will limit yields at a given time which implies that only a single nutrient could be considered in management (unless they are both close to limiting, i.e., co-limiting). However, in taking a single-nutrient approach to controlling eutrophication in rivers, one must give careful thought to the effects of the less-regulated nutrient on downstream beneficial uses, as nutrient limitation can quickly shift (Gibson, 1971). For this reason, both TN and TP criteria are recommended in this document.

3.3 Expected Difficulties with Numeric Nutrient Criteria Development

Because this is one of the first national efforts to derive model-based criteria for large rivers (Carleton et al., 2009; Reckhow et al., 2005; Smith and Tran, 2010; Weigel and Robertson, 2007), there will undoubtedly be difficulty. In our opinion, the major issues surrounding our approach include: (1) concerns about using water quality models for criteria development, (2) the spatial or geographic specificity of the criteria, (3) localized factors that cause deviations from proposed criteria, and (4) the achievability and affordability of the criteria. These items are briefly addressed below.

3.3.1 Concerns with Using Models

Water quality models are imperfect mathematical representations of complicated biogeochemical processes. This makes them easy to criticize. We recognize this, and debates regarding the use of models have been around for some time (Arhonditsis and Brett, 2004; Box and Draper, 1987). However, advancements in model theory, numerical methods, GIS capability, data visualization and display, and automation have made previous criticisms increasingly unfounded. Consequently, planning tools such as Q2K and others are being considered for regulatory purposes more and more, including criteria development (Carleton et al., 2005; Carleton et al., 2009). Use is advocated by decades of laboratory and field research [e.g., Streeter and Phelps, (1925) through current] with added latitude of sophisticated computing and highly accurate analytical data.

3.3.2 Longitudinal Variability of Proposed Criteria

Longitudinal variation in criteria is another important consideration in criteria development. River response to enrichment changes longitudinally as the physical continuum of the river is altered (Vannote et al., 1980). To simplify ecosystem structure and functional gradients into practical units for management, DEQ has used reach-indexing. Indexing effectively segments waterbodies at logical breakpoints according to major tributaries, shifts in river behavior, jurisdictional boundaries, or ecosystem or ecoregional boundaries. Descriptions of these breakpoints relative to the Yellowstone River are identified in **Section 4.4**.

3.3.3 Factors that Mitigate Eutrophication, Downstream Use Protection

Certain cases will also exist where localized or temporary environmental conditions mitigate the expected eutrophication response. As such, criteria for those locations will not be valid. Unusual flow events, uncharacteristic climatic conditions, or other atypical factors are examples that stretch the limits of criteria. These will be addressed on a case-by-case basis, if (or when) necessary.

Downstream requirements for lakes, reservoirs, and impoundments [75-5-306(2), §MCA], or interstate compacts or agreements were not considered. We recognize this to be a potential issue, but have no basis to do so with insufficient information on Lake Sacajawea (the first downstream reservoir located in North Dakota) for example, or for subsequent reservoirs downstream (or even the Gulf of Mexico for that matter). From a practical standpoint, low-flow criteria discussed herein are only a small percentage of the annual load to these waterbodies anyway, thus are likely insignificant.

3.3.4 Economics

Finally, it is apparent that the nutrient concentrations typically required to prevent unwanted aspects of eutrophication are relatively low when compared to current wastewater treatment technologies. The scientific literature indicates only small amounts of enrichment are needed to manifest large changes in stream productivity (Bothwell, 1989). Thus it is possible that endpoints determined through this modeling will be difficult to achieve. DEQ is developing implementation policies that will help stakeholders deal with this contingency on a case-by-case basis. Efforts are on-going, and are not detailed here. It is DEQ's general position that numeric nutrient criteria are ultimately achievable, even if time is needed for treatment technology to advance, and costs to come down.

4.0 CRITERIA DEVELOPMENT FOR THE YELLOWSTONE RIVER

Nutrient criteria development modeling work was initiated on the Yellowstone River in eastern Montana. It is a principal tributary to the Missouri River and one of the few remaining free-flowing rivers in the conterminous United States (Benke and Cushing, 2005). Identified by National Geographic magazine as “America's last best river” (Chapple, 1997), its prominence and importance make it an ideal candidate for criteria development testing. This is reinforced by the fact that a large proportion of Montana’s population lives along its banks.

4.1 WATERSHED DESCRIPTION

The headwaters of the Yellowstone River originate in Yellowstone National Park and drain 181,480 km^2 (70,100 mi^2) of the rugged Rocky Mountains and arid foothill prairies of the Northwestern Great Plains. The river flows 1,091 km (672.8 mi) through the landscapes of central Wyoming, southeastern Montana, and western North Dakota before reaching its endpoint with the Missouri River just east of the Montana-North Dakota state border (**Figure 4-1**). The criteria study reach (highlighted in red) extends from Forsyth to Glendive, MT. It is further detailed in **Section 4.2**.

Approximately 55% of the contributing watershed is part of the Northwestern Great Plains province whereas the remaining percentages come from the Wyoming Basin and Middle Rockies ecoregion (Zelt et al., 1999). Rangeland and brush are the dominant land cover types (combined 74%) while forest and agricultural lands comprise much of the remaining landscape (14 and 9% respectively) (Miller et al., 2004). The estimated basin population is 323,000 (Miller et al., 2004), and includes 38 municipal discharge facilities, 48 confined animal feeding operations, 78 stormwater permits, and 83 industrial facilities (in Montana alone). Those within Wyoming are not included.

Watershed relief is considerable and elevations span from 580-4200 meters (1,900-13,800 feet) (Miller et al., 2004). The variability in topography results in significant spatial differences in climate. Valleys are semiarid and temperate while the mountains are cold and moist. Average annual precipitation ranges from 150 mm (6 inches) to over 1,500 mm (60 inches) (Miller et al., 2004) while air temperatures fluctuate between -40˚C and 38˚C annually (-40˚F to 100˚F). Regional climate and seasonal regimen are determined by the interaction of air masses originating in the Gulf of Mexico, northern Pacific Ocean, and the Arctic regions (Zelt et al., 1999). Gulf air is prevalent in the spring and summer months while Pacific and Arctic air occur in the fall and winter.

Water yield comes principally from high elevation snowmelt runoff from the Absaroka-Beartooth, Wind River, and Bighorn mountain ranges (Thomas and Anderson, 1976). Runoff is second only to the Clark Fork River in Montana and mean annual streamflow at USGS 06329500 Yellowstone River near Sidney, MT is 365 $m^3 s^{-1}$ (12,900 $ft^3 s^{-1}$). Annual peaks approach 1,200 $m^3 s^{-1}$ (42,200 $ft^3 s^{-1}$) and low flows are near 143 $m^3 s^{-1}$ (5,060 $ft^3 s^{-1}$) (McCarthy, 2004). Both are typical of the project site. Major contributing tributaries include the Clark’s Fork of the Yellowstone River, Bighorn River, Tongue River, and Powder River. All originate from the south and west, and most are regulated by reservoirs.

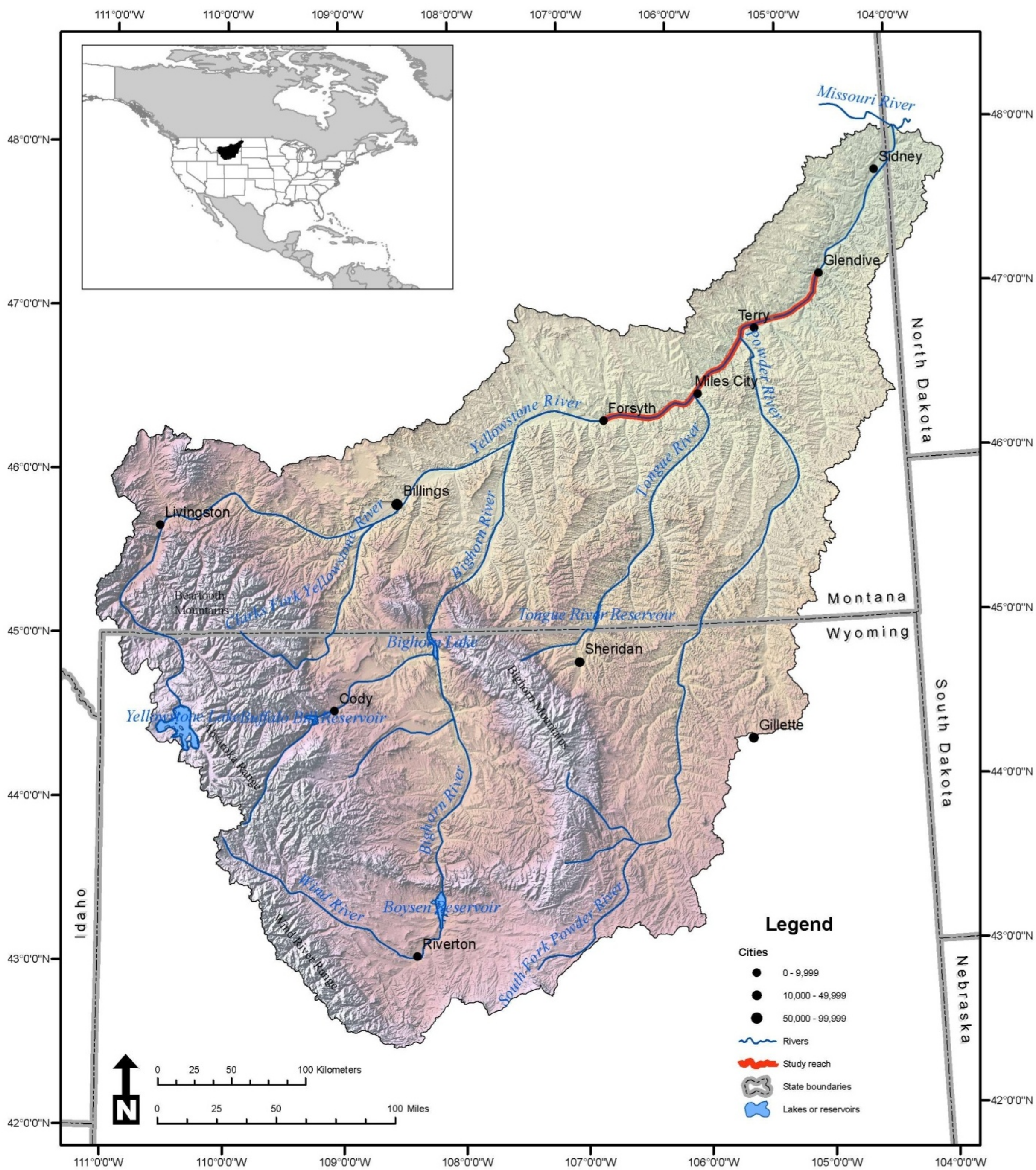


Figure 4-1. Yellowstone River area watershed in Montana, Wyoming, and North Dakota.
The reach evaluated in this study is shown in red.

4.2 Lower River Study Area

The focus of the modeling was on the lower part of the Yellowstone River between Forsyth and Glendive, MT. The reach is 232.9 km (144.7 miles) long and is most easily accessed by I-94 which parallels the river (**Figure 4-2**). The Highway 59 Bridge (at Forsyth) and Bell Street Bridge in Glendive designate the upper and lower study limits. Physiography is characteristic of the Great Plains ecoregion with expansive rolling hills and prairie and dissected and erodible topography (Smith et al., 2000; Zelt et al., 1999). Topographic relief is minimal, typically less than 150m (Zelt et al., 1999), limited mainly to the badlands east and south of Glendive (Smith et al., 2000). The rest of the reach contains gently sloped topography that has developed in the easily erodible shales of the region (Zelt et al., 1999).

River morphology is predominantly single thread with occasional braided channels (Benke and Cushing, 2005). The river has a wide and well armored low-flow channel and then a fairly expansive near-channel disturbance zone from annual flooding. Several natural bedrock grade controls exist at key locations which prevent major channel adjustments (AGDTM, 2004). Slopes of 0.0005-0.0007 m m^{-1} and sinuosities of 1.25 are common (Koch et al., 1977). Riparian vegetation communities consist of willow (*Salix* spp.), cottonwood (*Populus* spp.), blue grama (*Bouteloua gracilis*) and western wheatgrass (*Agropyron smithii*) (White and Bramblett, 1993). An overview of representative physiographic regions of the study reach is in **Figure 4-3**.

Climate of the lower river is semi-arid continental (Lesica and Miles, 2001; Peel et al., 2007; Smith et al., 2000). Three long-term climate stations provide daily information within the project site. These are Forsyth (243098), Miles City Municipal Airport (APT) (245690), and Glendive (343581) (**Table 4-1**). Normals for the 1971-2000 period at each location are shown in **Figure 4-4** (left). Air temperature ranges from -13.7 to 31.0˚C (7.4-87.9˚F) while cumulative precipitation is 340-360 mm (13.5-14.1 inches) (WRCC, 2009). Most of the precipitation comes as rainfall in the months of June and September. The frost-free summer period is 140-150 days (State Engineer's Office, 1948; Zelt et al., 1999) characteristic of hot and dry conditions with evaporation between 750-1000 mm (30-40 in).

Five active streamflow gaging stations are present to characterize hydrology within the project reach (**Table 4-2**). There are three are on the mainstem river: (1) USGS 06295000 Yellowstone River at Forsyth, MT, (2) USGS 06309000 Yellowstone River at Miles City, MT, and (3) USGS 06327500 Yellowstone River at Glendive, MT; while two are on the tributaries: USGS 06308500 Tongue River at Miles City, MT and USGS 06326500 Powder River near Locate, MT.

Flow at the mainstem locations is fairly similar throughout the study reach with the exception of runoff. During this time there is a notable increase in flow with drainage area. This suggests that much of the drainage basin is emphemeral and does not contribute significantly to low-flow. The streamflow regimen is characteristic of a snowmelt hydrograph with a small-magnitude early spring rise due to localized low-elevation runoff, and then a prolonged high-magnitude peak from the large stores of snow water equivalent in the upper basin (**Figure 4-4**).

The major tributaries (Tongue and Powder rivers) enter at roughly one-third and two-thirds of the overall project length. They contribute significantly during the snowmelt period (resulting in most of the variance between the gages on the mainstem river). However, they contribute only marginally to the overall water yield during the summer months.

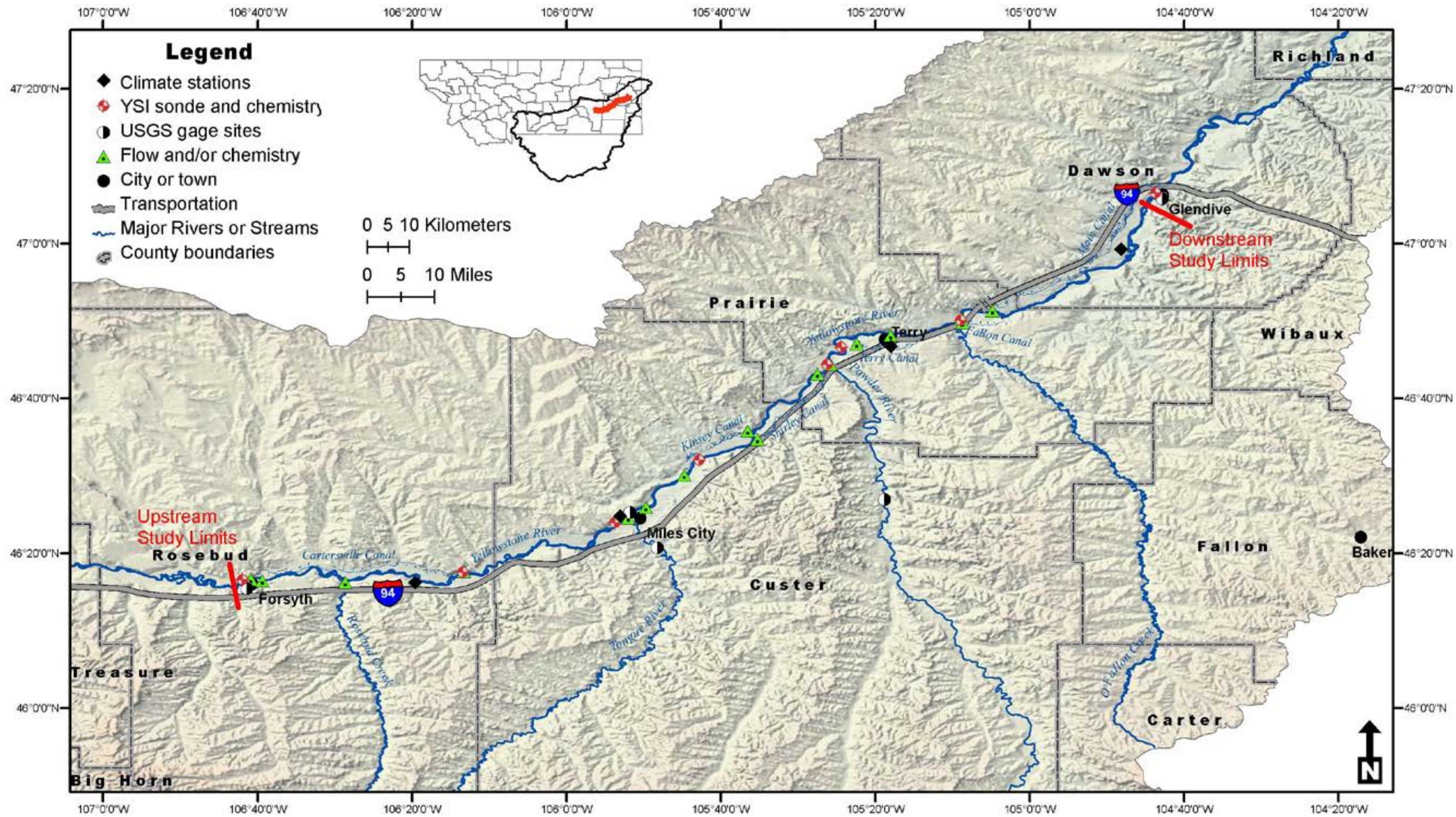
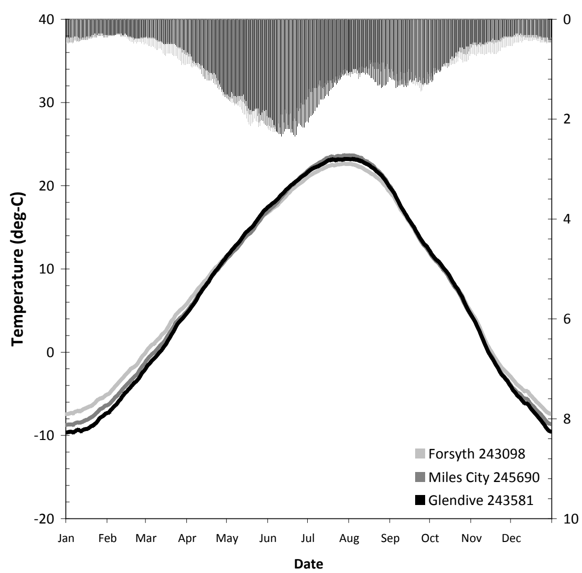


Figure 4-2. Lower Yellowstone River study area showing monitoring locations and other features.

Figure 4-3. Representative regions of the lower Yellowstone River project site.

Table 4-1. Long-term climatic stations on the lower Yellowstone River.

Station ID	Station	Latitude	Longitude	Station Elevation (m)	Period of Record
243098	Forsyth, MT	46.267	-106.667	767	1975-current
245690	Miles City APT, MT	46.433	-105.883	800	1936-current
243581	Glendive, MT	47.100	-104.717	633	1893-current



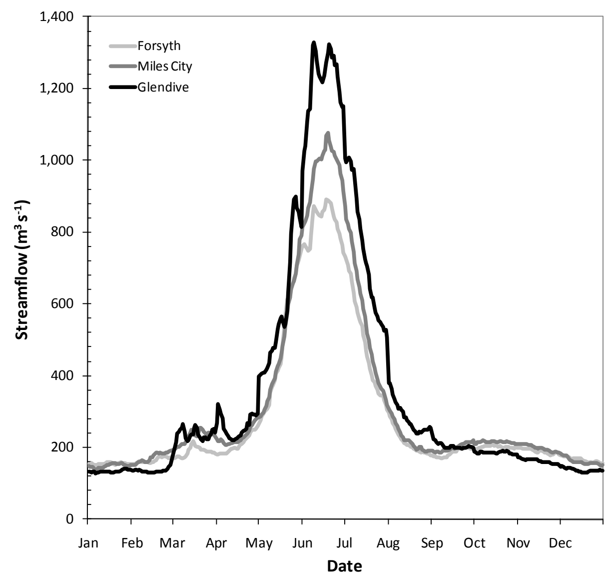


Figure 4-4. Mean daily normals for the lower Yellowstone River.
(Left panel) 1971-2000 precipitation and temperature for Forsyth (243098), Miles City Airport (245690), and Glendive, MT (243581), (Right panel) 2000-2008 mean daily streamflow at USGS Yellowstone River at Forsyth (06295000), Miles City (06309000), and Glendive, MT (06327500).

Table 4-2. Active streamflow gaging stations on the Yellowstone River.
Data from USGS NWIS (accessed 9/25/08).

Station ID	Description	Lat.	Long.	Drainage Area (km^2)	Mean annual streaflow	
					(m^3 s^{-1})	(ft^3 s^{-1})
06295000	Yellowstone River at Forsyth, MT	46.266	-106.690	103,933	287	10,150
06308500	Tongue River at Miles City, MT	46.385	-105.845	13,972	11	399
06309000	Yellowstone River at Miles City, MT	46.422	-105.861	124,921	316	11,160
06326500	Powder River near Locate, MT	46.430	-105.309	33,832	16	558
06327500	Yellowstone River at Glendive, MT	47.106	-104.717	172,779	356	12,560

4.3 Beneficial Uses and Water Quality Standards for the River

The beneficial use class designations for our study reach are found in the Administrative Rules of Montana (ARM 17.30.611). Accordingly, the lower Yellowstone River is a "B-3" type water (ARM, 17.30.625) and beneficial uses and criteria that DEQ is required to protect for such waterbodies are detailed in **Table 4-3** (established by ARM 17.30.625 and DEQ-7). The focus of modeling then will be to link already-established water quality standards (e.g., DO, pH, nuisance algae, etc. from **Table 4-3**) to nutrient concentrations that will be protective of beneficial uses.

Table 4-3. Water-use classification, beneficial uses, and standards for the lower Yellowstone River.

Segment Description	Use Class	Beneficial Uses
Yellowstone River mainstem from the Billings water supply intake to the North Dakota state line	B-3[8]	Drinking, recreation, non-salmonid fishery and associated aquatic life, waterfowl and furbearers, agricultural and industrial water supply
Standards for B-3 waters (e.g. lower Yellowstone River) are: 1. Dissolved oxygen levels ≥ 5 mg L^{-1} in order to protect aquatic life and fishery uses (early life stages; DEQ 2012). 2. Total dissolved gas levels, which must be ≤ 110% of saturation to protect aquatic life (Montana Department of Environmental Quality, 2012). 3. Induced variation of hydrogen ion concentration (pH), which must be less than 0.5 pH units within the range of 6.5 to 9.0, or without change if natural is outside this range [ARM 17.30.625(2)(c)] to protect aquatic life. 4. Turbidity levels, which a maximum increase of 10 nephelometric turbidity units (NTU) is acceptable; except as permitted in 75-5-318, MCA [ARM 17.30.625(2)(d)] to protect aquatic life. 5. Benthic algae levels, which DEQ interprets per our narrative standard (ARM 17.30.637(1)(e) should be maintained below a nuisance threshold of 150 mg Chl*a* m^{-2} to protect recreational use.		

4.4 Limits of Criteria Derived in this Study

Nutrient criteria derived in this study will be limited to specific longitudinal extents. Four candidate criteria assessment “units” (i.e., different longitudinal river reaches) were identified to accommodate changes in river behavior. These were based on waterbody segment IDs and are as follows: (Unit 1), the Middle Rockies region B-1 zone which extends from the Wyoming state-line to the Laurel public water supply (PWS) (MT42K001_010); (Unit 2), the B-2 and B-3 zone from the Laurel PWS to the Bighorn River (MT42K001_020); (Unit 3), the B-3 middle great plains region from the Bighorn River to the Powder River (MT42M001_011); and (Unit 4), the lower great plains region B-3 zone from the Powder River to the state-line (MT42M001_011) (**Table 4-4**). Only the latter two units are being evaluated as part of this study. The first two units (1 and 2) will be evaluated in the future. Field data collection for Units 1 and 2 was originally scheduled for summer 2011, but was completed in summer/fall 2012 due to other department commitments and unusually high flows in 2011.

Table 4-4. Waterbody segments proposed for nutrient criteria development.
Only Units 3 and 4 are being addressed as part of this report.

Criteria Unit	Waterbody Segment ID(s)	Segment Description(s)	Use Class
1	MT43B001_011 MT43B001_010 MT43B003_010 MT43F001_012	Montana State border to Yellowstone Park Boundary Yellowstone Park Boundary to Reese Creek Reese Creek to Bridger Creek Bridger Creek to City of Laurel PWS	B-1 B-1 B-1 B-1
2	MT43F001_011 MT43F001_010 MT43Q001_011	City of Laurel PWS to City of Billings PWS City of Billings PWS to Huntley Diversion Dam Huntley Diversion Dam to Big Horn River	B-2 B-3 B-3
3	MT42K001_020 MT42K001_010	Big Horn River to Cartersville Diversion Dam Cartersville Diversion Dam to Powder River	B-3 B-3
4	MT42M001_012 MT42M001_011	Powder River to Lower Yellowstone Diversion Dam Lower Yellowstone Diversion Dam to North Dakota border	B-3 B-3

[8] Water use classes B-1 and B-2 not evaluated as part of this study (they are located upstream).

4.5 Historical Water Quality Summary

A historical summary of water quality on the Yellowstone River is of importance because of the vast changes that have taken place over the past years. A cursory review is presented below so that readers may understand the current context with reference to historical conditions.

Interest in Yellowstone River water quality first peaked in the early 1950s as a result of Federal Water Pollution Control Act of 1948 (Pub. L. No. 80-845, 62 Stat. 1155). Taste and odor problems had become a problem because the river was effectively receiving untreated municipal and industrial wastewater (Montana Board of Health, 1952). Complaints had been filed at a number of locations regarding things such as oily wastes and oil-tasting fish, odors from sugar-beet discharges, contributions of blood and animal tissue from meat-packing plants, raw sewage, and other unpleasantries (Montana Board of Health, 1952; Montana Board of Health, 1956). Aggressive waste control policies were therefore recommended by the Montana Board of Health (1956) to mitigate these impacts.

Soon a number of municipal and industrial sewage treatment plants were in planning or already under construction (Montana Board of Health, 1963). By 1977 the river was declared as "nearly" meeting state water quality standards (Karp et al., 1977). Recent water quality assessments tend to support this assertion. DEQ currently identifies the river as either being "fully" or "partially" supporting uses on the lower river based on the most recent assessment record (DEQ, 2009).

From these past efforts, pollution in the watershed has been well characterized (Karp et al., 1977; Montana Board of Health, 1956; Montana Board of Health, 1963; Montana Board of Health, 1967). Major wastewater and industrial facilities are located in Livingston, Billings, Forsyth, Miles City, Glendive, and Sidney. A number of other MPDES permits are also present including industrial discharges, confined animal feeding operations (CAFOs), and stormwater permits. Nonpoint sources include agriculture, urban expansion, septic systems, land clearing, mining, and silviculture.

The single largest tributary nonpoint source is the Powder River, which is responsible for at least 30% of the annual suspended sediment load to the river (Zelt et al., 1999). It has been described as a mile wide, too thin to plow, and too thick to drink (Montana Board of Health, 1952). Much of its contribution is thought to be natural and based on the historical description below. For example, Vance et al. (2006) indicate that Francois Antoine Laroque passed through the lower Yellowstone in the early 1800s (prior to Lewis and Clark). He describes, "*The Powder River is here about ¾ acre in breadth, its water middling deep, but it appears to have risen lately as a quantity of leaves and wood was drifting on it…It is amazing how very barren the ground is between this and the less Missouri, nothing can hardly be seen but those Corne de Racquettes (prickly pear cactus). Our horses are nearly starved. There is grass in the woods but none in the plains…The current of the river is very strong and the water so muddy that it is hardly drinkable. The savages say that it is always thus and that is the reason that they call it Powder River; from the quantity of drifting fine sand set in motion by the coast wind which blinds people and dirtys the water*."

Similarly, on Friday July 30th, 1806, William Clark of the Lewis and Clark expedition noted, "*Here is the first appearance of Birnt hills which I have Seen on this river they are at a distance from the river on the Lard Side…after the rain and wind passed over I proceeded on at 7 Miles passed the enterance of a river the water of which is 100 yds wide, the bead of this river nearly ¼ of a mile this river is Shallow and the water very muddy and of the Colour of the banks a darkish brown. I observe great quantities of red Stone* *thrown out of this river that from the appearance of the hills at a distance on its lower Side induced me to call this red Stone river. [NB: By a coincidence I found the Indian name Wa ha Sah] as the water was disagreeably muddy I could not Camp on that Side below its mouth*."

Thus turbidity has always been associated with the Powder River confluence even when there is no anthropogenic source or flow contributions to account for such changes [also observed by us and Peterson and Porter (2002)]. As a consequence, we feel it is reasonable to conclude that there has always has been a very large natural sediment loading originating from this region and thus any turbidity that exists during low-flow conditions in the Yellowstone River is likely a natural source from the Powder River.

Three distinct water quality segments have also been delineated in the past to characterize water quality. These include: (1) the upper reach which drains the mountainous perennial streams and rivers upstream of Laurel, (2) a middle portion consisting of perennial headwaters and intermittent prairie regions extending from Laurel to Terry, and (3) a segment downstream of Terry to Sidney with primarily intermittent streams (Klarich and Thomas, 1977). These generally correspond with the locations identified in **Table 4-4** and reflect a steady and gradual decline in water quality that occurs due to both natural and anthropogenic causes (Klarich, 1976; Thomas and Anderson, 1976; Zelt et al., 1999). Relative contributions from these sources are not yet well-quantified.

Groundwater of the region is a final consideration and generally is of poor quality. Smith, et al., (2000) indicate that the shallow hydrologic unit (nearest the river) is moderately polluted. Wells within 70 feet of the ground surface have shown the greatest impact. It is believed that the interaction is related to agricultural management, native near-surface geologic materials, and aquifer recharge from irrigation infrastructure. The groundwater is highly mineralized naturally and the average dissolved constituent concentration is greater than 1,400 milligrams per liter (mg L^{-1}) (Smith et al., 2000). Nutrient increases are believed to be primarily man-caused.

5.0 MODELING STRATEGY

The modeling strategy for the project was to develop nutrient criteria limits (on a concentration basis) by using well-established water quality models to understand the linkage between nutrients and associated water quality responses. We could then use the model to simulate critical nutrient conditions and establish numeric nutrient criteria thresholds. The modeling rigor was matched with the necessary level of confidence required of the outcome. Given the socio-political and economic burden that can ensue from unneeded nutrient controls (e.g., waste water treatment plant upgrade costs, pollutant trading requirements, etc.), a high level of detail was necessitated. This requirement was then balanced with a number of other practical considerations including available funding and resources, data collection requirements, project scope, and management effort. A steady-state (as opposed to dynamic) modeling approach was selected due to its relative simplicity and more modest data requirements.

5.1 RATIONALE FOR THE MODEL DEQ SELECTED

The model selected by DEQ for the Yellowstone work was the enhanced river water quality model QUAL2K (Chapra et al., 2008). It was chosen for the following reasons: (1) its ability to simulate the eutrophication response state-variables of interest, (2) nationwide use in dissolved oxygen (DO) modeling, TMDL planning, and wasteload studies (Crabtree et al., 1986; Drolc and Koncan, 1996; Rauch et al., 1998), (3) modest data requirements, (4) relative simplicity in model application and development, (5) very good modeling documentation and user support, and (6) endorsement by EPA (Wool, 2009). Further details regarding its selection are described in the project QAPP (**Appendix A**).

5.2 QUAL2K DESCRIPTION

QUAL2K (Q2K) (Chapra et al., 2008) is a steady-flow, one-dimensional water quality model that solves advection and dispersion mass transport and constituent reactions along the direction of flow. It is a revision of the original QUAL2E (Brown and Barnwell, 1987) and includes the following improvements: variable sized elements, multiple loadings and withdrawals, carbonaceous biochemical oxygen demand (CBOD) speciation, sediment-water interactions, and the addition of bottom algae. Numerical computations in Q2K are programmed in Fortran 90, and are implemented from the Microsoft Excel and Visual Basic for Applications (VBA) environment. The addition of bottom algae and light extinction are significant improvements over QUAL2E given that benthic algae have an important biological role in regional rivers and a profound effect on river recreation. In addition, Q2K has an improved light-transmission model which is of great benefit given the impact of the Powder River on water clarity.

Over 20 water quality state-variables are simulated in Q2K, many which are eutrophication related. Included are: temperature, alkalinity, pH, conductivity, dissolved oxygen (DO), carbonaceous biochemical oxygen demand (CBOD), organic-nitrogen (N), ammonia-N, nitrate-N, organic-phosphorus (P), inorganic-P, suspended algae, attached algae, internal nitrogen and phosphorus of algae, and inorganic and volatile suspended solids. Total Organic Carbon (TOC) can also be readily calculated. A finite segment (or control volume) balance in terms of flow, heat, and concentration is written within each element for each constituent which in turn provides conditions for the adjacent elements in the model grid. Numerical backward difference schemes including both first (Euler) and fourth-order (Runge-Kutta) are available and Q2K can be used in a quasi-dynamic mode where water temperature, kinetics, and algal growth rates are allowed to vary diurnally so that the user can study the daily fluctuation over a 24-hr cycle (Chapra et al., 2008).

Q2K is not without limitations though and should only be applied when streamflow and input wasteloads are approximately steady-state, and where lateral and vertical gradients in water quality are negligible. Finally, it should not be applied and in cases where transient water quality conditions occur. Additional information about the modeling software and documentation can be found at: http://www.epa.gov/athens/wwqtsc/html/qual2k.html.

5.2.1 Conceptual Representation

Q2K represents a river as a series of interconnected reaches and elements that are in steady-state with one another. A prototype river is shown in **Figure 5-1** and consists of (1) a headwater boundary condition, (2) downstream reaches which are interconnected, and (3) a downstream boundary condition (all boundaries are Dirichlet or type 1, where the value of the unknown function is specified). Reaches can further be subdivided into elements of unequal length which are the fundamental computational unit of the model. Mass can be gained or lost from anywhere in the model network including tributaries and point and nonpoint source contributions, or withdrawals.

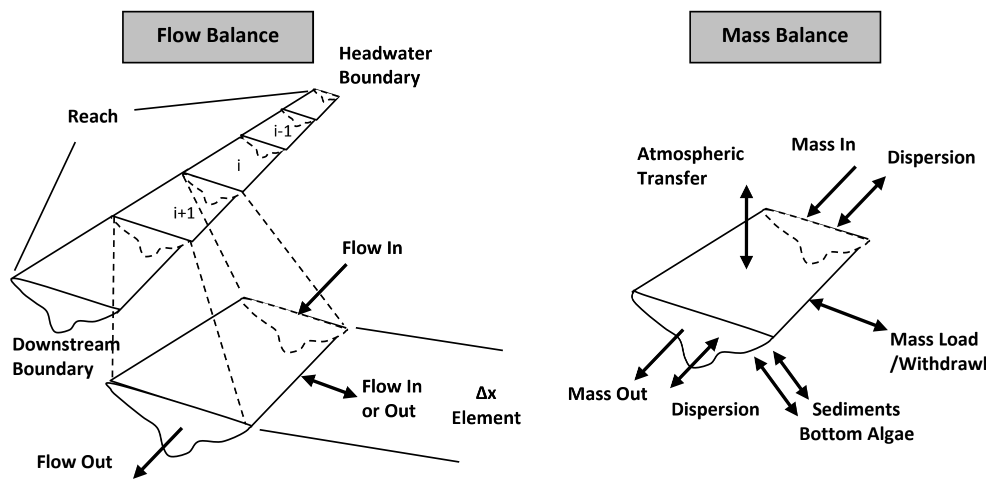


Figure 5-1. Conceptual representation of Q2K (redrawn from Brown and Barnwell, 2004).
(Left panel) Flow balance. (Right panel) Mass balance.

5.2.2 Temperature Model

The temperature algorithms of Q2K are deterministic and govern all reaction kinetics. Five heat exchange processes are simulated: net solar shortwave radiation into the water, longwave radiation from the atmosphere, longwave radiation from the water back to the atmosphere, conduction between the air/water and the water/bed boundary layers, and evaporative heat transfer. The overall mass balance is framed in the terms of heat (**Equation 5-1**), where T_i = temperature in element i [^{o}C], t = time [d], Q_i = outflow from element i to next downstream element [m^3 d^{-1}], $Q_{out,i}$ = total additional outflows from element i [m^3 d^{-1}], V_i = volume of element i [m^3], E'_i = the bulk dispersion coefficient between elements i and i + 1 [m^3 d^{-1}], $W_{h,i}$ = the net heat load from point and nonpoint sources into element i [cal d^{-1}], H_i = depth of element i [m], ρ_w = the density of water [g cm^{-3}], C_{pw} = the specific heat of water [cal (g ^{o}C)$^{-1}$], $J_{a,i}$ = the air-water heat flux [cal (cm^2 d)$^{-1}$], and $J_{s,i}$ = the sediment-water heat flux [cal (cm^2 d)$^{-1}$] (Chapra et al., 2008).

$$\frac{dT_i}{dt} = \frac{Q_{i-1}}{V_i} T_{i-1} - \frac{Q_i}{V_i} T_i - \frac{Q_{out,i}}{V_i} T_i + \frac{E'_{i-1}}{V_i}(T_{i-1} - T) + \frac{E'_i}{V_i}(T_{i+1} - T_i)$$

(Equation 5-1)

$$+ \frac{W_{h,i}}{\rho_w C_{pw} V_i}\left(\frac{\text{m}^3}{10^6 \text{ cm}^3}\right) + \frac{J_{a,i}}{\rho_w C_{pw} H_i}\left(\frac{\text{m}}{100 \text{ cm}}\right) + \frac{J_{s,i}}{\rho_w C_{pw} H_i}\left(\frac{\text{m}}{100 \text{ cm}}\right)$$

Incoming shortwave radiation is modeled via latitude, longitude, and time of the year. It is attenuated by atmospheric transmission, cloud cover, reflection, and topographic or vegetative shading. Longwave radiation is calculated according to the Stefan-Boltzmann law, and conduction and evaporation are calculated using wind-dependent relationships. As outlined in **(Equation 5-1)**, advection and dispersion are then used to calculate heat transfer from upstream to downstream elements.

5.2.3 Constituent Model

The constituent mass-balance within Q2K includes all key eutrophication components of interest including N and P cycling (e.g., hydrolysis, settling, uptake, nitrification, denitrification), algal growth processes (photosynthesis, respiration, death, and excretion), and oxygen kinetics and mass transfer (carbonaceous biochemical oxygen demand, reaeration, sediment oxygen demand). Model state-variables are shown in **Table 5-1** and a conceptual diagram of model kinetics is shown in **Figure 5-2**.

Table 5-1. Model state-variables in Q2K.

State-Variable	Symbol	Units
Conductivity	s	μmhos
Inorganic suspended solids	m_i	mg D L^{-1}
Dissolved oxygen	O_o	mg O_2 L^{-1}
Slowly reacting CBOD	c_s	mg O_2 L^{-1}
Fast reacting CBOD	c_f	mg O_2 L^{-1}
Organic nitrogen	n_o	μg N L^{-1}
Ammonia nitrogen	n_a	μg N L^{-1}
Nitrate nitrogen	n_n	μg N L^{-1}
Organic phosphorus	p_o	μg P L^{-1}
Inorganic phosphorus	p_i	μg P L^{-1}
Phytoplankton	a_p	μg Chl*a* L^{-1}
Phytoplankton nitrogen	IN_p	μg N L^{-1}
Phytoplankton phosphorus	IP_p	μg P L^{-1}
Detritus	m_o	mg D L^{-1}
Alkalinity	Alk	mg $CaCO_3$ L^{-1}
Total inorganic carbon	c_T	mole L^{-1}
Bottom algae biomass	a_b	mg Chl*a* m^{-2}
Bottom algae nitrogen	IN_b	mg N m^{-2}
Bottom algae phosphorus	IP_b	mg P m^{-2}

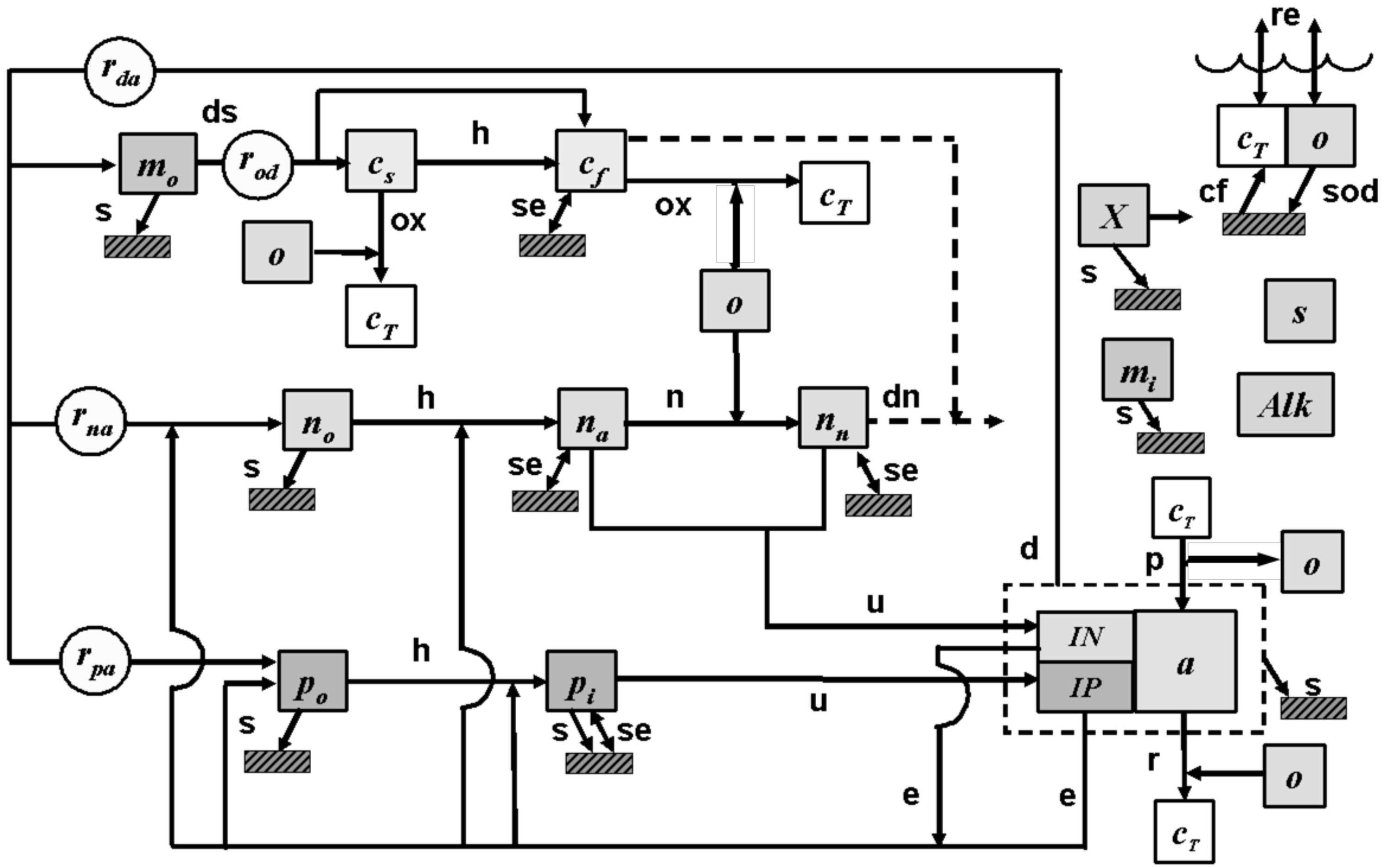


Figure 5-2. Diagram of model kinetics and mass transport processes in Q2K.
Redrawn from Chapra et al., 2008 (with permission). Kinetic processes are as follows: ds = dissolution, h = hydrolysis, ox = oxidation, n = nitrification, dn = denitrification, p = photosynthesis, r = respiration, e = excretion, d = death, r = respiration. Mass transfer processes are: re = reaeration, s = settling, SOD = sediment oxygen demand, se = sediment exchange, cf = inorganic carbon flux, u = uptake.

A general mass balance for constituents within each element are written as in **Equation 5-2**.

(Equation 5-2) $$\frac{dc_i}{dt} = \frac{Q_{i-1}}{V_i}c_{i-1} - \frac{Q_i}{V_i}c_i - \frac{Q_{out,i}}{V_i}c_i + \frac{E_{i-1}^{'}}{V_i}(c_{i-1} - c_i) + \frac{E_i^{'}}{V_i}(c_{i+1} - c_i) + \frac{W_i}{V_i} + S_i$$

where c_i = the constituent concentration in element *i*, W_i = the external loading of the constituent to element *i* [g d^{-1} or mg d^{-1}], and S_i = sources and sinks of the constituent due to reactions and mass transfer mechanisms [g $(m^3 d)^{-1}$ or mg $(m^3 d)^{-1}$]. For bottom algae variables, the transport and loading terms are omitted.

5.3 General Data Requirements for QUAL2K

The data requirements for Q2K are lengthy but generally include headwater and climatic forcings (e.g., streamflow, mass/quality constituents, climatic information, etc.), ancillary boundary condition information (e.g., point source inflows, diffuse flows, etc.), advection and dispersion mass transport formulations, rate and kinetic coefficients, and benthic processes. All of these are necessary to provide a good representation of the physical system and biogeochemical transformations. Ways to obtain such information include (in decreasing order of accuracy): (1) direct field measurements, (2) indirect observations from field data, (3) model calibration, or (4) the literature (Barnwell et al., 2004). Data collection for the project was structured to meet these data requirements as described in **Section 6.0**.

5.4 Assumptions and Limitations

A number of assumptions and limitations are implicit with the use of Q2K. Those of importance to our effort include:

- Complete mixing, both vertically and laterally.
- Approximate steady-state conditions[9].

In this instance, it is assumed that the major pollutant transport mechanisms (advection and dispersion) are significant only along the longitudinal direction of flow, which was confirmed as detailed in **Section 5.5**[10]. For the latter, our selection of the critical low-flow period largely result in steady-state conditions given that the river is both hydrologically and thermally stable (see **Section 5.5)**.

5.5 Verification of Model Assumptions

Assumptions framed in **Section 5.4** were verified in the field in 2006. Complete vertical and lateral mixing was confirmed at a number of cross-sections using a YSI 85 hand-held meter by taking

[9] Q2K simulates a single day's streamflow, water quality, and meteorological conditions (or an average of multiple days of conditions) repeatedly for a user-specified number of days. Thus, a dynamic steady-state is computed for that day, or period of days. Diurnal changes are brought about by shifts in hourly temperature, meteorological data, and solar radiation and photoperiod.
[10] The exception being areas directly downstream of WWTPs or tributary inflows where it is obvious that significant lateral water quality gradients exist. The modeling network was carefully constructed so that incomplete mixing at those sites did not affect modeling outcomes.

measurements both laterally and vertically in the water column (Suplee et al., 2006a). Site water quality was homogeneous at all sites in the river[11].

Steady-state streamflow, boundary condition, and biological assumptions were affirmed through a review of historical thermal and hydrologic data on the river. It was assumed that these measures would be a good surrogate of nutrient loading and biological activity[12]. Our analysis indicates that relatively stable conditions[13] occur around the second or third-week of August and persist through the end of September (**Table 5-2, Figure 5-3**). Hence this is a good time for field data collection and associated model development work. Please refer to the project QAPP for more information regarding these findings (**Appendix A**) (Suplee et al., 2006a).

Table 5-2. Verification of steady-state flow requirements for QUAL2K.
Based on analysis of USGS gage 06295000 Yellowstone River at Forsyth, MT (1977-2008)[1].

Week	Week Beginning Streamflow	Week End Streamflow	Change in Flow (%)
August 1-7	10,500	8,500	-19.0
August 7-13	8,500	7,280	-14.4
August 13-19	7,280	6,880	-5.5 (begin steady-flow)[2]
August 19-25	6,880	6,570	-4.5
August 25-31	6,570	6,210	-5.5
September 1-7	6,240	5,970	-4.3
September 7-13	5,970	6,370	+6.7
September 13-19	6,370	6,850	+7.5
September 19	6,850	7,160	+4.5
September 25	7,160	6,930	-3.2

[1]Comparisons made with published USGS data records.
[2]Variation <± 10% considered acceptable for steady-flow model applications.

[11] DO and temperature were used as the indicator. Surface-to-bottom dissolved oxygen gradients were negligible except in one instance, within a filamentous *Cladophora* bed. These gradients did not extend up into the water column above the algae beds however.
[12] Streamflow stability would tend to suggest that nutrient loads would be fairly constant over time. This is true for tributaries and natural settings, but could be altered by anthropogenic effects (e.g., WWTP, septic, etc.). Water temperature is a good indicator of biological activity. It governs all of the rate constants in the model (according to the Arrhenius equation).
[13] Here we define stability as a change no greater than 10% over a one week period.

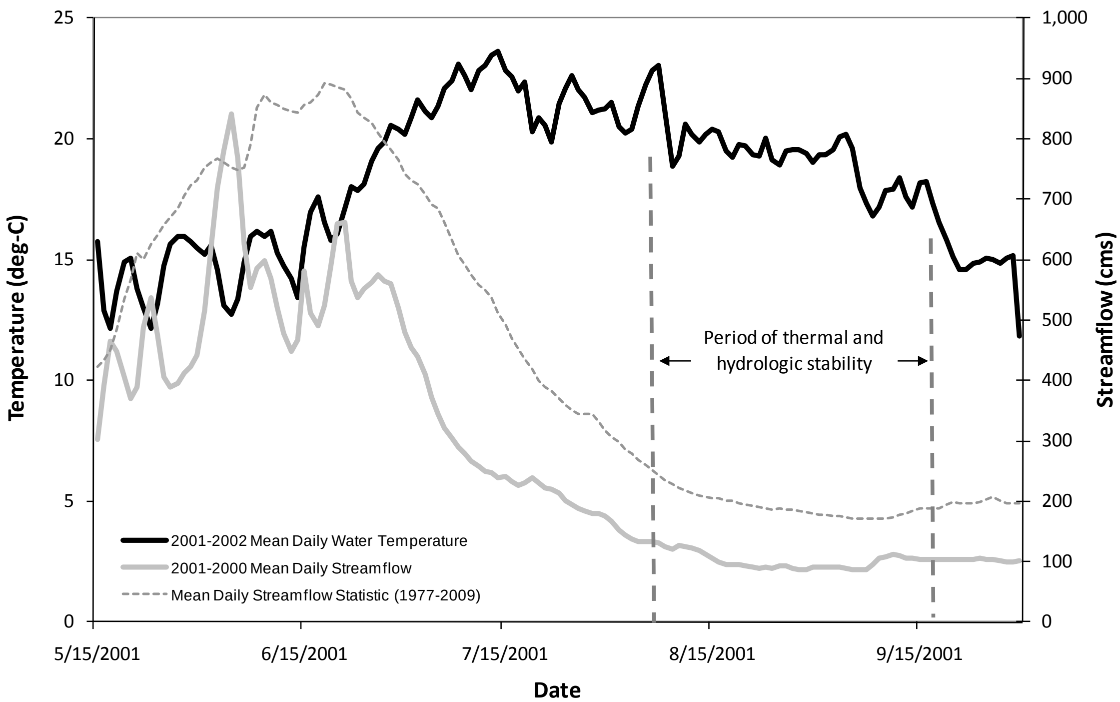


Figure 5-3. Typical occurrence of thermal and hydrologic stability in the Yellowstone River.
The onset of thermal and hydrologic stability begins approximately August 1 and continues into late September. Temperature data from 2001-2002 were obtained by taking the flow-weighted average of USGS 06214500 Yellowstone River at Billings, MT and USGS 06294500 Bighorn River above Tullock Creek, near Bighorn, MT.

5.6 AlgaeTransect2K (AT2K; a Q2K Cross-section Model)

DEQ worked cooperatively with Tufts University to develop a new model, AlgaeTransect2K (AT2K), which relates longitudinal Q2K model output to lateral benthic algae densities. A tool such as this was needed because bottom algae typically exhibit lateral heterogeneity in rivers with higher densities in the shallow near-shore areas and lower biomasses in deeper areas. The importance of these shallow (or wadeable) areas is reinforced by the fact that human use and perception is often inclined towards these locations (i.e., they are the locations where recreational use is highest) and excessive levels of benthic algae greatly diminish people's recreational experience (Suplee et al., 2009). River margins are also important nursery areas for fish larvae and young-of-year juveniles (Scheidegger and Bain, 1995). Consequently, AT2K was developed to fill the mean cross-sectional river biomass deficiency in Q2K, that is, to simulate the actual distribution of benthic algae within a given Q2K model element.

AT2K's conceptual representation is shown in **Figure 5-4**. A single river element is represented by lateral transect variation in depth z [m] with distance y [m] over an element of wetted width B [m], where algal biomass (mg Chl*a* m^{-2}) is computed as a function of attenuated light to the channel bottom, soluble nutrient concentrations (N and P), and algal growth kinetics. Rather than running the calculation for every station, AT2K first develops a table of biomasses and associated depth increments, and then linearly interpolates mean biomass levels for each depth in the transect. A finer, more uniformly-spaced depth profile can also be generated between soundings if desired. Assumptions of AT2K encompass all of those identified for Q2K, including that constituent water quality is sufficiently well-mixed vertically and laterally[14] and that the effects of velocity, channel substrate, and riparian shade are insignificant.

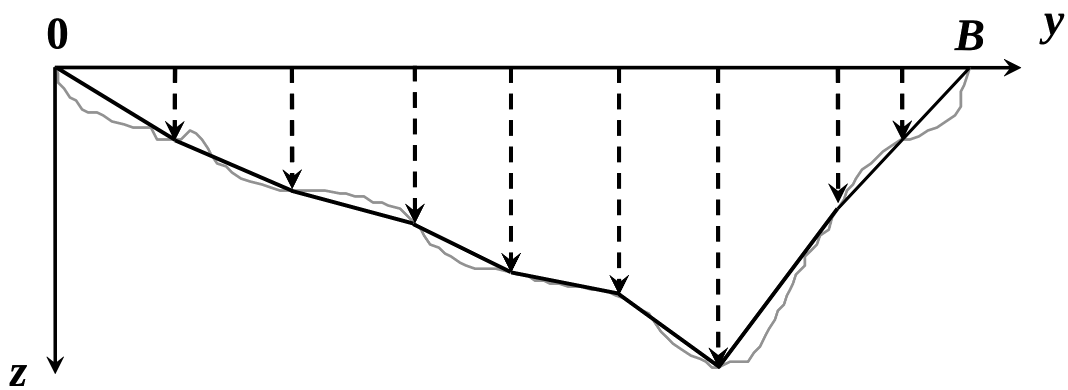


Figure 5-4. Conceptual representation of the AlgaeTransect2K (AT2K) model.
The model represents a river transect as a single Q2K element with variable depth to evaluate the effect of lateral light attenuation on algal growth. The primary consideration in the development of AT2K was to make it consistent with the existing version of Q2K. As a result Q2K optics and algal growth submodels are used for all calculations[15].

It is not entirely clear how well AT2K works when applied as a post-processor to QUAL2K although from initial testing we found that (1) simulated areal biomasses when laterally averaged were nearly identical to the lateral average in QUAL2K (meaning both models converge on the same areal biomass) and (2) calibration of both models could be done with only a single set of rate coefficients so that the kinetics in each model are identical (despite their difference in conceptual representation). That said, there is a possibility that transect station-specific computations from AT2K could in fact differ theoretically from laterally averaged computations in Q2K especially with regard to spatial differences in river productivity. These differences would be most likely to affect the oxygen and pH mass balances (although in later testing we found that these spatial errors seem to cancel) otherwise depth- and width- averaged results from the longitudinal model would not be correct.

5.7 Why the Transect Model (AT2K) was Needed

Two primary considerations necessitated AT2K development for large river settings. First, lateral benthic algal dynamics in large rivers are poorly understood and require a better understanding of the relationship between nutrients and algal density. For example, we may over- or under-state eutrophication potential if we do not consider the integrated response to alterations in light and depth. Second, current information seems to point to the fact that adverse water column responses (i.e.,

[14] The assumption of a homogeneous water column is often true, however, it could be violated immediately downstream from a major point sources such as a WWTP or tributary inflow. Such considerations should be taken into account during model development. Currently the effects of velocity or channel substrate are not included explicitly included in the model simulation.
[15] The following mechanistic processes are represented in the model: optics (light extinction over depth, i.e., Beer-Lambert law), photosynthetic light use efficiency (Baly, 1935; Smith, 1936; Steele, 1962), nutrient uptake (Rhee, 1973), and nutrient limitation (Droop, 1973). State-variables simulated include: (1) bottom-algae biomass, a_b, mg Chl*a* m^{-2}, (2) bottom-algae internal phosphorus, IP_b, mgP m^{-2}, and (3) bottom-algae internal nitrogen, IN_b, mgN m^{-2}. Please refer to Chapra et al., (Chapra et al., 2008) for further details.

standards violations for things such as DO) may be unlikely except in cases of gross or negligent pollution. The Yellowstone River is a good example. Even during times of heavy historic pollution (Montana Board of Health, 1956; Montana Board of Health, 1967) the river rarely exhibited water column impairment (e.g., DO minima, pH, etc.). Four general attributes of high-gradient large rivers like the Yellowstone in Montana seem to support a higher assimilative capacity:

- Turbidity and depth. Both are naturally greater in large systems in so naturally pushing them towards light limitation (Hynes, 1969).
- The volume of water per unit area is also high, which makes the biomass per unit volume low thereby limiting the eutrophication response (Hynes, 1969).
- Atmospheric oxygen/carbon dioxide reaeration coefficients are high which lend themselves to naturally fast purification processes.
- Channel bottoms are gravel/cobble with only minor amounts of fine sediment and organic matter and as a result have low sediment oxygen demands (SOD) (e.g., <0.5 g O_2 m^{-2} day^{-1}). (Note: that this last attribute will not hold true in many large rivers.)

Consequently, there is a natural propensity for large rivers to be less sensitive to nutrient pollution than smaller streams. However, proper assessment of support/non-support of beneficial uses in large rivers still requires evaluation of nutrient levels and associated water quality responses (both in the water column and specifically on algae). A model such as AT2K is needed to conduct such evaluations.

6.0 Project Design, Data Collection, and Supporting Studies

The project design for the Yellowstone River was reflected in the overarching question posed at the beginning of the study (Suplee et al., 2006a): "*In a segment of the lower Yellowstone River, what are the highest allowable concentrations of nitrogen and phosphorus that will not cause benthic algae to reach nuisance levels, or dissolved oxygen concentrations to fall below applicable state water quality standards*"? Specifically, the inquiry called for the use of a water quality model to link stressors with responses and to establish relationships between nutrient concentrations and eutrophication concerns (e.g., DO, pH, benthic algae, etc.). At the core of any model is its data. The data for the Q2K modeling effort is expounded upon in this section.

6.1 Summary of Field Data Collection to Support Modeling

A comprehensive field measurement program was initiated in 2007 to support modeling which meets/exceeds most steady-state modeling applications. This is described in the attached Quality Assurance Project Plan (QAPP) and Sampling and Analysis Plan (SAP) (**Appendix A)**. A cursory review is presented here so that the reader does not have to refer back to the Appendix.

Two synoptic river surveys were initiated during the summer of 2007 (August and September) to support model development. Collections were made to provide research quality data for the model. The following was characterized: water column chemistry and site biology; real-time water quality field parameters (using YSI datasondes); meteorological data; mainstem and tributary streamflow records; sediment oxygen demand (SOD); river productivity and respiration rates, and time of travel. The data collection took place during two separate 10-day periods in both August and September respectively (e.g., water samples, algal collections, rate measurements, etc.). All activities were carried out under the direction of the DEQ Quality Assurance (QA) program.

YSI 6600 extended deployment datasondes were deployed and maintained throughout the summer (approximately 2 months) to support the effort. Eight mainstem river sites and over a dozen tributaries/irrigation return flows were monitored. The following locations were of interest: (1) the Rosebud West FAS (at Forsyth, MT) to the Cartersville Canal return flow, (2) Cartersville Canal return flow to the 1902 Bridge (near Miles City, MT); (3) 1902 Bridge to the Kinsey Bridge FAS, (4) Kinsey Bridge FAS to the Powder River (near Terry, MT); (5) Powder River to Calypso Bridge, (6) Calypso Bridge to O'Fallon Creek, and (7) O'Fallon Creek to the Bell Street Bridge (at Glendive, MT). Sampling locations were shown previously in **Figure 4-2** and were chosen in accordance with Mills et al., (1986) and Barnwell et al., (2004) to describe the longitudinal profile of the river. We accommodated variability such as incoming tributaries, waste water treatment plant discharges, critical downstream points of concentration, and spatial differences in temperature brought about by climatic gradients and hydrogeomorphology. Full details are described in **Appendix A**.

6.2 Data Compilation and Supporting Information

A data compilation was undertaken to fill data gaps and provide supporting information for the model. An overview of this work is described in this section.

6.2.1 Sources

Sources of streamflow, climatic, and physical feature data used in the project are shown in **Table 6-1**. Streamflow records were acquired from the USGS via their National Water Information System (NWIS) (USGS, 2008) and were stored in Watershed Data Management (WDM) files for processing (Hummel et al., 2001). Water quality and chemistry data were retrieved from NWIS (USGS, 2008) and were combined with data from EPA's STOrage and RETrieval (STORET) database (EPA, 2008b). These were archived into a Microsoft Access™ project database. Records were also pulled from the Integrated Compliance Information System (ICIS) (EPA, 2010b), Ground Water Information Center (GWIC) (MBMG, 2008), and USGS National Water Quality Assessment (NAWQA) database. They were stored in their original format.

Climatic data for the project were obtained from the National Climatic Data Center (NCDC) (NOAA, 2009), Great Plains AgriMET Cooperative Agricultural Weather Network (BOR, 2009), and MesoWest (Mesowest, 2009). Supporting atmospheric information (CO_2 data, etc.) was also acquired from the Clean Air Status and Trends Network (CASTNET) (EPA, 2010a) and GlobalView-CO_2 (NOAA, 2010a) Planimetric data for Geographic Information System (GIS) analysis (including aerial photographs, river hydrography, bank lines, digital elevation model (DEM)/terrain data, and other features) were obtained from the Yellowstone River Corridor Resource Clearinghouse (NRIS, 2009). Data were saved in their original formats and were modified as the project necessitated.

Table 6-1. Data sources used in development of the Yellowstone River nutrient model.

Type of data	Sources
Streamflow	USGS NWIS, http://waterdata.usgs.gov/nwis EPA STORET, http://www.epa.gov/storet ICIS, http://www.epa.gov/compliance/data/systems/icis GWIC, http://mbmggwic.mtech.edu/ BRID (available by hardcopy request only)
Climatic, atmospheric	NWS, http://www.ncdc.noaa.gov/oa/ncdc.html BOR, http://www.usbr.gov/gp/agrimet MesoWest, http://mesowest.utah.edu/index.html EPACASTNET http://www.epa.gov/castnet/sites/thr422.html GLOBALVIEW-CO2 http://www.esrl.noaa.gov/gmd/ccgg/globalview/co2/co2_intro.html
Water quality	NWIS, STORET, ICIS (same as above)
Physical features	Yellowstone River Corridor Resource Clearinghouse http://nris.mt.gov/yellowstone

6.2.2 Personal Communications and Supporting Data

Data were also acquired through a number of direct personal communications. These included contact with (in alphabetical order): Army Corps of Engineers (ACOE, L. Hamilton, personal communication, Oct. 2, 2009); Bureau of Reclamation (BOR, D. Critelli, personal communication, Jul. 10, 2009 and T. Grove, personal communication, May 21, 2009); Buffalo Rapids Irrigation District (BRID, D. Schwarz, personal communication May 19, 2008); City of Forsyth, MT (P. Zent, personal communication Dec. 17, 2009); City of Miles City (A. Kelm, personal communication, May 23, 2008); Department of Natural Resources and Conservation (DNRC, L. Dolan, personal communication, Dec. 23, 2009 and T. Blandford, personal communication Jan. 6, 2010); Montana Bureau of Mines and Geology (MBMG, J. LaFave, personal communication, Mar. 24, 2010), Montana Department of Transportation (MDOT, B. Hamilton, personal communication Aug. 18, 2008); Montana State University (MSU; H. Sessoms, personal communication, Sept. 9, 2008); National Weather Service (NWS, J. Branda, personal communication Aug. 13, 2008); USGS (M. White, personal communication Mar. 30, 2009 and D. Peterson, personal communication Jan. 19, 2009); and the U.S. Range & Livestock experiment station (K. Molley, personal communication Jun. 3, 2008). DEQ maintains these communication records in our project logs.

6.2.3 Database

Attributes of the database developed for the project are shown below. Sites on the mainstem river with sufficient data for characterization of water quality are identified in **Table 6-2**. Tributary sites are shown in **Table 6-3**. Included is location[16], site ID, constituent of interest, gaging or sampling history and the number of independent observations. Only gaging records greater than 5 years and with more than 10 different sampling dates were included.

Table 6-2. Mainstem water quality stations on the Yellowstone River with sufficient data.

Site Location	USGS Site ID	DEQ Site ID(s)	Constituent	Number of Obs.	Period of Record
Laurel	06205200	2659YE03, 2659YE01, Y06YSR400, Y06YSR395, Y06YELSR01	Flow Climate/Air Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	none 134 132 56 181 59 95 2 yes 	n/a n/a 1974-2007 1974-2007 1975-2007 1974-2007 1974-2007 1974-2007 2007 2001,2004
Billings	06214500	Y06YSR470, Y06YSR520, Y12YSR550, Y12YSR549	Flow Climate/Air Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	daily hourly 172 202 160 180 138 189 13 yes	1928-2008 1935-2008 1967-2001 1969-2001 1971-2001 1969-2003 1970-2001 1965-2003 1975-2000 2001,2004
Forsyth	06295000	Y17YELSR09	Flow Climate/Air Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	daily hourly 176 181 99 197 103 184 19 yes	1977-2008 1998-2008 1974-2007 1974-2007 1974-2007 1974-2007 1973-2007 1975-2007 1978-2007 2001,2007

[16] Location was considered the same if within two kilometers spatially of one another, and no incoming tributaries, point sources, etc. were identified between each.

Table 6-2. Mainstem water quality stations on the Yellowstone River with sufficient data.

Site Location	USGS Site ID	DEQ Site ID(s)	Constituent	Number of Obs.	Period of Record
Miles City	06296120 06309000 (flow)	Y17YELSR01, 3682YE01, 3682YE02	Flow	daily	1946-2008
			Climate/Air	hourly	1936-2008
			Total N	184	1974-2007
			Ammonia ($NH_{3/4}$)	188	1974-2007
			NO_2+NO_3	134	1971-2007
			Total P	214	1974-2007
			SRP	136	1971-2007
			Solids	127	1965-2007
			Algae (either)	13	1975-2007
			Feature	yes	2001,2007
Terry	06326530	4086YE01, Y23YELLR02, Y23YELLR03	Flow	16	1974-1979
			Climate/Air	hourly	1998-2008
			Total N	109	1974-2007
			Ammonia ($NH_{3/4}$)	112	1974-2007
			NO_2+NO_3	19	1974-2007
			Total P	122	1974-2007
			SRP	20	1974-2007
			Solids	103	1975-2007
			Algae (either)	14	1975-2007
			Feature	yes	2001,2007
Glendive	06327500	4490YE01, Y23YELLR04	Flow	daily	2002-2008
			Climate/Air	hourly	1973-2008
			Total N	2	2007
			Ammonia ($NH_{3/4}$)	2	2007
			NO_2+NO_3	16	1976-2007
			Total P	14	1976-2007
			SRP	17	1973-2007
			Solids	9	1975-2007
			Algae (either)	4	2007
			Feature	yes	2001,2004
Sidney	06329500	NA	Flow	daily	1933-2008
			Climate/Air	hourly	1973-2008
			Total N	333	1970-2007
			Ammonia ($NH_{3/4}$)	468	1969-2007
			NO_2+NO_3	281	1971-2007
			Total P	426	1969-2007
			SRP	272	1971-2007
			Solids	427	1965-2008
			Algae (either)	10	1975-2005
			Feature	yes	2001,2004

Table 6-3. Major tributary water quality stations with sufficient data.

Location	USGS Site ID	DEQ Site ID (s)	Constituent	Number of Obs.	Period of Record
Rosebud Creek	06296003	Y14ROSBC01, Y14ROSBC04, Y14ROSBC05, Y17ROSEC01	Flow Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	daily 108 133 44 154 56 169 0 no	1974-2006 1975-2007 1975-2007 1975-2007 1975-2007 1974-2007 1974-2007 n/a n/a
Tongue River	06308500	Y16TONGR02, Y16TONGR03, Y16TR99, Y17TONGR01	Flow Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	daily 158 195 177 203 158 246 0 0	1938-2008 1974-2008 1974-2008 1971-2008 1971-2008 1971-2008 1974-2007 n/a n/a
Powder River	06326520 06326500 (flow)	3985PO01, 3985PO02, Y21PR40, Y21PWDRR01, Y21PWDRR02	Flow Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	daily 229 293 234 285 212 323 11 0	1938-2008 1974-2008 1977-2008 1975-2008 1974-2008 1973-2008 1965-2008 2000-2003 n/a
O'Fallon Creek	06326600	3989OF01, 4087OF01, Y22OFALC16, Y22OFALC08, Y22OFALC13	Flow Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	Daily 46 59 23 61 16 76 0 0	1977-1992 1977-2007 1977-2007 1975-2007 1977-2007 1973-2007 1975-2007 n/a n/a

6.3 Data Analysis

The sites identified previously (**Section 6.2**) were analyzed so that long term statistical information such central tendency (i.e., mean or median concentrations), variance, and distribution function could be ascertained. This allowed us to fill data gaps, draw conclusions from historical data, and better understand relational information about the river. Two examples are provided in this section. Similar comparisons are drawn in the rest of the document.

In **Figure 6-1**, a compilation of total nitrogen (TN) and total phosphorus (TP) samples for each of the mainstem river sites is shown (e.g. Laurel, Billings, Forsyth, Miles City, Terry, Glendive, and Sidney). While there is considerable variability in the data (as evidenced by the maximum and minimum whiskers), nutrient concentrations clearly tend to go up in the downstream direction. They also far exceed suggested TN and TP Level III Ecoregion nutrient criteria from the U.S. EPA (2001), even at the 25th percentile.

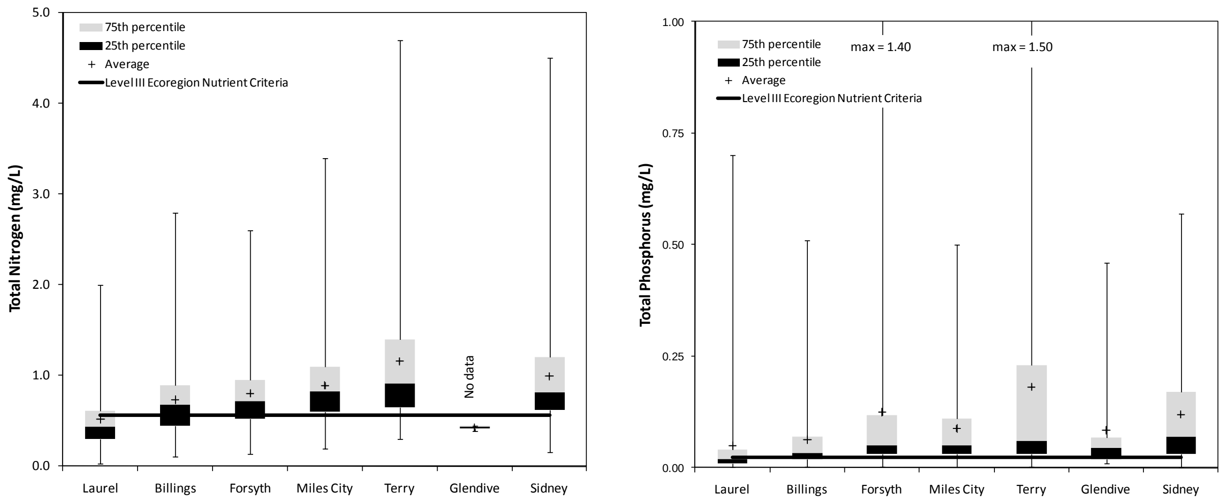


Figure 6-1. Historical nutrient concentrations in the lower Yellowstone River.
(Left panel) Historical TN data on the Yellowstone River. (Right panel). Same but for TP. Data are shown over the period of record for each site (1969-2007), and in both instances are well above the Level III ecoregional criteria (560 µg L^{-1} TN or 23 µg L^{-1} TP) proposed by EPA (2001).

In **Figure 6-2**, diurnal dissolved oxygen (DO) data were evaluated. Measurements from different locations and diel cycles during the month of August were compared (Klarich, 1976; Montana Board of Health, 1967; Peterson et al., 2001) and show good agreement between DO percent saturation in all years (**Figure 6-2**, left). This suggests that DO saturation in all studies, irrespective of the flow condition or even decade collected, is similar. It also demonstrates that our selection of a steady-state model is a reasonable choice as nearly all of the data falls within the ±10% fitted saturation curve.

Dissolved oxygen shows a fairly consistent longitudinal tendency in the river (**Figure 6-2**, right) as well. Daily diurnal DO flux (i.e., maximum daily DO minus minimum daily DO) is typically higher in the upper reaches of the river near Billings (i.e., more productive) and then diminishes in the downstream direction. Findings are consistent with Peterson and Porter (2002), as well as our observations that indicate longitudinal increases in turbidity influence (dampen) primary productivity. Data again have consistent spatial tendencies and again fall within the ±10% envelope identified previously.

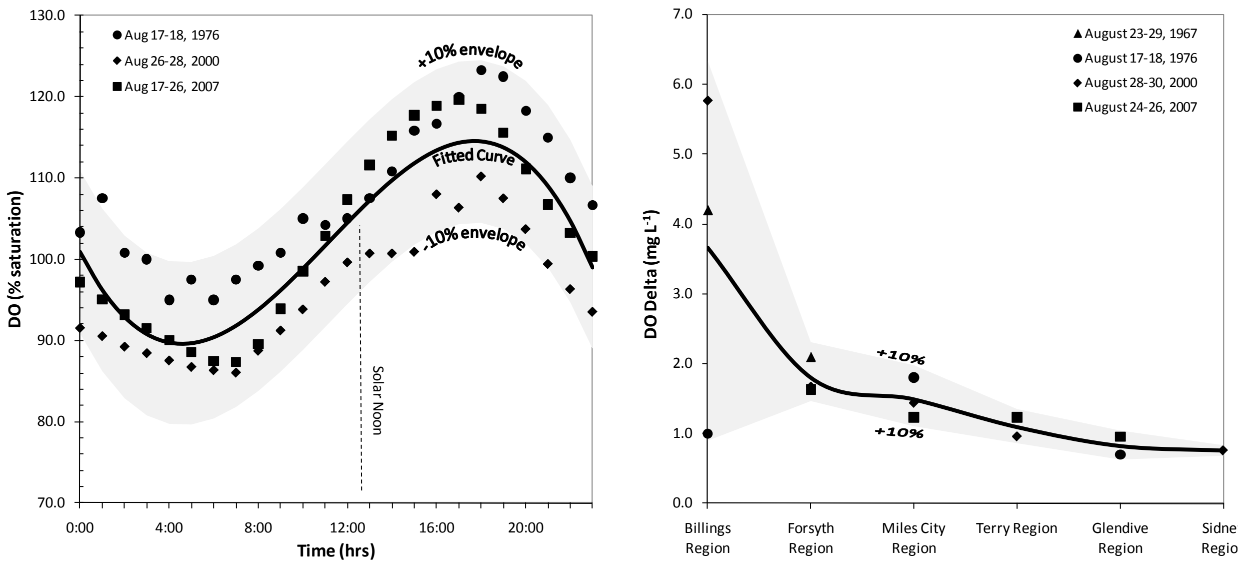


Figure 6-2. Dissolved oxygen data on the Yellowstone River for August 1967, 1976, 2000, and 2007. (Left panel) Typical diurnal pattern at Forsyth, MT over the August 1976, 2000, and 2007 period. A fitted curve is shown along with as ±10% saturation envelope. (Right panel) Longitudinal diurnal fluctuation in DO (i.e. max-min) over all years. Envelope shown as ±10% of the reported maximum or minima.

6.4 Outside Studies Useful for Modeling

Among the studies identified in **Section 6.2**, one was particularly useful because it had all of the necessary data for model development (e.g., water chemistry data, diurnal field parameters, and benthic and phytoplankton algae). This information was collected as part of the USGS National Water Quality Assessment (NAWQA) program (Peterson et al., 2001) and was quite comparable to the DEQ effort. Attributes of these two independent measurement programs are compared in **Table 6-4** and **Table 6-5.** They provide a good basis for which to make comparisons for August low-flow river conditions during two different years.

Table 6-4. Data collection matrix for the DEQ 2007 and USGS 2000 monitoring programs.
Comparisons between the USGS 2000 and DEQ 2007 effort.

Monitoring Location[1]	Climate	Streamflow	Water Chemistry[2]	Diurnal WQ[3]	Transport[4]	Kinetics[5]	Benthics[6]	Light[7]
Yellowstone River at Laurel			U				U	U
Yellowstone River at Billings		U	U	U	U		U	U
Yellowstone River at Custer				U	U			U
Yellowstone River at/near Forsyth		U	D,U	D,U			U	U
- Forsyth WWTP		D	D					
- Rosebud Creek		D	D					
Yellowstone River at Far West FAS			D		U	D	D	
Yellowstone River above Cartersville Canal			D	D				
Yellowstone River at/near Miles City	D	U	D,U	D,U	U	D	D,U	U
- Tongue River		D,U	D,U					

Table 6-4. Data collection matrix for the DEQ 2007 and USGS 2000 monitoring programs.
Comparisons between the USGS 2000 and DEQ 2007 effort.

Monitoring Location[1]	Climate	Streamflow	Water Chemistry[2]	Diurnal WQ[3]	Transport[4]	Kinetics[5]	Benthics[6]	Light[7]
- Miles City WWTP		D	D					
Yellowstone River at Pirogue Island			D			D	D	
Yellowstone River below Pirogue Island			D	D				
Yellowstone River at Kinsey FAS			D	D	U			
Yellowstone River above Powder River			D	D		D	D	
- Powder River		D,U	D					
Yellowstone River at/near Terry			D	D,U	U		U	U
Yellowstone River above O'Fallon Creek			D	D	U	D	D	
- O'Fallon Creek		D	D					
Yellowstone River at Glendive		U	D,U	D	U		U	U
Yellowstone River at Sidney		U		U			U	U

[1]U = monitored by USGS in 2000 or 2008, D = monitored by DEQ in 2007
[2]Equal width integrated (EWI) samples
[3]YSI model 6600EDS sonde or equivalent
[4]From USGS dye-tracer study in 2008
[5]Productivity using light-dark bottles; reaeration using delta method (Chapra and Di Toro, 1991).
[6]Benthic algae and SOD
[7]Photosynthetically active radiation (PAR) at depth

Table 6-5. Water chemistry comparisons for the DEQ 2007 and USGS 2000 data programs.

Constituent[1]	Mainstem	Point Source	Tributary	Irrigation
Total Nitrogen	D,U	D	D,U	D
Nitrate plus Nitrite (NO_2^-+NO_3^-)	D,U	D	D,U	D
Ammonia (NH_4^+)	D,U	D	D,U	D
Total Phosphours	D,U	D	D,U	D
Soluble Reactive Phosphorus (SRP)	D,U	D	D,U	D
Total Suspended Solids (TSS)	D,U	D	D,U	D
Volatile Suspended Soils (VSS)	D	D	D	D
CBOD5-day	D	D		
Seston Stoichiometry	D			
Phytoplankton	D,U		U	

[1]U = monitored by USGS in 2000, D = monitored by DEQ in 2007.

Ambient conditions during these two periods are shown in **Figure 6-3**. Both climate (as represented by mean daily air temperature and precipitation) and streamflow (as annual hydrograph) compare favorably during both studies. The meteorological conditions were very similar to that of the 1970-2001 climate normals (NOAA, 2009). Streamflow was well below average both years, between the 5th and 25th percentile. This is roughly equivalent to somewhere between a 10 and 20 year low-flow condition (McCarthy, 2004).

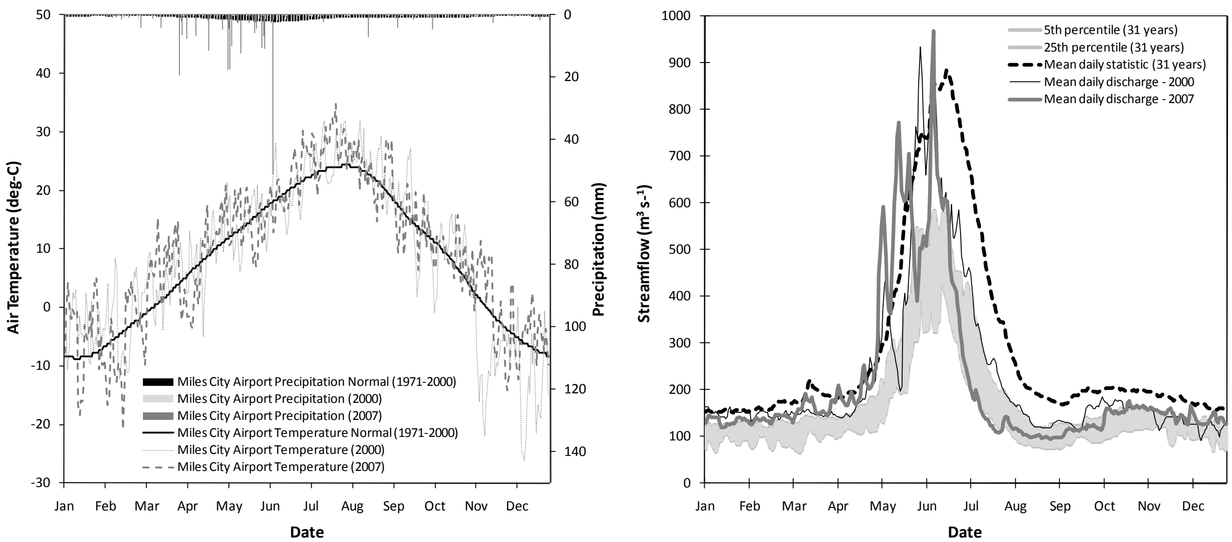


Figure 6-3. Conditions encountered during the 2000 and 2007 field data collection efforts.
(Left panel) Climatological data. (Right panel) Streamflow hydrology.

Comparative water quality results for each period (August 2000, August 2007, and September 2007) are shown in **Table 6-6**. Again, conditions were similar both years (e.g., temperature, DO, SC, pH), with noted exceptions of soluble nitrogen (NO_2+NO_3), TSS, phytoplankton, and temperature. Overall, September was the most different of all periods as temperature was approximately 4-5°C cooler and phytoplankton concentrations were about half of the other time-frames.

Table 6-6. Summary of water quality data during the 2000 and 2007 field data collection efforts.

Location and Monitoring Period	Temperature (°C)	pH	SC (µS cm^{-1})	DO (mg L^{-1})	Turbidity (ntu)	TSS (mg L^{-1})	TN (mg L^{-1})	NO_2+NO_3 (mg L^{-1})	TP (mg L^{-1})	SRP (mg L^{-1})	Phyto (µg L^{-1})
Forsyth											
Aug. 2000	21.4	8.55	673	7.05	6.4	18[1]	0.39	<0.05	0.031	<0.01	6.9
Aug. 2007	20.8	8.58	767	8.06	28	31	0.51	0.104	0.042	<0.004	8.8
Sept. 2007	16.2	8.65	693	8.97	14	20	0.47	0.144	0.040	0.003	3.9
Miles City											
Aug. 2000	20.4	8.58	692	7.91	13	23	0.32	<0.05	0.029	<0.01	6.0
Aug. 2007	21.6	8.72	731	9.01	17	31	0.46	0.003	0.051	<0.004	11.2
Sept. 2007	16.7	8.74	695	9.32	15	42	0.46	0.069	0.046	<0.004	3.7
Terry[2]											
Aug. 2000	18.1	8.58	660	8.37	12	23	0.39	<0.05	0.037	<0.01	5.3
Aug. 2007	21.2	8.55	771	8.76	17	32	0.45	0.002	0.045	<0.004	11.2
Sept. 2007	16.5	8.60	655	9.65	25	26	0.34	0.018	0.034	<0.004	4.8
Glendive[3]											
Aug. 2000	20.0	8.42	739	8.05	19	30	0.39	<0.05	0.038	<0.01	5.7
Aug. 2007	20.7	8.42	822	8.24	38	51	0.44	0.006	0.057	<0.004	15.6
Sept. 2007[4]	16.2	8.45	772	8.96	25	107	0.45	0.014	0.045	<0.004	12.1

[1]Two values reported, 8/18/2000 TSS = 18 mg L^{-1}, 8/26/2000 TSS = 58 mg L^{-1}.
[2]Diurnal data at Terry collected in September 2000.
[3]No diurnal data collected at Glendive, substitute Sidney observations.
[4]Grab sample (no EWI), suggestive of why the data is so different.

6.5 Other Pertinent Information

A considerable amount of other work has been done on the Yellowstone River; far more than can adequately be addressed in this document. Unfortunately, most of this information is not useful for supporting water quality model development. For example, Knudson and Swanson (1976) measured diel dissolved oxygen at a number of sites in August of 1976, but collected no water chemistry data. The Montana Board of Health (Montana Board of Health, 1952; Montana Board of Health, 1956) did significant work on the river in August and September of 1952 and 1955, including substantial water quality data collections, however, the analytical results were of poor resolution due to the laboratory methods available at the time. Diurnal measurements were not made either. Lastly, many efforts have been completed, mainly in the Billings region (Bahls, 1976a; Karp et al., 1977; Montana Board of Health, 1967), but it is not clear whether they are directly comparable to our study area. In most instances, they are absent of the data requirements for modeling anyway.

In any case, the work identified previously, along with any not specifically mentioned here but perhaps cited in other parts our report, provide useful information to support the modeling, but do not directly aid in model development. Their utility lies in such things as filling data gaps, estimating model rates, or deriving an understanding of water quality responses.

6.6 Data Quality, Detection Limits, and Significant Figures

DEQ completed quality assessments of all data sources mentioned previously to the extent possible. These included: standard DEQ quality checks to evaluate information against historical conditions;

performing station comparisons, time-series validation, and checks for data outliers; and *posteriori* scrutiny with the model. The QC revealed correctable laboratory errors and other minor inconsistencies in the data. Overall, the data were generally of good quality. In instances where analytical detection limits were an issue (i.e., non-detect laboratory values), ½ the detection limit was used. Rounding and other significant-figure use conventions were also applied as outlined in Section 1050B of American Public Health Association (APHA, 2005). Data flags were considered on a case-by-case basis, and outliers were verified prior to use. For time-series, if there were minor periods of missing data or errant data, these were filled using standard scientific procedures such as the normal-ratio method (Linsley et al., 1982) or distributions from an adjacent station. If no suitable replacement data could be established, data were excluded altogether. Centrally tendency statistics were reported as geometric means, or medians, rather than averages to eliminate right data skew (i.e., lognormally distributed data).

7.0 MODEL SETUP AND DEVELOPMENT

This section identifies the physical attributes used in the Yellowstone River model setup and development. Included are things such as centerline flow path delineation, mass transfer locations, transport mechanisms, and air and water boundary interactions. General data types or sources used to define these inputs (**Table 7-1)** are described in the following sections.

Table 7-1. Data sources used in the lower Yellowstone River QUAL2K model development.

Data Type	Source(s)	Increment
Flow Path	1. Air photo assessment and lower Yellowstone River digitized centerline (AGDTM, 2004)	n/a
Streamflow	1. DEQ field observations 2. U.S. Geological Survey gaging stations 3. Buffalo Rapids Irrigation District pumping rates 4. DNRC Water Resource Surveys	Instantaneous Daily Daily n/a
Transport	1. DEQ field observations 2. U.S. Geological Survey travel time study; rating measurements	Instantaneous Hourly
Climate	1. Bureau of Reclamation AgriMET stations (Terry & Glendive) 2. DEQ weather station (Miles City) 3. Montana Department of Transportation Road Weather Information System station (RWIS) 4. National Weather Service stations (NOAA)	15-minute 15-minute varies hourly
Shade	1. DEQ shade analysis with Shadev3.0 model	hourly
Other boundary conditions (quality/quantity)	1. DEQ field observations 2. USGS field measurements	Instantaneous Instantaneous

7.1 FLOW PATH (CENTERLINE) DEFINITION

Aerial photography was used to define the low-flow centerline and establish gradient in the model. A number of aerial photo flights have been made on the river (**Table 7-2**) and we used the 2001 color-infrared (IR) flight as it was most similar to field conditions encountered during 2007 (from a hydrologic standpoint). The length was also already digitized (AGDTM, 2004) which was an added advantage.

Table 7-2. Aerial photography summary of the lower Yellowstone River.

Photo Series	Source	Photo Date(s)	Flow at Miles City (m^3s^{-1})	Gage Height (m)
2001 Color Infrared (IR)	NRCS	Aug. 3-5, 2001	107-121	0.79-0.85
2004/2007 Color Floodplain Mapping	LYRCC	Jul. 12 – Aug. 5, 2005	159-168	0.98-1.00
2005 NAIP	NAIP	Oct. 15 – Nov. 2, 2007	159-496	0.98-1.80
Field Conditions 2007	------	------	**106-120**	**0.79-0.85**

The channel length and associated river stationing (in kilometers) used for the modeling is shown in **Table 7-3**. Ascribed values make an excellent comparison against previous efforts by the Department of Natural Resources and Conservation (DNRC, 1976) and a separate DEQ quality assurance (QA) check [with the 2007 National Agriculture Imagery Program (NAIP) photography]. The overall difference between the three efforts is less than 1%. Thus we feel confident about our length estimate as well as the placement of model features such as incoming tributaries or point or nonpoint source withdrawals, and calibration locations.

Table 7-3. Representative flow path lengths of the lower Yellowstone River.
River stationing is based on distance downstream from the headwater boundary condition which in this case was Forsyth. Glendive was at the lower end of the study reach, 232.9 km from the origin.

Reach	2001 color- IR (km)	DNRC, 1976 (km)	DEQ QA, 2007 (km)
Forsyth to Rosebud Creek	22.6	22.0	22.8
Rosebud Creek to Tongue River	65.3	64.9	66.0
Tongue River to Powder River	57.7	55.8	56.8
Powder River to O'Fallon Creek	32.2	32.2	32.6
O'Fallon Creek to Glendive	55.1	59.7	57.1
Total Length	**232.9**	**234.6**	**235.3**

Gradient is also a necessary input for Q2K. Station and elevation information were determined with the centerline described previously and using a digital elevation model (DEM) of the project site[17]. ArcGIS TTools (Boyd and Kasper, 2003) was used to complete elevation sampling every 100-meters along the channel centerline. The results are shown in **Figure 7-1**. Overall, the profile is fairly consistent from Forsyth to Miles City (km 232 to 140), shifts between Miles City and Terry (river station 140 to 100 km), and then approximates prior conditions from Terry to Glendive (km 100-0). From review of the profile, 31 unique hydraulic reaches were identified for use in Q2K which included major slope breaks, breaks at tributaries, or rapids. These were picked out visually by DEQ, or were identified in other documents related to the morphology of the river (AGDTM, 2004). The rapids occurred at river kilometers 130, 125, 95, and 80, and are discussed in more detail in **Section 7.3**.

Lastly, aerial photography was used to determine additional channel properties including mean channel wetted width, bankfull width, etc. This information is described in more detail in **Section 7.3** as well as **Appendix C**. Values were averaged over 1-km increments to make reach specific estimates.

[17] The DEM was developed from light detection and ranging (LIDAR) data and channel bathymetric surveys from 2004 and 2007. Coordinate system and datum used for this effort were State-Plane NAD83 and NAVD88. Raw triangular irregular network (TIN) files were taken from the NRIS (NRIS, Montana Natural Resource Information System, 2009) and were converted to 2.5 meter resolution DEM which were subsequently mosaiced into a single contiguous DEM of the project site (from slightly upstream of Forsyth to downstream of Glendive). This included the addition of elevation data outside of the LIDAR and bathymetric survey area using the 10 m National Elevation Dataset (NED). The area where the bathymetric survey was completed in the lower river was smoothed to remove the undulating bed profile (see **Figure 7-1**).

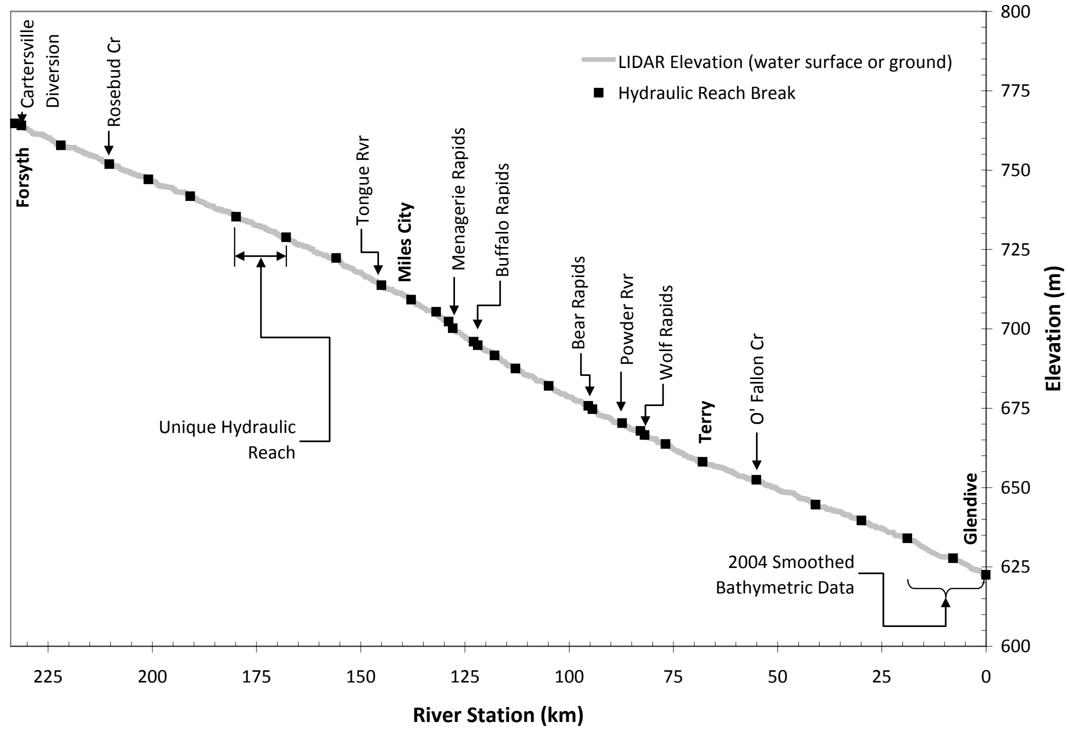


Figure 7-1. Longitudinal profile of the Yellowstone River.
Estimated from 2.5 meter DEM of the project site (see previous footnote for details on the DEM). Thirty-one hydraulic reaches were defined based on subtle changes in gradient. This included identification of several rapids in the project site.

7.2 STREAMFLOW

A steady-state streamflow balance was applied according to **Equation 7-1** for flow in the model where outflow of a gaged segment in $m^3\ s^{-1}$ ($Q_{gage,i}$) was equal to the sum of the inflow from the upstream gage ($Q_{gage,i-1}$), plus or minus any point source or diffuse inflows ($Q_{in,j}$) or abstractions ($Q_{ab,j}$).

(Equation 7-1)

$$Q_{gage,i} = Q_{gage,i-1} + Q_{in,i} - Q_{ab,i}$$

Meaningful input to **Equation 7-1** was provided from the 2007 field effort. Those who contributed to its development included DEQ, USGS, and the Buffalo Rapids Irrigation District (BRID). Sources, details, and assumptions regarding the streamflow water balance development are described in subsequent sections. A ten day average streamflow condition was used which reflects the time over which the water quality samples were collected.

7.2.1 Surface Water Summary

Aspects of the surface water balance are detailed in this section, i.e., any water that could be measured in flowing channels.

7.2.1.1 Mainstem River Flow

Mainstem river flow measurements were used to provide $Q_{gage,I}$ and $Q_{gage,i-1}$ in **Equation 7-1** and were taken from mean daily flows reported by USGS for the three active gages on the river: USGS 06295000 Yellowstone River at Forsyth, USGS 06309000 Yellowstone River at Miles City, and USGS 06327500

Yellowstone River at Glendive. Flows for these sites during the summer 2007 are shown in **Figure7-2** (left). Conditions were primarily steady-state during the 10-day data collection period as indicated by an average coefficient of variation of 2.5% and 1.8% for August and September respectively. Correlations between gage sites were good (r^2>0.90), with the exception of Glendive in early August. During this period irrigation varied between the gages and changed the ratio at various locations in the river. Transient conditions occurred only once (in September), defined by variation of greater than 10% per week. The shift was related to precipitation in the upper basin, cooler fall temperatures, and reductions in irrigation throughout the watershed. As identified previously, flows were quite low, between a seasonal 10- to 20-year low-flow condition (**Table 7-4**) which was based on McCarthy (2004).

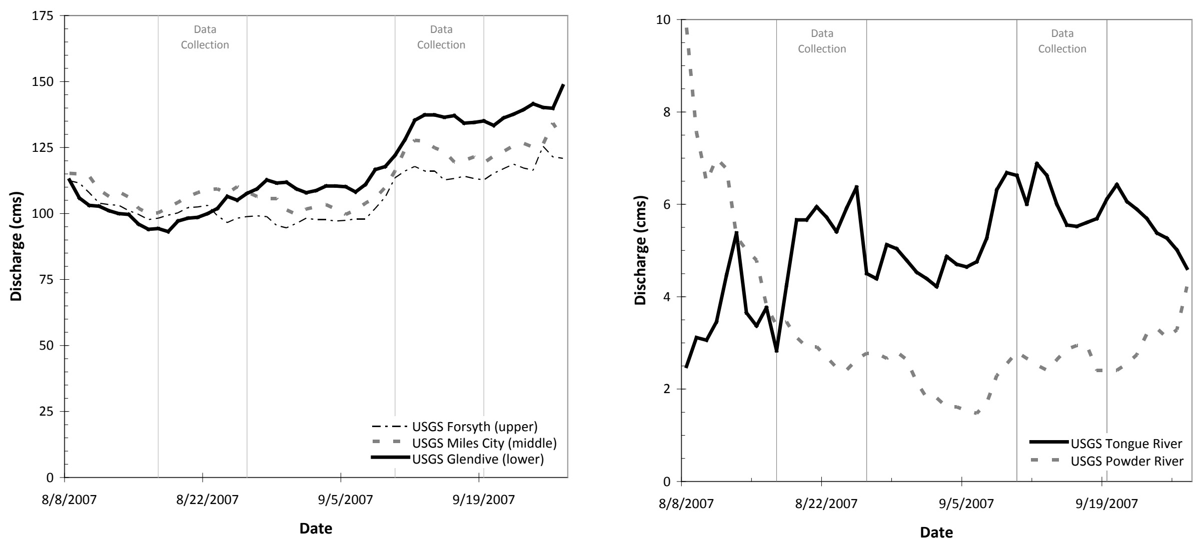


Figure 7-2. Surface water summary for the lower Yellowstone River during 2007.
(Left panel) Streamflow on the mainstem river for USGS 06295000 Yellowstone River at Forsyth (upper reach), USGS 06309000 Yellowstone River at Miles City (middle reach), and USGS 06327500 Yellowstone River at Glendive (lower reach). (Right panel) Streamflow but for the major gaged tributaries, which include USGS 06308500 Tongue River at Miles City and USGS 06326500 Powder River near Locate.

Table 7-4. Magnitude and probability of seasonal low flow for the Yellowstone River.
Data shown for the July-October seasonal low-flow period at USGS 06309000 Yellowstone River at Miles City[1]. For comparative purposes, flows during 2007 were approximately 100 $m^3 s^{-1}$.

Period of consecutive days	Discharge in $m^3 s^{-1}$ for indicated recurrence interval (yrs) and non-exceedance probability (%)			
recurrence interval non-exceedance probability	**2 50%**	**5 20%**	**10 10%**	**20 5%**
1	169	126	106	90
3	173	128	107	91
7	177	131	109	93
14	183	135	112	94
30	194	142	118	99

[1]Taken from McCarthy (2004) over 36 seasons of record.

7.2.1.2 Tributary Flow

Tributary flow to the Yellowstone River was identified as $Q_{in,j}$ in the water balance. It was somewhat more variable than the mainstem river and hydrographs for the Tongue and Powder rivers (which are the two major contributors to the lower Yellowstone River project reach) are shown in **Figure 7-2** (right).

The Powder River exhibited somewhat oscillatory but stable streamflow over the summer period with a coefficient of variation (COV) of 7.5% and 9.0% for August and September respectively. The Tongue River is reservoir regulated, and shows distinct operational shifts and somewhat higher COVs (10.4% and 9.3% respectively). Since both waterbodies comprise a very small percentage of the overall streamflow to the river (e.g., less than 5% each), their overall influence is minimal.

Other inflows or outflows of potential significance were also integrated to better describe the hydrologic regime of the watershed. These measurements are shown in **Table 7-5** and were made by either boat or wading with a Marsh-McBirney Model 2000 Flo-Mate solid state current meter (Rantz, 1982). Actual discharge measurement forms are located in **Appendix B**.

Table 7-5. Instantaneous field measurements completed by Montana DEQ during 2007.

Site	August Measured Flow ($m^3 s^{-1}$)	September Measured Flow ($m^3 s^{-1}$)	Change (%)
Cartersville Canal diversion	5.701	5.227	-8%
Forsyth WWTP	0.006	0.009	50%
Rosebud Creek	0.180	0.122	-32%
Cartersville Canal return flow	1.987	1.330	-33%
Tongue River[1]	3.822	6.037	58%
Kinsey Canal diversion	2.592	2.650	2%
Kinsey Canal return flow	0.101	0.791	683%
Shirley Canal return flow	0.500	0.461	-8%
Powder River	3.093	2.235	-28%
O'Fallon Creek	0.101	0.166	64%

[1]QA check completed for this site with USGS mean daily reported streamflow.

7.2.1.3 Unmeasured Tributaries

Over 80 smaller tributaries contribute to the lower Yellowstone River between Forsyth and Glendive (DNRC, Montana Department of Natural Resources and Conservation, 1976). These range in size from a few square kilometers to over 33,000 km^2. They are problematic in that their sheer number alone would preclude effective monitoring for modeling. As a result, DEQ monitored only the largest ones (e.g.., those ≥3,000 km^2, as described in the previous section) and estimated the rest through regression (termed here 'unmeasured tributaries').

Twelve previously gaged sites (USGS, 2008) within the study area were used in to develop a low-flow drainage area regression relationship. Mean streamflow ($m^3 s^{-1}$) for the month of August and September (as applicable to the calibration and validation models) was regressed against drainage area (km^2) to determine the net contribution of inflow from ungaged sites. These estimates were then corrected to 2007 conditions based on the ratio of the mean monthly flow during 2007 and that of the overall period. Sites used in linear regression model are shown in **Table 7-6**.

Predicted flows from this exercise provided a good fit (r^2>0.90, see **Appendix B**) and were applied to the net unmonitored area between each mainstem gage site based on the difference between reported areas less any area accounted for by gaged tributaries[18]. The net unmonitored tributary inflow to the Yellowstone River from this method was small, approximately 1.243 and 1.119 $m^3 s^{-1}$ during August and September respectively (or 1.2 and 1.0% of the overall headwater boundary condition).

Table 7-6. Sites used in estimation of unmonitored tributaries.
Data taken from NWIS (accessed 9/22-23, 2008).

Site Id	Description	Drainage Area (km^2)	Period of Record
06296003	Rosebud Creek at mouth near Rosebud MT	3,371	1974-10 to 2006-09
06296100	Snell Creek near Hathaway MT	27	1981-10 to 1985-09
06308500	Tongue River near Miles City MT (pre-dam record)	11,751	1929-04 to 1932-09
06309075	Sunday Creek near Miles City MT	1848	1974-10 to 1984-09
06309079	Muster Creek near Kinsey MT	74	1978-03 to 1980-08
06309145	Custer Creek near Kinsey MT	391	1978-03 to 1980-08
06326500	Powder River near Locate MT	33,831	1938-03 to 2007-09
06326555	Cherry Creek near Terry MT	927	1979-09 to 1994-09
06326850	O'Fallon Creek at Mildred MT	3,614	1975-09 to 1978-09
06326952	Clear Creek near Lindsay MT	261	1982-03 to 1988-09
06327000	Upper Sevenmile Creek near Glendive MT	NA	1921-03 to 1922-05
06327450	Cains Coulee at Glendive MT	10	1992-05 to 2004-09

7.2.1.4 Municipalities

Domestic water withdrawals or waste water treatment plant (WWTP) inflows were also incorporated (**Table 7-7**). Information was either directly measured in the field, was provided by request from the discharger, or was retrieved from monthly reports of finished clearwell effluent or Montana Pollutant Discharge Elimination System (MPDES) discharge monitoring reports (DMRs).

Table 7-7. Municipal discharges in the lower Yellowstone River study reach during 2007.

Municipality	Type	Aug 17-26 Transfer ($m^3 s^{-1}$)	Sep 11-20 Transfer ($m^3 s^{-1}$)	Data Source or Comment[1]
City of Forsyth	Water Intake WWTP Outfall	-0.022 +0.011	-0.017 +0.011	Clearwell logs From City/Pat Zent
City of Miles City	Water Intake WWTP Outfall	-0.102 +0.052	-0.089 +0.048	Clearwell logs From City/Allen Kelm
City of Terry	WWTP Outfall	no discharge	+0.004	Field measured
Fallon-Prairie County	WWTP Outfall	no discharge	no discharge	N/A
City of Glendive	Water Intake WWTP Outfall	--- ---	--- ---	DS of study reach DS of study reach

[1] Water intake data taken from monthly reports of finished clearwell effluent.

7.2.1.5 Irrigation

Large-scale irrigation exchanges ($Q_{ab,j}$ and $Q_{in,j}$, depending on inflow or outflow) were also incorporated because of their known influence on water quality (Law and Skogerboe, 1972; Miller et al., 1978;

[18] For example, the gaged area at Forsyth is 103,933 km^2 while at Miles City it is 124,921 km^2. Hence, the unaccounted area is 20,988 km^2. However, both Rosebud Creek (3,371 km^2) and the Tongue River (13,972 km^2) enter between these two gages. Thus the actual ungaged area is 3,645 km^2.

Ongley, 1996). Major units were identified through review of historical DNRC Water Resource Surveys (WRS). Those believed to be of primary importance are identified in **Table 7-8**.

Table 7-8. Summary of major irrigation units on the lower Yellowstone River.

Irrigation Unit[1]	Irrigated Area at time of publication (hectares)	Maximum Irrigated Area (hectares)	County	Publication Date
Cartersville Irrigation District	3,651	4,243	Rosebud	1948
Baringer Pumping Project	380	467	Rosebud	1948
Private Irrigation (pumps from YR) [2]	1,160	1,870	All	Various
T & Y Irrigation District (return flow)[2]	3,598	4,077	Custer	1948
Kinsey Irrigation Company	2,511	2,827	Custer	1948
Shirley Unit - Buffalo Rapids	1,823	2,018	Custer	1948
Terry Unit-Buffalo Rapids	1,282	1,357	Prairie	1970
Fallon Unit – Buffalo Rapids	1,204	1,238	Prairie	1970
Glendive Unit – Buffalo Rapids	5,758	6,152	Prairie/Dawson	1970

[1] As described in the Water Resource Surveys.
[2] Data gap, estimated as described in next paragraph.

Despite our best efforts, we were unable to monitor all of the sites identified in **Table 7-8**. To make reasonable estimates for the missing information, a regression approach similar to that described for the tributaries was used. In this instance, regressions were carried out using maximum irrigated area (to characterize irrigation withdrawals and return flow) and results were fairly good (r^2=0.91 and 0.76) as shown in **Table 7-9**. The actual regression models are detailed in **Appendix B**. An estimate for lateral return flow was also made and is detailed in the next paragraph.

Table 7-9. Summary of irrigation water transfers on the Yellowstone River during 2007.

Irrigation Unit	Period	Irrigation Withdrawal (cms)	Main Canal Return Flow (cms)	Estimated Lateral Return Flow (cms)
Cartersville Irrigation District	Aug 07 Sep 07	5.975 2.519	1.987 1.330	1.052 [est] 0.990 [est]
Baringer Pumping Project	Aug 07 Sep 07	0.635 [est] 0.355 [est]	0.000 [est] 0.000 [est]	0.070 [est] 0.159 [est]
Private Irrigation (pumps from YR)	Aug 07 Sep 07	2.543 [est] 1.421 [est]	0.164 [est] 0.311 [est]	0.435 [est] 0.468 [est]
T & Y Irrigation District (return flow)	Aug 07 Sep 07	N/A	1.407 [est] 1.039 [est]	N/A
Kinsey Irrigation Company	Aug 07 Sep 07	2.572 2.650	0.101 0.797	0.684 [est] 0.678 [est]
Shirley Unit - Buffalo Rapids	Aug 07 Sep 07	3.228 1.420	0.454 0.440	0.401 [est] 0.431 [est]
Terry Unit-Buffalo Rapids	Aug 07 Sep 07	1.584 0.528	0.000 0.000	0.255 [est] 0.306 [est]
Fallon Unit – Buffalo Rapids	Aug 07 Sep 07	2.039 1.359	0.000 0.027	0.229 [est] 0.283 [est]
Glendive Unit – Buffalo Rapids	Aug 07 Sep 07	9.232 5.295	N/A	1.548 [est] 1.410 [est]

[est]=Values estimated using regression procedure; n/a – not applicable, location outside of project area.

Lateral return flows in **Table 7-9** are entirely estimated. They comprise irrigation waste drain laterals which are small canals that branch off the main canal and could not be measured due to their diffuse nature. A study by Montana State University (MSU) was used to fill this deficiency (H. Sessoms, personal communication)[19] by relating irrigated area [as determined by landcover (e.g., pasture/hay and row crops) (Homer et al., 2004)] with the return flow values provided by MSU. The regressions were quite good for August (r^2=0.96), and poor for September (r^2=0.25), which reflects the variability in return flow at the close of the irrigation season.

7.2.2 Groundwater

The contribution of groundwater from **Equation 7-1** is the only term left in the water balance (i.e., either $Q_{in,j}$ or $Q_{ab,j}$ depending on conditions). Accretion was estimated according to **Equation 7-2**, where g_w is the groundwater contribution in [m^3 s^{-1}] and $Q_{gage,I}$, $Q_{gage,i\text{-}1}$, $Q_{in,j}$, and $Q_{ab,j}$ were defined previously. Given the short duration of the study, it was assumed that there was no change in storage (ΔS).

(Equation 7-2)

$$g_w = \Delta S + Q_{gage,i} - Q_{gage,i-1} + Q_{in,i} - Q_{ab,i}$$

Groundwater inflow comprised most of the influent (i.e., $Q_{in,j}$) water to the study reach (40-50%) but was still only a small percentage (10-15%) of the total flow in the river ($Q_{gage,I}$, $Q_{gage,i\text{-}1}$). Most of the exchange likely comes from the shallow hydrologic unit which is less than 200 feet below the land surface (Smith et al., 2000). The primary mechanism of recharge is believed to be leaky irrigation ditches or regional groundwater flow systems (Moulder et al., 1953; Moulder and Kohout, 1958; Torrey and Kohout, 1956; Torrey and Swenson, 1951), which seems to fit with the spatial orientation of our field observations.

7.2.3 Evaporation

Evaporation is not computed in Q2K20 but DEQ made estimates to determine its significance. Published pan data from the Huntley Experimental Station (244345) and Sidney Airport (247560) were used. Pan coefficients from Farnsworth, et al., (1982)21 were used to correct the data to free water surface (FWS) evaporation which yielded daily rates of 4 and 3 mm day^{-1} (0.16, 0.12 inches day^{-1}) for August and September respectively. Such estimates compare well with Pochop, et al., (1985) and indicate

[19] The waste drain lateral return flow study was completed on Clear Creek, Sand Creek, and Whoopup Creek. These data were extrapolated to other areas in the project site. According to Schwarz (1999), the Buffalo Rapids Unit II has a conveyance efficiency of 89.3% while Unit I is only 73.7% efficient. Complete details regarding the irrigation estimates can be found in **Appendix B**.

[20] A beta version of Q2K is now available with this functionality (at the time of final publication of this report) but it is not practical to apply the new version of the model given the significant effort to reconfigure the report and associated modeling results.

[21] A pan coefficient of 0.72 was used which compares reasonably with most work in the United States (Linsley et al., 1982). It does not compare that well with reported values for Fort Peck Reservoir (0.64 and 1.21 each month) (Army Corps of Engineers, 2003). Given the inability of DEQ to verify the source of the Corps data [i.e., their cited values could not be found in Farnswoth and Thompson (1982)] where it supposedly should have been found], DEQ used the standard NOAA methodology instead.

approximately 1.710 $m^3 s^{-1}$ (60.4 $ft^3 s^{-1}$) and 1.318 $m^3 s^{-1}$ (46.5 $ft^3 s^{-1}$) of evaporation occur during each period in the river (which were applied as a diffuse abstraction in the model)[22].

7.2.4 Water Balance During Summer 2007

The water balance as determined from prior information is shown in **Figure 7-3**, **Table 7-10**, and **Table 7-11** for 2007. Its most important consideration was flow at the upstream boundary (Forsyth) which comprised nearly 70% of the inflow to the study reach. Of the other inflows (normalized to each other), groundwater was the biggest contributor at 41% and 52%, followed by the Tongue River (17% and 16%), unmeasured waste drains (16% and 13%), and the Powder River (8% and 6%). Irrigation and domestic water withdrawals were significant and amounted to 30 and 15% of the overall flow in the river (in August and September, respectively). Consequently a large portion of water in the river is removed for the purpose of irrigation. The largest diversions were the Buffalo Rapids Irrigation District which removed over 14 $m^3 s^{-1}$ (≈500 $ft^3 s^{-1}$) (including the Shirley, Terry, and Glendive units) followed by the Cartersville Irrigation District which removed nearly 6 $m^3 s^{-1}$ (≈200 $ft^3 s^{-1}$). Some of this water makes its way back to the river as return flow.

[22] It should be noted that the way in which we have applied evaporation in the model is a slight simplification. We have implemented it as a mass removal, which also removes constituent mass. The model is being modified to make such changes (personal communication, S. Chapra).

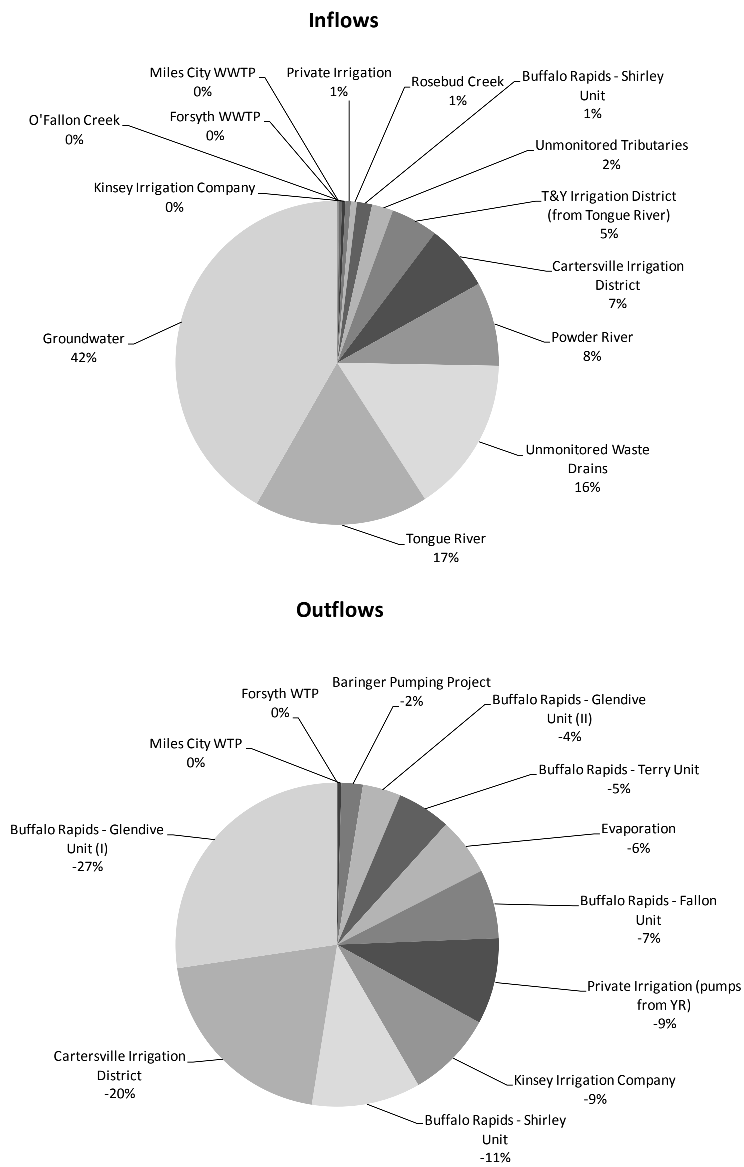


Figure 7-3. Graphical summary of water exchanges in the Yellowstone River during August 2007. (Top panel) Summary of inflows to the river during August of 2007. Note that the values shown are relative to one another; river flow at the upstream boundary (at Forsyth) accounted for 70% of the total inflow (thus inflow fractions represent the remaining 30%). (Bottom panel). Summary of outflows (i.e., diversions) during August of 2007. Withdrawals shown as negative to reinforce the fact that water is being removed from the river.

Table 7-10. Tabular summary of Yellowstone River water balance for August 2007.

Unit	Site Name	Flow ($m^3 s^{-1}$)	Balance	Groundwater ($m^3 s^{-1}$)	Comment
1	USGS (06295000) Yellowstone at Forsyth	99.849	99.849	+7.259	Avg. 8/17-26
	Forsyth WTP[1]	-0.022	99.827		8/17
	Cartersville Irrigation District DVT	-5.975	93.852		Avg. 8/17-26
	Forsyth WWTP[2]	+0.011	93.864		Avg. 8/17-26
	Rosebud Creek	+0.180	94.044		8/18
	Cartersville Irrigation District RTN	+1.987	96.031		8/20
	Baringer Pumping Project DVT	-0.635	95.396		Avg. 8/17-26
	Baringer Pumping Project RTN	+0.000	95.396		Avg. 8/17-26
	Private Irrigation (pumps from YR)	-2.543	92.853		Avg. 8/17-26
	Private Irrigation (pumps from YR)	+0.164	93.017		Avg. 8/17-26
	Miles City WTP[1]	-0.102	92.915		Avg. 8/17-26
	Tongue River	+5.227	98.142		Avg. 8/17-26
	Unmonitored Tributaries	+0.173	98.316		Avg. 8/17-26
	Unmonitored Waste Drains	+1.558	99.873		Avg. 8/17-26
	Evaporation	-0.601	99.273		Avg. 8/17-26
	USGS (06309000) Yellowstone at Miles City	106.532	99.273		Avg. 8/17-26
2	Miles City WWTP[2]	0.052	106.584	+4.627	Avg. 8/17-26
	Kinsey Irrigation Company DVT	-2.572	104.012		8/18
	T&Y Irrigation District RTN (from Tongue R.)	1.407	105.419		Avg. 8/17-26
	Buffalo Rapids - Shirley Unit DVT[2]	-3.228	102.191		Avg. 8/17-26
	Kinsey Irrigation Company RTN	0.101	102.292		8/18
	Buffalo Rapids - Shirley Unit RTN[2]	0.454	102.746		8/23
	Powder River	2.519	105.266		Avg. 8/17-26
	Buffalo Rapids - Terry Unit DVT[2]	-1.584	103.682		Avg. 8/17-26
	Terry WWTP	0.000	103.682		Avg. 8/17-26
	Unmonitored Tributaries	0.227	103.909		Avg. 8/17-26
	Unmonitored Waste Drains	1.340	105.249		Avg. 8/17-26
	Evaporation	-0.569	104.680		Avg. 8/17-26
	USGS (06326530) Yellowstone near Terry	109.307	104.680		Avg. 8/17-26
3	Buffalo Rapids - Terry Unit RTN	0.000	109.31	+0.647	Avg. 8/17-26
	O'Fallon Creek	0.082	109.39		8/26
	Buffalo Rapids - Fallon Unit DVT[2]	-2.039	107.35		Avg. 8/17-26
	Buffalo Rapids - Fallon Unit RTN	0.000	107.35		Avg. 8/17-26
	Buffalo Rapids - Glendive Unit (I) DVT[2]	-8.099	99.25		Avg. 8/17-26
	Buffalo Rapids - Glendive Unit (II) DVT[2]	-1.133	98.12		Avg. 8/17-26
	Unmonitored Tributaries	0.242	98.36		Avg. 8/17-26
	Unmonitored Waste Drains	1.778	100.14		Avg. 8/17-26
	Evaporation	-0.541	99.60		Avg. 8/17-26
	USGS (06327500) Yellowstone at Glendive	100.245	99.60		Avg. 8/17-26

[1]From monthly reports of finished clearwell effluent.
[2]Provided directly by city or irrigation district.

Table 7-11. Tabular summary of Yellowstone River water balance for September 2007.

Unit	Site Name	Flow ($m^3 s^{-1}$)	Balance	Groundwater ($m^3 s^{-1}$)	Comment
1	USGS (06295000) Yellowstone at Forsyth	114.744	114.744	+3.459	Avg. 9/11-20
	Forsyth WTP[1]	-0.017	114.727		8/17
	Cartersville Irrigation District DVT	-2.519	112.208		Avg. 9/11-20
	Forsyth WWTP[2]	+0.011	112.219		Avg. 9/11-20
	Rosebud Creek	+0.122	112.341		9/12
	Cartersville Irrigation District RTN	+1.330	113.671		9/15
	Baringer Pumping Project DVT	-0.355	113.316		Avg. 9/11-20
	Baringer Pumping Project RTN	+0.000	113.316		Avg. 9/11-20
	Private Irrigation (pumps from YR)	-1.421	111.895		Avg. 9/11-20
	Private Irrigation (pumps from YR)	+0.311	112.206		Avg. 9/11-20
	Miles City WTP[1]	-0.089	112.117		Avg. 9/11-20
	Tongue River	+6.043	118.160		Avg. 9/11-20
	Unmonitored Tributaries	+0.212	118.372		Avg. 9/11-20
	Unmonitored Waste Drains	+1.617	119.989		Avg. 9/11-20
	Evaporation	-0.463	119.526		Avg. 9/11-20
	USGS (06309000) Yellowstone at Miles City	122.985	119.526		Avg. 9/11-20
2	Miles City WWTP[2]	+0.048	123.033	+2.983	Avg. 9/11-20
	Kinsey Irrigation Company DVT	-2.650	120.383		9/11
	T&Y Irrigation District RTN (from Tongue R.)	+1.039	121.423		Avg. 9/11-20
	Buffalo Rapids - Shirley Unit DVT[2]	-1.420	120.002		Avg. 9/11-20
	Kinsey Irrigation Company RTN	+0.797	120.799		9/11
	Buffalo Rapids - Shirley Unit RTN[2]	+0.440	121.240		9/16
	Powder River	+2.206	123.445		Avg. 9/11-20
	Buffalo Rapids - Terry Unit DVT[2]	-0.528	122.917		Avg. 9/11-20
	Terry WWTP	+0.004	122.921		Avg. 9/11-20
	Unmonitored Tributaries	+0.268	123.190		Avg. 9/11-20
	Unmonitored Waste Drains	+1.415	124.605		Avg. 9/11-20
	Evaporation	-0.440	124.165		Avg. 9/11-20
	USGS (06326530) Yellowstone near Terry	127.147	124.165		Avg. 9/11-20
3	Buffalo Rapids - Terry Unit RTN	0.000	127.147	+12.629	Avg. 9/11-20
	O'Fallon Creek	0.166	127.314		9/11
	Buffalo Rapids - Fallon Unit DVT[2]	-1.359	125.954		Avg. 9/11-20
	Buffalo Rapids - Fallon Unit RTN	0.027	125.981		Avg. 9/11-20
	Buffalo Rapids - Glendive Unit (I) DVT[2]	-5.295	120.686		Avg. 9/11-20
	Buffalo Rapids - Glendive Unit (II) DVT[2]	0.000	120.686		Avg. 9/11-20
	Unmonitored Tributaries	0.285	120.971		Avg. 9/11-20
	Unmonitored Waste Drains	1.693	122.664		Avg. 9/11-20
	Evaporation	-0.415	122.249		Avg. 9/11-20
	USGS (06327500) Yellowstone at Glendive	134.878	122.249		Avg. 9/11-20

[1]From monthly reports of finished clearwell effluent.
[2]Provided directly by city or irrigation district.

7.3 Hydraulics and Mass Transport

After the flow balance was finalized, mass transport functions (i.e., for advection and dispersion) were determined. These can be calculated in one of three ways in the model: weirs, rating curves, or

Manning's equation. For sharp-crested weirs, flow is related to head by **Equation 7-3** where B_w = width of the weir [m] and H_h = height of the water flowing over the weir [m]. The equation is then rearranged to solve for the depth upstream of the weir (for the purpose of advection and gas transfer computations). This method was used for the Cartersville Diversion Dam near Forsyth, MT.

(Equation 7-3) $$Q_i = 1.83 B_w H_h^{3/2}$$

At other locations, rating curves were employed. In the rating curve approach, the empirical coefficients *a* and *b*, and exponents α and β are used to relate depth *H* [m] and velocity *U* [m] to streamflow *Q* [m^3 s^{-1}] through the power relationships shown in **Equation 7-4** and **Equation 7-5** (Leopold and Maddock, 1953). The continuity equation then used to compute the remaining hydraulic properties including cross-sectional area, top width, surface area, volume, and hydraulic residence time.

(Equation 7-4) $$U = aQ^b$$

(Equation 7-5) $$H = \alpha Q^\beta$$

Also represented in the rating curves were natural grade controls (i.e., rapids). These had been identified previously by others (AGDTM, 2004) and include Menagerie Rapids, Buffalo Shoals, Bear Rapids, and Wolf Rapids. All are between Miles City and Glendive and result from entrenchment in erosion resistant sandstones and shales of the Fort Union Formation. Their location and associated features are shown in **Table 7-12.** They were incorporated into the model through adjustment of rating curve properties thereby making them fast and shallow.

Table 7-12. Locations of natural grade controls on the Yellowstone River.

Name	Q2K Station (km)	Approximate Location	Estimated Depth (m)	Estimated Velocity (m s^{-1})
Menagerie Rapids	128.9	12 miles DS of Tongue River	0.56	0.88
Buffalo Shoals	122.9	Kinsey	0.63	0.75
Bear Rapids	95.4	20 miles DS of Buffalo Shoals	0.79	0.79
Wolf Rapids	82.9	3 miles DS of Powder River	0.56	0.77

In determining the rating curve relationships described previously, a number of methods have been proposed. This includes physical field measurement of widths, depths, and velocities (Drolc and Koncan, 1996; Park and Lee, 2002; Van Orden and Uchrin, 1993), output from water surface profile models (Dussaillant et al., 1997; Tischler et al., 1985), and residence time/dye tracer fluorescence studies (Kuhn, 1991). A combination of methods were used in the Yellowstone River work. Subsequent lines of evidence included:

- Field observations of hydraulic properties at specified transects during 2007, (detailed in **Section 7.3.1**).
- Compilation of USGS rating measurements to evaluate depth- and velocity-discharge curves, (detailed in **Section 7.3.2**).
- GIS analyses of historical low-flow aerial photographs to assess channel hydraulic conditions, (detailed in **Section 7.3.3**).
- Dye tracer and associated travel time studies, (detailed in **Section 7.3.4**).

7.3.1 Field Observations of Hydraulic Properties

Width, depth, and cross-sectional area were measured at 23 transect locations between DEQ and USGS to provide ground-truth data for model mass transport. Measurements were made using a sounding weight, fiberglass tape, and laser range finders, or were surveyed using a total-station and fiberglass rod. In some instance, measurements were made at bridges. The channel approach angle and associated correction was necessary in such instances to account for bridge skew.

Measurements are shown in **Table 7-13**. Cross-sectional plots for each of these sections are in **Appendix B** and are also discussed in **Section 10.0** regarding the application of AT2K.

Table 7-13. Hydraulic property transects within the lower Yellowstone River study reach.

Monitoring Site	Width (m)	Depth (m)	Area (m^2)
Yellowstone River at Forsyth Bridge[1]	124	2.58	321
Yellowstone River at Old Forsyth Bridge[1]	81	3.70	300
Yellowstone River at Rosebud West FAS (e.g. near Forsyth)[1]	102	3.4	348
Yellowstone River at Far West FAS (near Rosebud)	117	0.9	104
Yellowstone River at Rosebud Bridge	145	1.98	286
Yellowstone River at Paragon Bridge	312	0.91	312
Yellowstone River at Ft. Keogh Bridge (1902 Bridge)	179	1.59	285
Yellowstone River below 1902 Bridge US of Tongue River	132	1.5	194
Yellowstone River at Highway 59 Bridge (at Miles City)	171	1.06	182
Yellowstone River at Pirogue Island (near Miles City)	134	0.9	119
Yellowstone River at Kinsey Bridge	187	0.93	174
Yellowstone River at Kinsey FAS	198	0.7	132
Yellowstone River US of Powder River	112	2.1	236
Yellowstone River US of Calypso Bridge	120	1.1	136
Yellowstone River at Calypso Bridge	130	1.58	206
Yellowstone River at Terry Highway Bridge	129	1.48	191
Yellowstone River US of O'Fallon Creek	174	1.4	235
Yellowstone River at Fallon Interstate Bridge	183	1.21	222
Yellowstone River at Fallon Frontage Bridge	183	1.20	220
Yellowstone River near Fallon Bridge	196	1.1	220
Yellowstone River at Glendive RR Bridge	339	2.88	977
Yellowstone River above Bell St. Bridge (e.g. Glendive)	133	1.6	210
Yellowstone River at Bell St. Bridge (e.g. Glendive)	141	1.84	141

[1] In backwater of Cartersville diversion dam.

In addition to the previous measurements, one low-head diversion dam (i.e., the Cartersville diversion dam near Forsyth) was also surveyed. This was done to estimate weir properties and storage upstream of the weir. The structure consisted of riprap capped by concrete (U.S. Fish and Wildlife Service, 2008) and based on measurements on the south (right bank) it was 1.6 meters (5.3 feet) high and 236 meters wide (using an automatic level and laser range finder). Depth of water flowing over the weir was 0.3 meters (0.95 feet). The Cartersville Irrigation District was contacted to verify these field measurements, however, no information existed (P. Ash, personal communication).

More recently however DOW-HKM Engineering has conducted fish passage studies of the structure. Based on field topographic surveys and 2-D hydraulic modeling at the site, they believe the dam to be 2.1 meters (7 ft) high with a crest elevation of 2507 feet above mean sea level (amsl) and a base of 2500 feet amsl (G. Elwell, personal communication). These values were subsequently used in the modeling.

HKM drawings of the dam are shown in **Appendix B** and oxygenation coefficients for water quality and dam-type were selected by DEQ to be 1.6 and 0.75 which are representative of slightly polluted waters and a round broad-crest weir.

7.3.2 USGS Rating Measurements

USGS field measurements[23] were compiled from NWIS to provide data to estimate the coefficient and exponent of **Equation 7-4** and **Equation 7-5**. These were subsequently regressed against discharge for all gages in the project site (**Table 7-14**)[24]. The regression results are shown in **Figure 7-4**.

Table 7-14. USGS gage sites having rating measurement data on the Lower Yellowstone River.

Description	Station ID	Observations (*n*)	Period
Yellowstone River at Billings MT	06214500	320	1968-2010
Yellowstone River at Forsyth MT	06295000	229	1953-2010
Yellowstone River at Miles City MT	06309000	268	1974-2010
Yellowstone River at Glendive MT	06327500	40	2002-2010
Yellowstone River at Sidney MT	06329500	331	1967-2010

A best-fit curve was determined using least squares in Excel™ and an envelope of possible outcomes (i.e. upper and lower bounds) was identified to represent uncertainty in the observations. Due to the relative uniformity and similarity of the river, a single exponent was deemed sufficient for the Q2K model which required the coefficient be adjusted to match observed depths, velocities, and time of travel (e.g., through calibration). Overall, an exponent of 0.45 for depth and 0.41 for velocity were determined with coefficients ranging from 0.1-0.2 and 0.05-0.15. Values are reasonable according to other studies (Barnwell et al., 1989; Flynn and Suplee, 2010; Leopold and Maddock, 1953) and are shown in (**Table 7-15**).

[23] These are determined in the field as part of the process of rating a gage site and provide information on mean velocity and hydraulic depth at a particular location in the river.

[24] Hydraulic depth was assumed to be the cross-sectional area divided by channel top width.

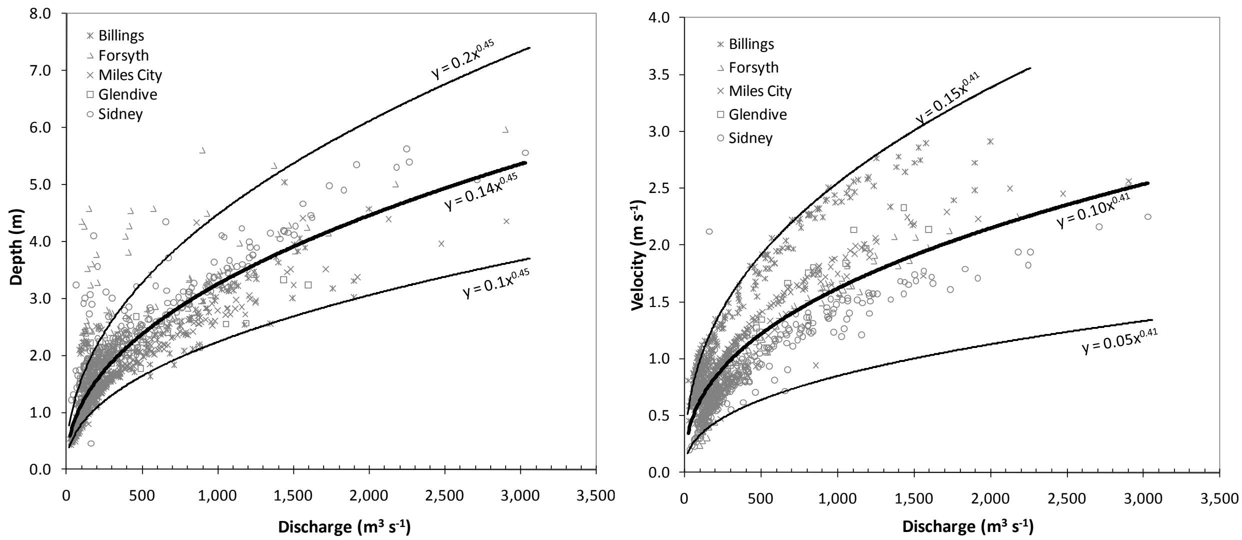


Figure 7-4. Depth and velocity rating curves derived for the lower Yellowstone River.
(Left panel). Depth vs. discharge. (Right panel) Velocity vs. discharge.

A final caveat about this effort is that in some instances the mean depth or velocity of a specific river segment will inevitably differ from what is determined through the use of the rating curve. This is a function of the idealized mathematical descriptions of channel hydraulic geometry, natural site variability, or un-described river mechanics. Thus the rating measurements are really estimates. Measurements themselves are variable according to field conditions and therefore represent the general behavior of the river rather than unique site conditions.

Table 7-15. Rating curve exponents derived for the Lower Yellowstone River.

Equation Form	Exponent	Typical Value[1]	Range[1]	Yellowstone River
Velocity $U = aQ^b$	b	0.43	0.4-0.6	0.41
Depth $H = \alpha Q^\beta$	β	0.45	0.3-0.5	0.45

[1]From Barnwell and Brown (Barnwell et al., 1989).

7.3.3 Dye Tracer Time of Travel Study

Time of travel estimates were made by USGS in 2008 as part of a cooperative study with DEQ (McCarthy, 2009). Seven unique reaches were considered for slug injections of dye and subsequent observation of the dye centroid as it passed through the points along the river. Those locations included: (1) Forsyth Bridge to the Cartersville Diversion Dam, (2) Cartersville Diversion Dam to Rosebud Bridge, (3) Rosebud Bridge to Fort Keogh Bridge, (4) Fort Keogh Bridge to Kinsey Bridge, (5) Kinsey Bridge to Calypso Bridge, (6) Calypso Bridge to Fallon Bridge, and (7) Fallon Bridge to Glendive Bridge. Results are shown in **Table 7-16** and the overall travel time for the river was 73.4 hours (3.1 days) from Forsyth to Glendive [at flows of approximately 200 $m^3 s^{-1}$ (7,000 $ft^3 s^{-1}$)].

Table 7-16. Travel-time data and mean streamflow velocities for the Yellowstone River in 2008.
Data from McCarthy (2009).

Site	Distance downstream from dye injection (mi)	Instant-aneous streamflow (ft3/s)	Elapsed traveltime after dye injection (hours)		Mean streamflow transport velocity of dye cloud for upstream reach (ft/s)	
			Peak	Centroid	Peak	Centroid
Slug injection of dye (21 liters) at 1700 hours on September 29, 2008 at Myers Bridge						
Myers Bridge	0.0	6,750	0.00	0.00	--	--
Forsyth Bridge	44.5	6,890	22.3	22.9	2.93[1]	2.85[1]
Forsyth Dam	45.6	6,890[2]	23.1	23.8	2.10	1.83
Rosebud Bridge	59.1	6,890[2]	30.0	31.0	2.83	2.75
Slug injection of dye (33 liters) at 1000 hours on September 26, 2008 at Forsyth Dam						
Forsyth Dam	0.0	6,860[2]	0.0	0.00	--	--
Rosebud Bridge	13.5	6,860[2]	5.90	6.32	3.36[1]	3.13[1]
1902 Bridge	51.5	7,320[2]	25.5	26.2	2.85	2.80
Kinsey Bridge	65.8	7,350[2]	32.2	33.1	3.14	3.07
Slug injection of dye (51.5 liters) at 1003 hours on September 23, 2008 at Miles City Bridge						
Miles City Bridge	0.0	7,420	0.00	0.00	--	--
Kinsey Bridge	11.8	7,470[2]	4.98	5.11	3.48[1]	3.39[1]
Calypso Bridge	38.8	7,570	16.7	17.6	3.39	3.18
Fallon Bridge	56.8	7,380	25.2	26.3	3.08	3.02
Glendive Bridge	89.1	7,480	42.4	43.6	2.76	2.74

[1]Mean streamflow transport velocity of dye cloud affected by incomplete lateral mixing of dye.
[2]Instantaneous streamflow estimated where discharge measurements could not be attained.

Flow conditions during 2008 were unfortunately very different to those encountered in 2007 (nearly double). Consequently, we relied on several methods to render the travel times in 2008 useful:

- Direct adjustment of the values calculated using McCarthy's (2006) Microsoft VBA travel-time calculator from which relates flood wave velocity to most probable baseflow velocity (using corrections obtained during 2008).
- Actual simulation of the 2008 flow condition and travel-time within Q2K.
- Adjustment of the dye study of 2008 (McCarthy, 2009) to 2007 conditions using interpretive hydraulics.

The latter is described in the next section. Results of all three methods are presented in **Section 10.0**.

7.3.4 Interpretive Hydraulics

An interpretive hydraulics analysis was completed as well to determine depth and velocity coefficients for individual hydraulic reaches in **Section 7.1** (thereby providing a better model parameterization). Under conditions of steady flow, Manning's equation (**Equation 7-6**) can be used to express the relationship between velocity and depth by assuming a wide rectangular channel approximation where V = velocity [m s^{-1}] , n = the Manning roughness coefficient, w = channel width [m], d = channel depth [m], S_f = bottom slope [m m^{-1}] and where "*wd*" is also equal to the cross-sectional area [m^2], and "*w+2d*" is the wetted perimeter [m].

(Equation 7-6)

$$V = \frac{1}{n}\left(\frac{wd}{w+2d}\right)^{2/3} S_f^{1/2}$$

The equation can be rearranged and simplified as shown in **Equation 7-7**, with substitution according to the continuity equation[25] thereby providing an equation with one unknown (depth) that can be solved iteratively provided the remaining variables are known.

(Equation 7-7)

$$0 = \frac{wd}{Qn}\left(\frac{wd}{w+2d}\right)^{2/3} S_f^{1/2} - 1$$

We identified the known values of **Equation 7-7** as shown in the bullets below. A M.S. Excel™ macro was then used to solve for depth simultaneously and complete the analysis for the river.

- **Width** – Relationships between discharge and wetted width were used to estimate river width during 2007 according to GIS data identified in **Section 7.1**. Three different photo/data series were considered: (1) 2001 color infrared photos, (2) 2004/2007 aerial photography, and (3) digitized interpreted bankfull dimensions from Applied Geomorpholgy/DTM Consulting (2004)[26].

- **Slope** – Channel gradient (e.g. friction slope) for each 100 m evaluation length was determined from the mosaiced 2.5 meter DEM of the lower Yellowstone River described in **Section 7.1**.

- **Flow** – Flows based on the water balance output identified in **Section 7.2**

- **Manning's "n" –** Roughness values as estimated using calibrated roughness values from recent flood insurance studies (L. Hamilton, personal communication, n=0.028) with additional adjustment for the flow condition being evaluated (Chow, 1959). Recall Manning's "*n*" varies with flow (**Figure 7-5**)[27] and was believed to be around 0.050 in August and 0.049 in September.

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[25] Continuity equation is as follows, $Q = wdV$, where *Q*, *w*, *d*, and *V* are defined in the text.

[26] Simple and consistent relationships were established between flow and wetted channel width at the time of imaging [log(w)=0.15 log (Q) + 1.867]. This lead to very minor adjustments of the original widths determined from the 2001 color infrared photos (corrections of -2 and +3 meters).

[27] Assuming no change in water surface slope over the range of flow conditions evaluated.

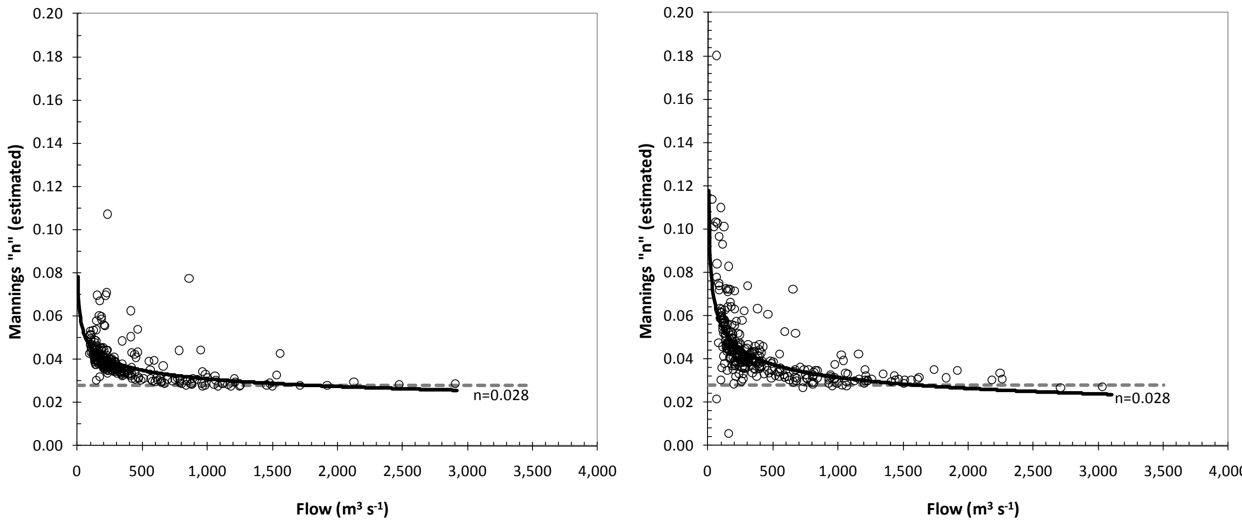


Figure 7-5. Estimated variation of Manning's n with flow for the Yellowstone River.
(Left panel) USGS 06309000 Yellowstone River at Miles City; (Right panel) 06329500 Yellowstone River near Sidney.

Results from the analysis are presented on the next page (**Figure 7-6**). Shown are: (1) the estimated values determined from the GIS analysis (width, depth, and velocity every 100 meters along the centerline of the channel), (2) values averaged over the hydraulic reaches identified in **Section 7.1** (note: these are already adjusted based on the 2008 to 2007 velocity correction), (3) velocities and depths determined from the 2008 dye tracer study, (4) the 2008 to 2007 dye tracer study correction[28], and (5) actual field data.

Computed wetted widths from this exercise ranged from approximately 175-350 m (575-1150 feet); depths were 0.3-2.9 m (1.0-9.5 feet); and velocities were 0.3-1.0 m s^{-1} (1.0-3.3 ft s^{-1}). All estimates reasonably reflect observed 2007 field observations and were used to translate rating coefficients to the model for each unique hydraulic reach. Values used in the model are found in **Appendix C** and range from 0.067-0.160 and 0.083-0.130 for depth and velocity respectively. As mentioned previously, they are within the ranges established in the literature (**Section 7.3.2**) yielding a travel time estimate of 4.1 days for August of 2007 (see **Section 10.2**).

[28] Adjusted dye velocities were determined from the rating curve in **Figure 7-4** where the difference in *Q* between 2007 and 2008 was used to determine the change in velocity (*ΔV)*. The adjustment was then applied to determine travel times during 2007 for which all hydraulic reaches were adjusted up or down so that the model matched the 2007 streamflow condition. This adjustment was made so that overall results of the Manning's equation representation (i.e., over the 100 meter lengths, and subsequent averages that comprise the hydraulic reach breaks) exactly matched the adjusted dye depths and velocities.

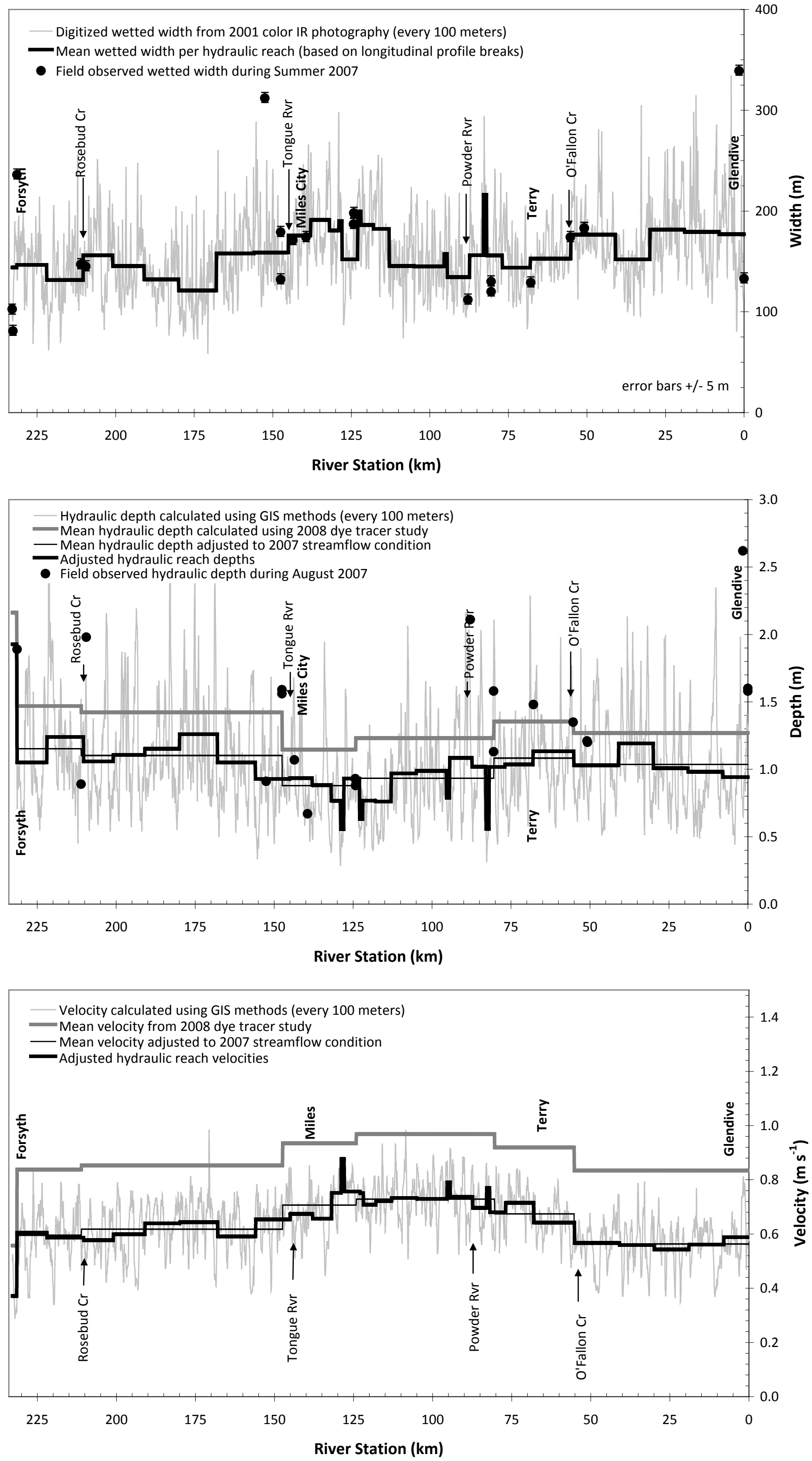


Figure 7-6. Estimated width, depth, and velocity over 1 km increments in the Yellowstone River.
Data shown for the August 2007 flow condition.

7.4 ATMOSPHERIC MODEL INPUT

7.4.1 Climatic Forcings

Required climatic input data for Q2K include air temperature [°C], dew point [°C], wind speed [m s^{-1}], solar radiation [cal cm^{-2}], and cloud cover [%]. Seven hourly climate stations were in operation in the lower Yellowstone River corridor during 2007. These were: (1) Forsyth W7PG-10 (AR184), (2) Sweeney Creek MT Department of Transportation (DOT) Road Weather Information System station (RWIS; MSWC), (3) National Weather Service (NWS) Miles City Municipal Airport (APT) station (COOP 245690), (4) DEQ Fort Keogh Agricultural Experiment station, (5) Bureau of Reclamation (BOR) Buffalo Rapids Terry AgriMet station (BRTM), (6) BOR Buffalo Rapids Glendive AgriMet (BRGM) station, and (7) NWS Glendive Community Airport (COOP 243581).

Information for the sites was retrieved via electronic download from the National Climatic Data Center (NCDC; www.ncdc.noaa.gov), MesoWest climate center (http://www.met.utah.edu/mesowest), and Bureau of Reclamation Great Plains AgriMet system (http://www.usbr.gov/gp/agrimet). Station attributes and climatic information for the August and September 2007 periods are shown in **Table 7-17.**

Table 7-17. Hourly climatic stations and associated mean daily observations.
Data shown for the average of the August and September analysis periods.

Station	Station ID	Station Elevation (m)	Elevation above River (m)	Mean Air Temp. (˚C)	Mean Dew point Temp. (˚C)	Mean[2] Wind Speed at 7m (m s^{-1})
Forsyth W7PG-10	AR184	887	120	Insufficient data		
Sweeney Cr (MDT)	MSWC	792	50	18.2	8.0	1.6
DEQ Ft. Keogh Ag. Exp.	DEQH	724	2	18.1	7.1	1.3
Miles City APT (NWS)	245690	803	90	18.2	5.5	3.9
AgriMET – Terry (Buffalo Rapids)	BRTM	692	30	16.8	6.1	2.9
AgriMet-Glendive (Buffalo Rapids)	BRGM	652	20	16.1	6.0	2.6
Glendive Community Airport (AWOS[1]; NWS)	726676	749	130	16.4	4.5	4.4

[1]AWOS = Automated Weather Observation Station.
[2]Wind speed adjusted to 7 meter height using the wind power-law profile[29].

From the data in **Table 7-17**, it is apparent that stations close to the river have different climatic conditions than those outside its influence (using elevation as a surrogate for proximity). This is best illustrated in comparison of the Miles City Municipal Airport site with DEQ's Yellowstone River station near Fort Keogh located on Roche Juan Island. The two sites were paired as part of the original project design (Suplee et al., 2006b) and show major differences in windspeed and dewpoint although being located just 2.5 km (1.5 miles) apart (note: the airport is on an elevated bluff adjacent to the river while the DEQ site was on a slightly vegetated island near the water surface).

[29] The power wind law profile equation is $\frac{v}{v_7} = \left(\frac{z}{z_7}\right)^k$ (Linsley et al., 1982), where *v* and v_7, and *z* and z_7 are velocity and measurement heights at their respective elevations above the ground (i.e., v_7 and z_7 at 7 meters). A $k={}^1/_7$ has traditionally provided acceptable results over a wide range of meteorological conditions (Linsley et al., 1982).now.

Wind magnitudes were substantially less in the river corridor than on the surrounding plateau. Differences in surface roughness (i.e., trees) and river corridor entrenchment are the primary causes. Sheltering and turbulent eddies result in both magnitude and directional shifts as observed in the inverse relationship between wind direction and river aspect (**Figure 7-7).** For dew point, values were much higher nearer the river than outside the river corridor which was expected due to the continuous source of evaporating water. Differences are consistent with Troxler and Thackston (1975) and Barthalow (1989) who suggest that river corridor effects cause considerable variability in climate.

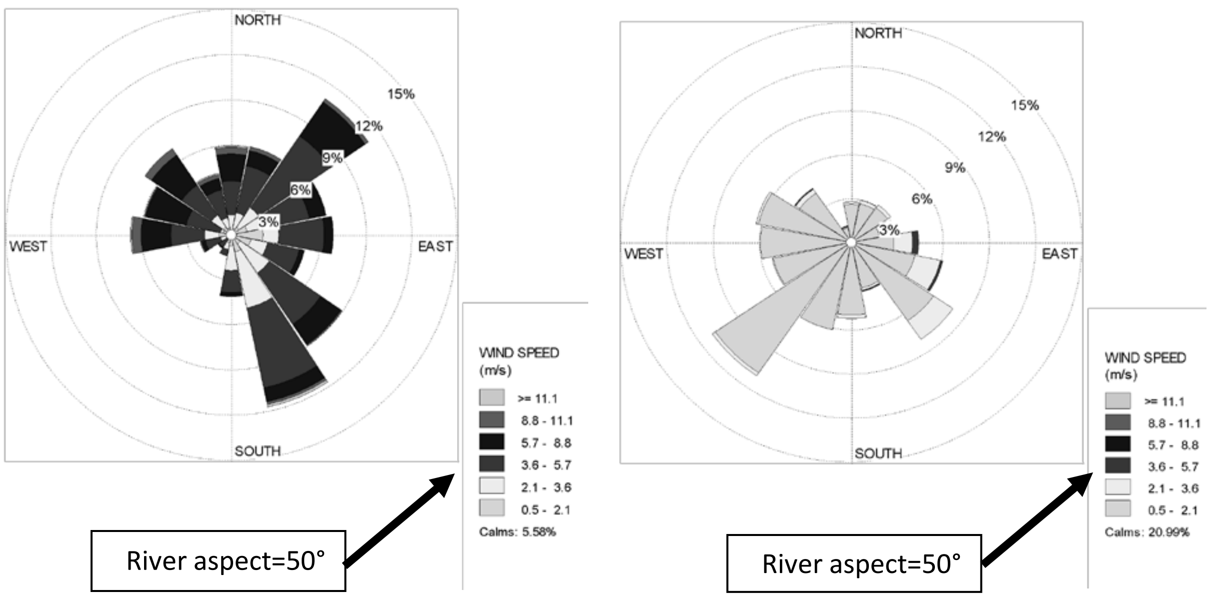


Figure 7-7. Paired wind rose data for the lower Yellowstone River during 2007.
(Left panel) Wind magnitude and direction at the Miles City Municipal Airport. (Right panel) Same, but for the DEQ station on Roche Juan Island. The reversal in direction and decline in magnitude from turbulence was used to justify wind speed correction factors for the model.

Given the prior knowledge, only climatological sites in close proximity to the river were used for model development. Those satisfying our requirements were assigned to spatially unique climatic zones in the model: (zone 1-Forsyth region) Sweeney Creek DOT station; (zone 2-Miles City region) Fort Keogh Agricultural Experiment Island station; (zone 3-Terry region) Buffalo Rapids Terry AgriMet station; and (zone 4-Glendive region) Buffalo Rapids Glendive AgriMet station. Time-series (air temperature, dewpoint, wind speed, and solar radiation) for these stations are shown in **Figure 7-8** for the model development period (i.e., calibration and validation). The shaded ribbon reflects the maximum and minimum of the four climatic zones.

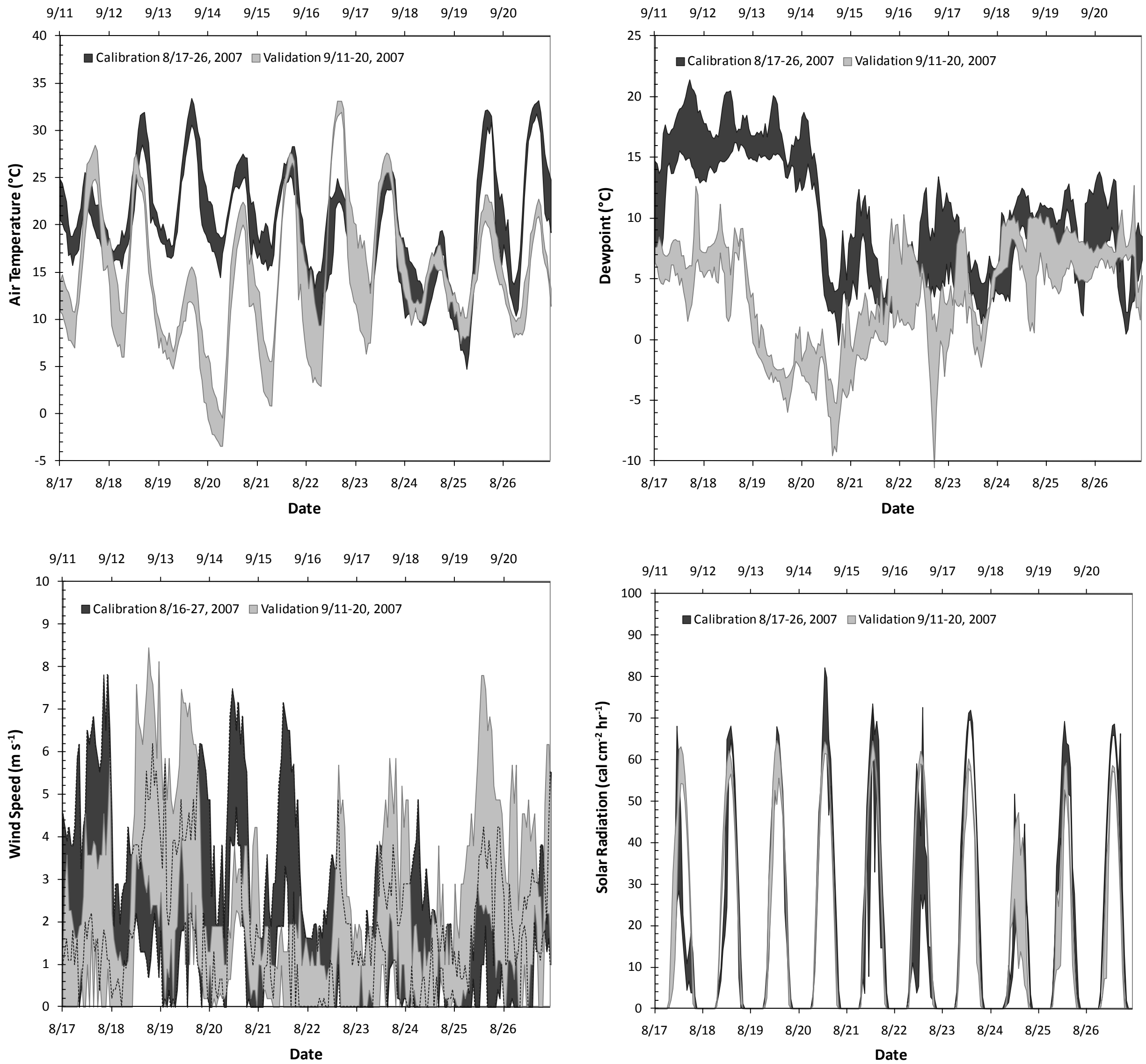


Figure 7-8. Hourly meteorological data summary for lower Yellowstone River in 2007.
(Top left/right panel) Air temperature and dew point for the four climatic zones in the lower Yellowstone River during the August 17-26 (calibration) and September 11-20, 2007 (validation) period (the shaded ribbon represents the min/max of the four climatic zones referenced in the previous paragraph). (Bottom left/right panel) Same but for wind speed and dew point.

In **Figure 7-8**, the biggest difference between the calibration and validation is air temperature and dew point (and to a lesser extent wind). This is related primarily to time of year and the difference between summer and fall conditions. What is not apparent from this figure is that there is a spatial climatic gradient. The upper portion of the river experiences warmer air temperatures, less wind, and higher humidity than the lower river. This is apparent in **Table 7-17** (shown previously).

The data from **Figure 7-8** was aggregated into mean repeating day hourly distributions (**Figure 7-9**) for the model (recall that Q2K operates on a repeating day simulation where every day in the model run has the same hourly conditions). Subsequently, observations at 6:00 a.m., 7:00 a.m., and so on were averaged over the analysis period (10 days) so that one day's weather pattern is repeated. In this instance, the daily distribution of data and diurnal differences between the calibration and validation time periods are more apparent. For example, August is warmer than September but both periods have similar patterns with air temperature reaching minimum at around 6:00 a.m. and peaking around 5:00 p.m. Dew point has an inverse relationship to temperature and again was much less in September. Winds were similar both periods and are calmest around daybreak and peak in the midday or early evening. Solar radiation and day length were slightly greater in August than September and sunrise and sunset occur at 6:00-7:00 a.m. and 8:00-9:00 p.m., respectively, with a solar radiation peak at around 1:00 p.m. (solar noon).

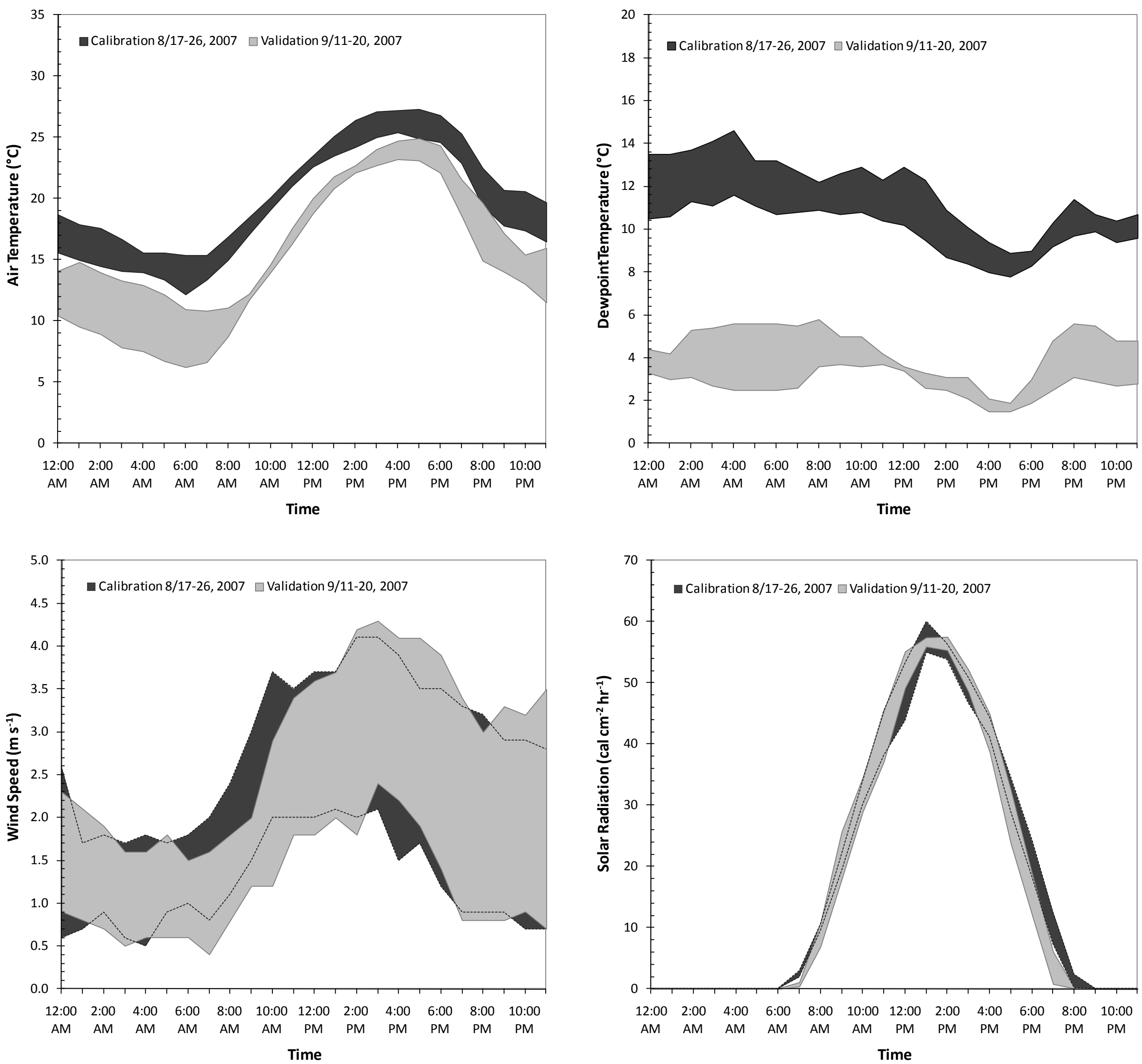


Figure 7-9. Mean repeating day climatic inputs for lower Yellowstone River Q2K model.
(Top left/right panel) Air temperature and dew point for the four climatic zones for the August 17-26 (calibration) and September 11-20, 2007 (validation) period. (Bottom left/right panel) Same but for wind speed and dew point. Data reflects a mean repeating day and the shaded ribbon represents the range of data for the climatic zones used in the model.

Of everything shown so far, all are direct input variables to Q2K except solar radiation. Instead, solar radiation is modeled by prescribing cloud cover, solar constant, cloud scattering coefficients, atmospheric transmission, and topographic/vegetative shade. To establish these values, observed radiation was used in conjunction with other field measurements.

Sky cover descriptions from the Miles City Municipal Airport and Glendive Airport and were translated to cloud cover percentages according to NOAA procedures (**Table 7-18, Figure 7-10**) and these estimates were used to evaluate solar radiation simulations from the model. It was found that the Bras solar model with atmospheric turbidity coefficient of 2.8 provided the most realistic estimate of incoming solar radiation for the August calibration period (**Figure 7-11**). Because atmospheric conditions were clearer in September (i.e., it was hazier in August according to field observations) a turbidity coefficient of 2.0 was used for the validation.

Table 7-18. Cloud cover classes and associated conversions (from NOAA).

Sky Cover Summation	Description	Translated Cloud Cover (%)
0: CLR	No coverage	0.00
1: FEW	2/8 or less coverage (not including zero)	0.13
2: SCATTERED	3/8 to 4/8 coverage	0.44
3: BROKEN	5/8 to 7/8 coverage	0.75
4: OVERCAST	8/8 coverage	1.00

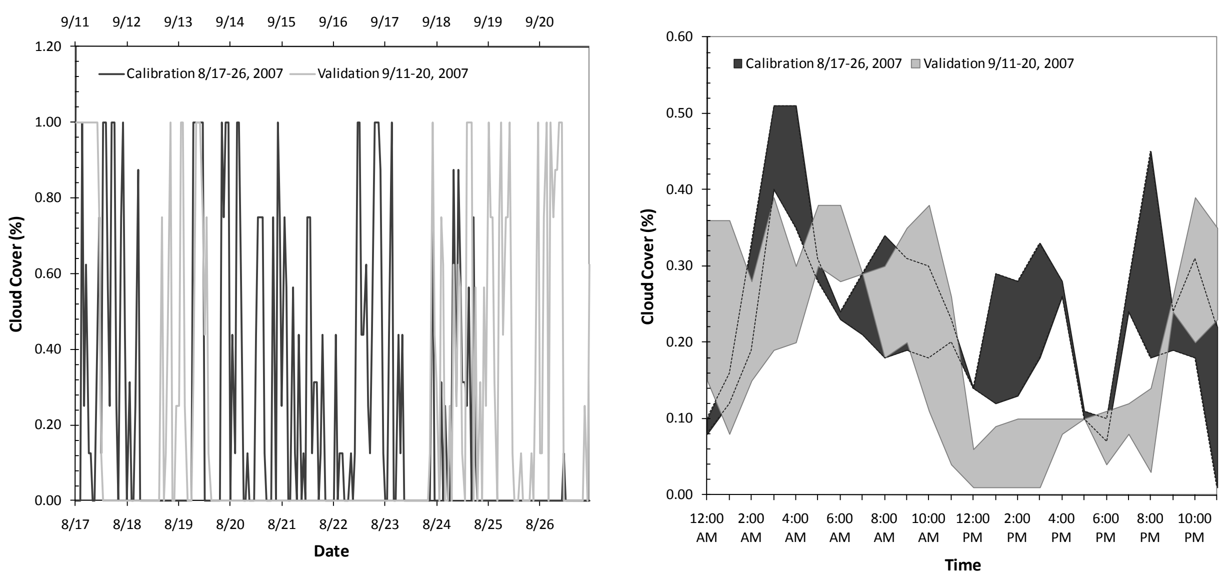


Figure 7-10. Hourly and repeating day cloud cover data for the lower Yellowstone River.
(Left panel) Cloud cover data for the 2007 period. (Right panel) Same but in a mean repeating day format. The range of the climatic stations used in the modeling reflects the shaded ribbon.

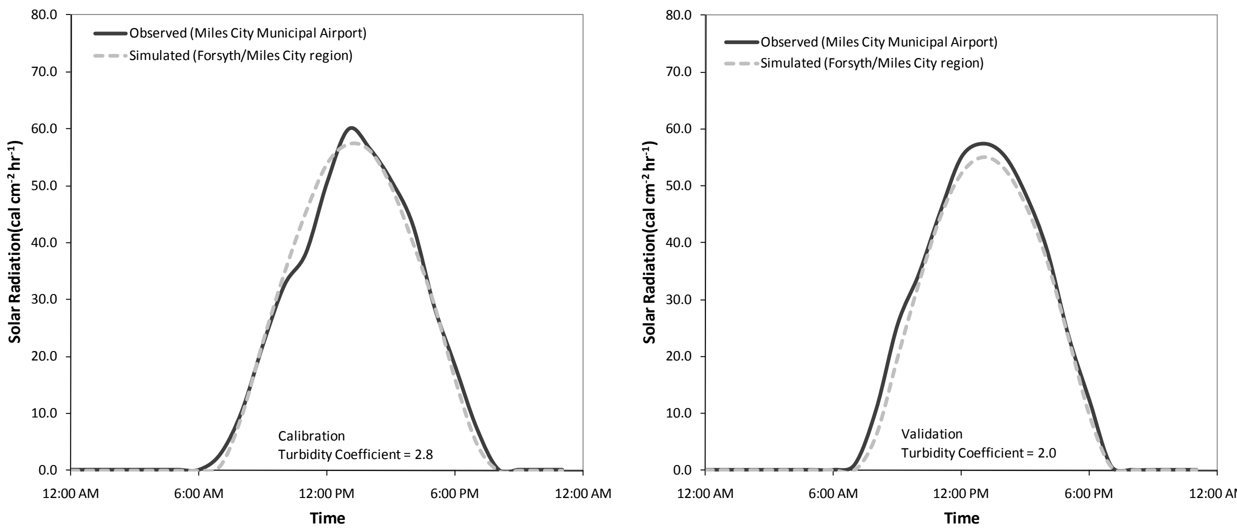


Figure 7-11. Simulated and observed solar radiation for the lower Yellowstone River.
(Left panel) Simulated and observed solar radiation for August 17-26, 2007 at Miles City Airport. (Right panel) Same but for September 11-20, 2007.

7.4.2 Carbon Dioxide and Aerial Deposition

Besides the climate data described previously, CO_2 concentrations and dry deposition rates of nutrients are also needed for the model. Such information is not readily available near the project site however. The closest observation stations were the Dahlen, ND GLOBALVIEW-CO2 monitoring site which is at the Fargo Jet Center (in eastern ND) and an EPA CASTNET site at the Theodore Roosevelt National Park-Painted Canyon in ND (THR422, NADP site ND00) (EPA, 2010a). Both locations are similar in climatically and topographically to the lower Yellowstone River and therefore provide good approximations.

Atmospheric carbon dioxide concentrations were determined every 8 days in 2007 (NOAA, 2010a). Observations during August and September were approximately 375 and 378 ppm. A historical chart showing concentrations from that site are shown in **Figure 7-12** (left). Dry deposition was estimated from the CASTNET site using concentrations of nitric acid, ammonium, and particulate nitrate in the weekly filter pack samples and deposition velocity from the Multi-Layer Model. Accordingly, nitrogen dry deposition levels averaged 0.71 and 0.66 kg N ha^{-1} yr^{-1} in August and September 2007 (**Table 7-19**) and have historically been consistent over time (**Figure 7-12,** right). Fluxes were applied to the channel surface area (m^2 converted to ha for a total of 3,084 ha of total river surface area) but were hardly worth considering as daily deposition was about 6 kg N per day (much less than even a single small tributary flow into the river).

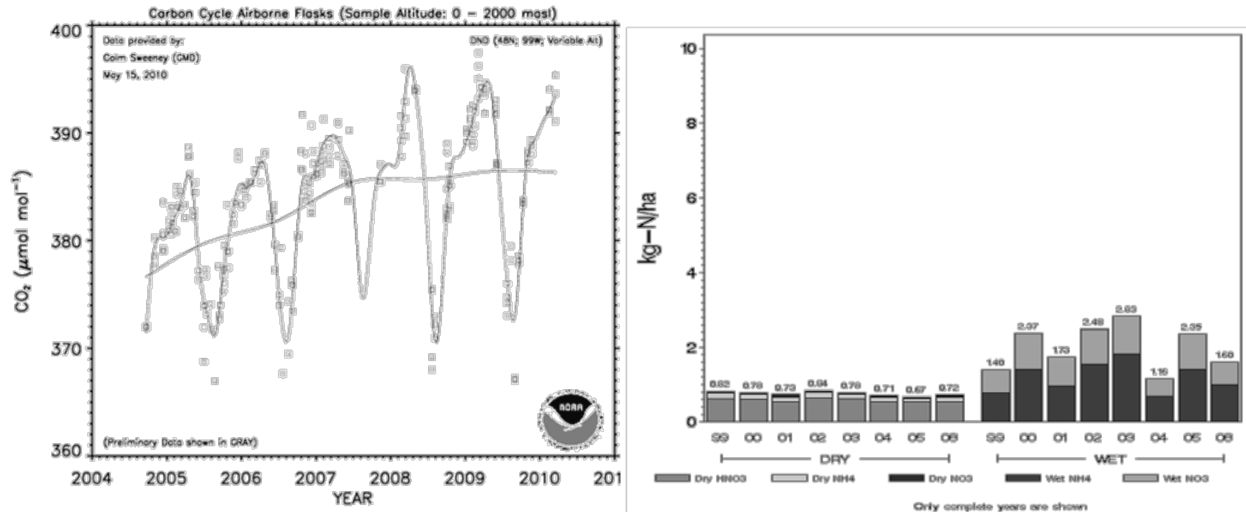


Figure 7-12. CO_2 data and nitrogen dry deposition by species for the Yellowstone River.
(Left panel) CO_2 data from the Dahlen, ND GLOBALVIEW-CO2 monitoring site (2004-2010). (Right panel) Nitrogen deposition data from Theodore Roosevelt, National Park EPA CASTNET site (1999-2008). Both figures taken directly from the data provider with permission.

Table 7-19. Dry deposition by nitrogen species estimated for lower Yellowstone River.
Data from Theodore Roosevelt National Park EPA CASTNET site for the August and September calibration and validation periods.

Species	Flux (kgN ha^{-1} yr^{-1})	Molar ratio (massN:mass)	Flux (kgN ha^{-1} yr^{-1})
Nitric Acid - HNO_3	2.73 2.29	0.222	0.61 (August) 0.51 (September)
Total ammonium - NH_4	0.09 0.16	0.777	0.07 (August) 0.12 (September)
Particulate nitrate - NO_3	0.09 0.10	0.292	0.03 (August) 0.03 (September)
Total N	-------	-------	**0.71 (August)** **0.66 (September)**

7.5 Shade Analysis

Shade is an optional requirement in Q2K and we did a simplified analysis to estimate its importance in the model. We applied the Shadev3.0.xls model which is a visual basic applications (VBA) software originally developed by the Oregon Department of Environmental Quality and modified by Washington Ecology to estimate shade as a function of aspect, channel width, vegetation canopy, bank elevations, near stream disturbance zone, and solar position (Pelletier, 2007). DEQ was unable to acquire all of the input for the model (e.g., vegetation characteristics and channel entrenchment) therefore we substituted data from other rivers in the state (tree height, density, etc.) to complete our estimates. Vegetation information came from the 2003 assessment of the river (NRCS, 2003) and was supplemented by the National Land Cover Dataset (Homer et al., 2004). The layers together were used to identify species in **Table 7-20**. Using a riparian zone sampling distance of 25 meters, the Chen method (which includes both topography and vegetation), and other assumptions in **Table 7-20**, daytime shade in August ranged from 0.3-13.3%, and averaged 2.5% over the project reach. Values for September were 0.3-15.8% and 2.9% respectively. Simulated shade is shown in **Figure 7-13**.

Table 7-20. Riparian landcover types and associated attributes used to estimate shade.

Vegetation Type[1]	Height (m)	Density (%)	Overhang (m)
Open Area or Primary Outwash	0.0	0%	0.0
Urban Areas	0.0	0%	0.0
Barren Land, Rock, Sand, Clay	0.0	0%	0.0
Deciduous Forest (sparse)	17.2	38%	0.1
Deciduous Forest (dense)	18.9	85%	0.3
Evergreen Forest	15.3	70%	0.0
Shrub, Scrub	1.0	50%	0.0
Grassland, Herbaceous	0.4	50%	0.0
Pasture, Hay	0.5	70%	0.0
Cultivated Crops	0.5	70%	0.0
Woody Wetlands (sparse)	4.9	40%	0.0
Woody Wetlands (dense)	5.7	75%	0.3
Emergent Herbaceous Wetlands	0.5	70%	0.0

[1]Data taken from Big Hole and Bitterroot Rivers. Channel incision was estimated to be 2.0 m throughout the project reach.

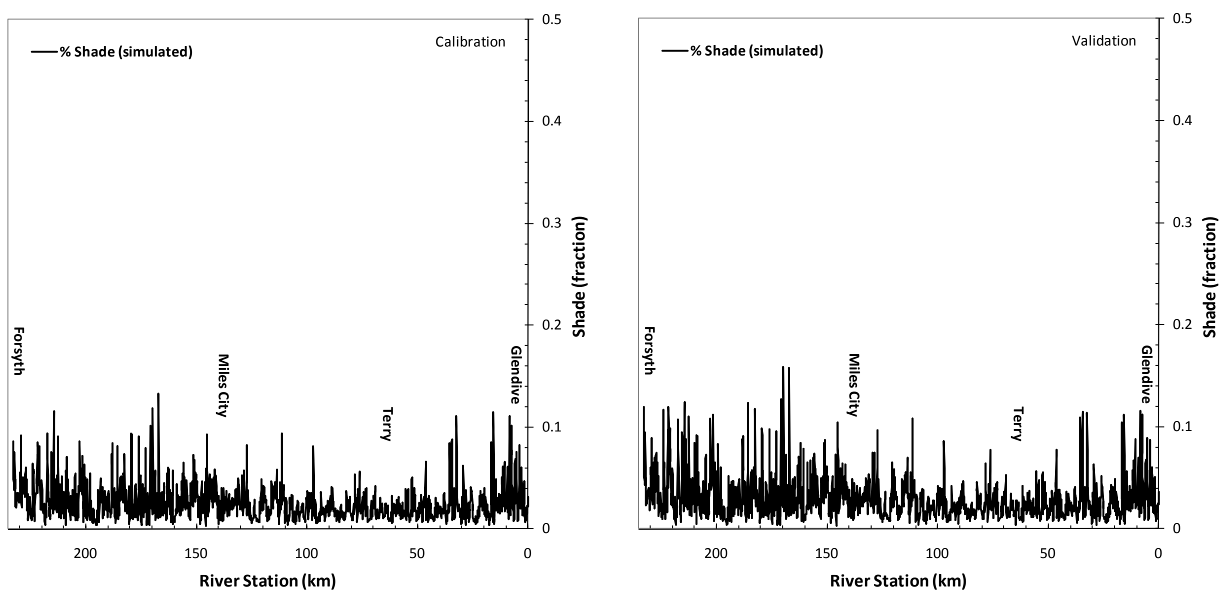


Figure 7-13. Simulated mean daily shade for the Yellowstone River.
(Left panel) Simulated shade for the August 17-26, 2007 period. (Right panel) Same but for September 11-20. No field data were available to verify the simulations. In both cases, shade is a minor component as indicated by mean daily shading of less than 20% throughout the river.

7.6 Boundary Condition Data

The final Q2K requirement is boundary condition data. This information was measured in the field to the extent possible, and several aspects of the data have been detailed previously [e.g., **Section 6.4** (headwater boundary conditions) and **Section 7.4** (air-water interface)]. The remaining uncharacterized components include constituent loadings from surface and groundwater which are described below.

7.6.1 Inflow Water Chemistry Data

A summary of the influent water chemistry to the Yellowstone River is shown as boxplots in **Figure 7-14**[30] (e.g., from WWTPs, tributary inflows, irrigation canal return flows, etc.). The maximum and minimum values, associated percentiles, and observed values during 2007 are identified.

[30] The database constructed in **Section 6.2** was used to provide the information for **Figure 7-14**. In several instances, values were scaled down by a factor of 10 (e.g., WWTPs) for plotting purposes (or were truncated). Refer to the comments in the figures in these instances.

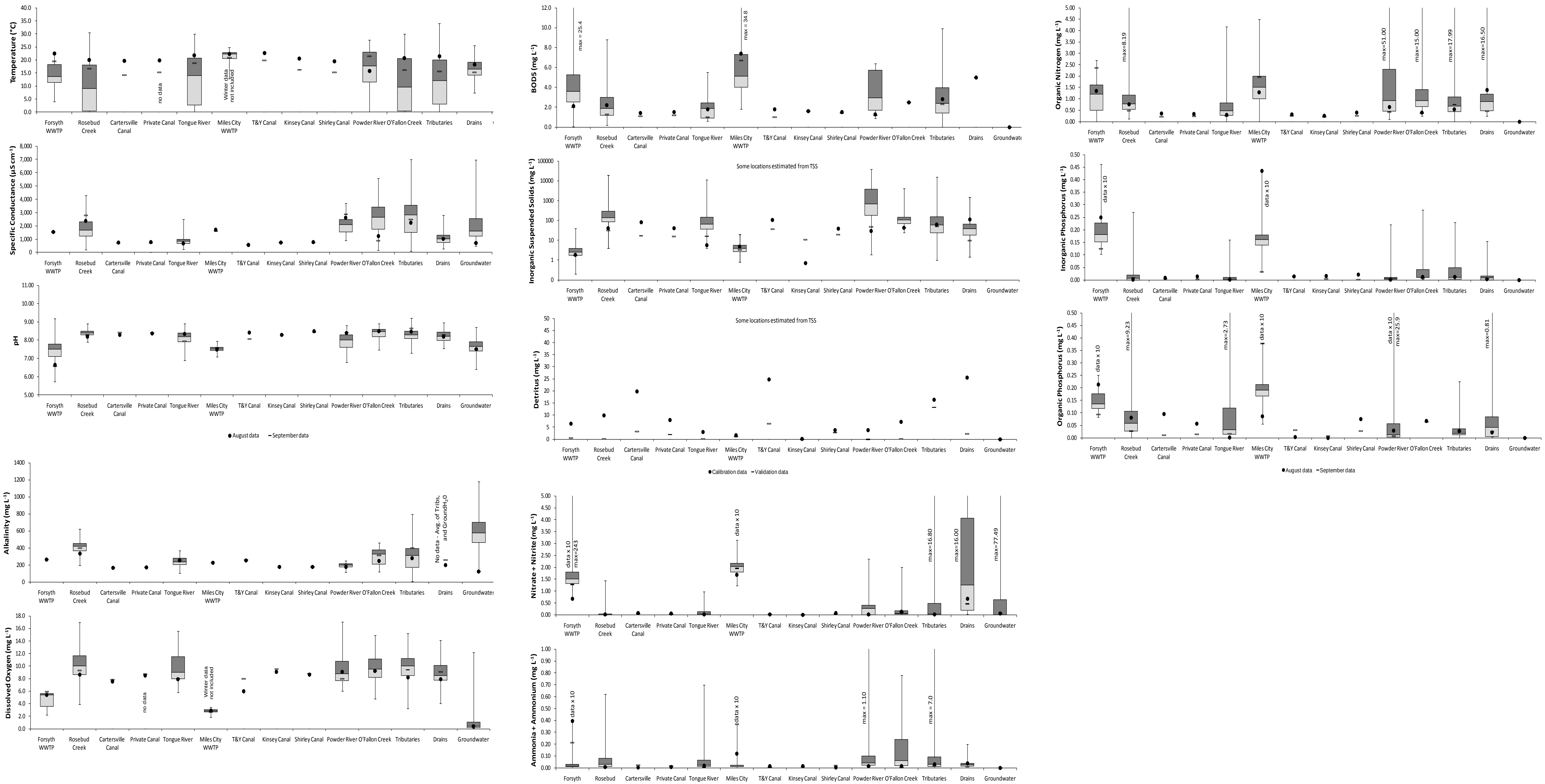


Figure 7-14. Comparative water quality inflow plots for the Yellowstone River.

In review of the prior plots, it is important to note that some data were not actually field-measured but were estimated. This is particularly true for unmeasured tributaries, waste drains, and groundwater. To assist the reader, methodologies to derive these concentrations are detailed below. Geometric means were used in all instances to reduce the right-skew bias.

Estimates for unmeasured tributaries were taken from the sites identified in **Table 7-10** (shown previously). Laboratory measurements were compiled together as shown in **Table 7-21**. Geometric means for August and September were used in the modeling

Table 7-21. Unmeasured tributary water quality data summary (1973-2007).

Monitoring Period[1]	**Temperature (°C)**	**pH**	**SC (µS cm^{-1})**	**DO (mg L^{-1})**	**TSS (mg L^{-1})**	**TN (mg L^{-1})**	**NO_2+NO_3 (mg L^{-1})**	**TP (mg L^{-1})**	**SRP (mg L^{-1})**
1973-2007 Max	34.0	9.20	7000	15.20	21,800	19.0	16.80	0.97	1.8
1973-2007 Min	0.0	7.13	101	3.21	1	0.12	ND[1]	ND[1]	ND[1]
1973-2007 Average	12.0	8.31	2699	9.77	980	1.57	0.64	0.23	0.066
August Geometric Average	**21.5**	**8.46**	**2229**	**8.18**	**61**	**0.90**	**0.02**	**0.04**	**0.013**
September Geometric Average	**15.52**	**8.65**	**2473**	**9.41**	**50**	**0.71**	**0.01**	**0.04**	**0.012**

[1]ND = no data.

Waste drains estimates again were made from previous investigations. The Buffalo Rapids Irrigation District routinely sampled for nutrients and field water quality from 1999-2002 (Schwarz, 2002) (*n*=129 samples). Similarly, Montana State University (MSU) (H. Sessoms, personal communication) made a detailed study of a subset of drains in the Clear Creek, Sand Creek, and Whoopup Creek drainages in 2007 (*n*=36 observations). Using this information, we estimated water quality constituent summaries for these types of features in the model network (**Table 7-22**).

Table 7-22. Irrigation waste-drain water quality data summary (1999-2007).

Monitoring Period[1]	**Temperature (°C)**	**pH**	**SC (µS cm^{-1})**	**DO (mg L^{-1})**	**TSS (mg L^{-1})**	**TN (mg L^{-1})**	**NO_2+NO_3 (mg L^{-1})**	**TP (mg L^{-1})**	**SRP (mg L^{-1})**
1999-2007 Max	25.51	8.96	2794	14.12	2082	14.25	16.0	0.97	ND[1]
1999-2007 Min	7.47	7.54	268	4.06	2	0.28	0.03	0.01	ND[1]
1999-2007 Average	16.55	8.21	949	8.49	47	3.61	0.74	0.03	ND[1]
August Geometric Average	**18.24**	**8.20**	**1007**	**7.90**	**159**	**0.89**	**0.67**	**0.03**	**ND[1]**
September Geometric Average	**15.22**	**8.30**	**1020**	**ND[1]**	**14**	**0.17**	**0.46**	**0.03**	**ND[1]**

[1]ND = no data.

Groundwater quality estimates were taken from a compilation of the Montana Groundwater Information Center (GWIC) database (MBMG, 2008). Data from the two drainage basins that overlap the study reach were used: (1) 10100001-Yellowstone River between Bighorn River and Powder River and (2) 10100004-Yellowstone River below Powder River. The search was constrained to wells that were less than 200 feet deep (Smith et al., 2000) and within 5 kilometers of the river. Estimates are shown in **Table 7-23**.

Table 7-23. Groundwater water quality data summary for the Yellowstone River.

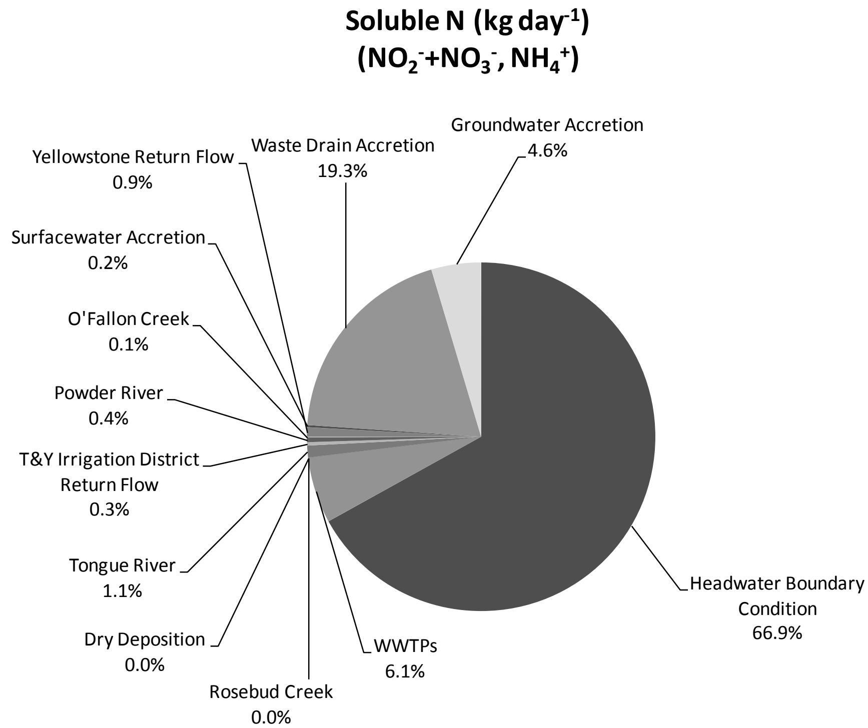
Monitoring Period	Temperature (°C)	pH	SC (µS cm^{-1})	DO (mg L^{-1})	Alkalinity (mg L^{-1})	NO_2+NO_3 (mg L^{-1})	TP (mg L^{-1})
1923-2008 Max	21.5	8.71	6970	12.19	1818	77.49	ND
1923-2008 Min	9.1	4.40	493	0.06	122	0.01	ND
1923-2008 Average	12.1	7.49	2121	1.14	609	2.89	ND
August Geometric Average	**11.7**	**7.50**	**1824**	**0.44**	**560**	**0.06**	**ND**
September Geometric Average	**11.7**	**7.50**	**1824**	**0.44**	**560**	**0.06**	**ND**

7.6.2 Nutrient Load Estimates to the River

Water quality (mg L^{-1}) and measured inflows (m^3 s^{-1}) detailed previously were used to make nutrient load estimates to the river. Loads were calculated for both soluble and organic forms, where soluble nutrients reflect the summation of nitrate+nitrate with aqueous ammonia (NO_2+NO_3, NH_4) while the only form of soluble P exists, soluble reactive phosphorus (SRP). The organic fraction reflects the summation of all nutrient species minus soluble nutrients, or those bound intracellularly within phytoplankton.

Estimated loads to the river for the 2007 period are shown in **Figure 7-15** and **16**. In review, the primary contribution of soluble N was the headwater boundary condition (66.9%) followed by irrigation waste drains[31] (19.3%) and WWTPs (6.1%). The primary contribution of soluble P was the headwater boundary condition (47.1%), WWTPs (36.2%), and irrigation return flow (4.3%). The greatest organic loads came from the headwater boundary (68.8-73.8%), and to a lesser extent the Powder River (19.3%) and irrigation waste-drain accretions (2.7-14.5%). Thus the headwater boundary condition is the major source of nutrients entering the project reach.

[31] In regard to the irrigation waste drains, these values are estimates only. During model calibration, it was identified that the contribution of N from waste-drains is likely over-estimated. This is a consequence of two things: (1) uncertainty in the flow estimates made by DEQ (recall that they were estimated using the relationship between irrigated area and return flow measured by MSU); and (2) uncertainty about the quality of water originating from these drains (the water quality estimates were made from data from 1999-2007 and were highly variable between sites). DEQ felt the most objective thing to do would be to include these estimates, but calibrate them down in the model.



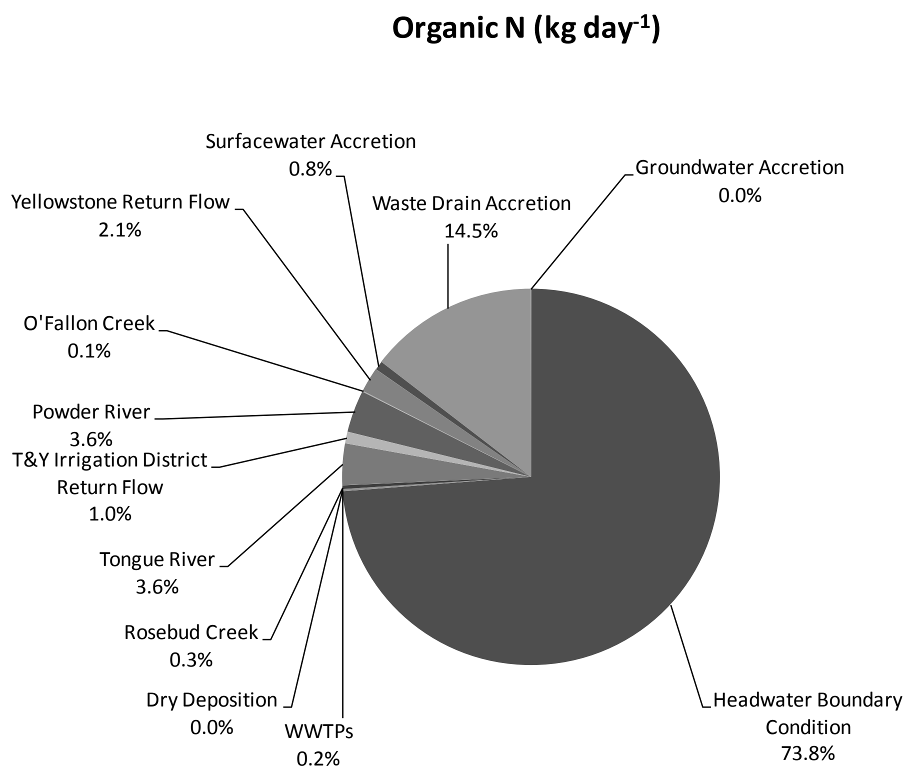
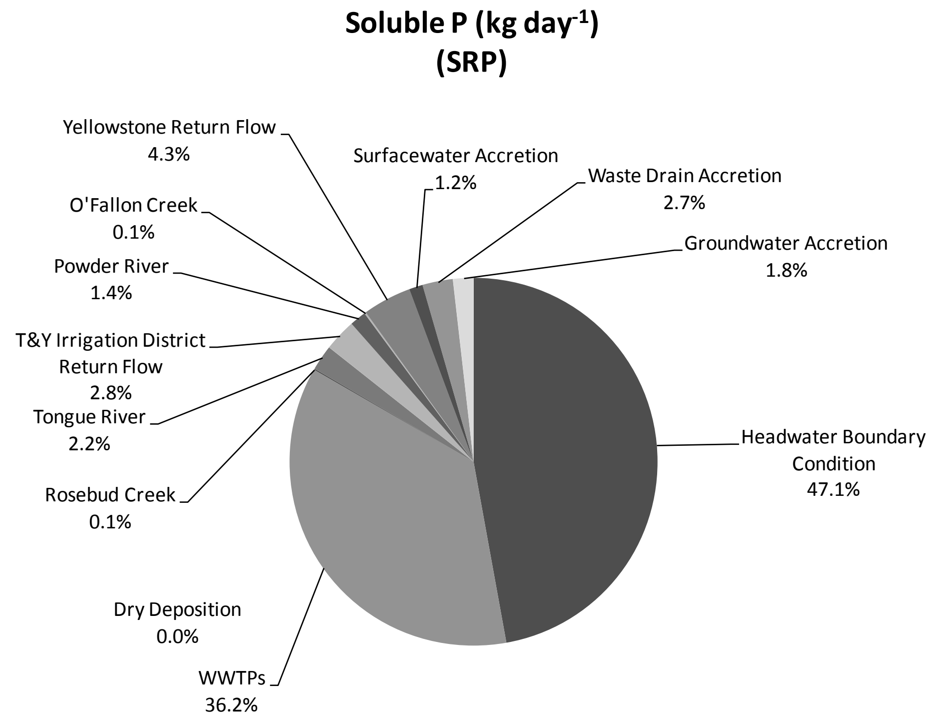


Figure 7-15. Estimated nitrogen contributions to the lower Yellowstone River during 2007.



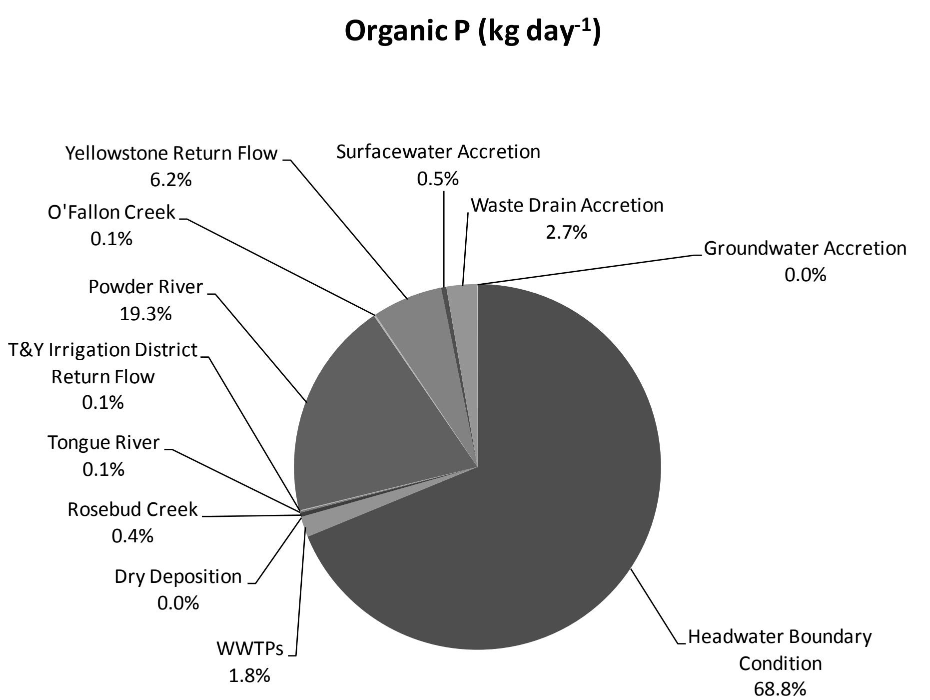


Figure 7-16. Estimated phosphorus contributions to the lower Yellowstone River during 2007.

7.7 DATA UNCERTAINTY

Clearly there is uncertainty in the estimates presented previously in this section. The extent depends on the type of measurement made, methodology, and in some cases, whether the value was measured at all (as opposed to an estimated value). We will address aspects of uncertainty in the Monte Carlo simulation described in **Section 14.0**. However, with regard to the data itself, work by Harmel et al., (2006) is perhaps useful. Probable errors of water quality monitoring field data (and associated instrument accuracy) are shown in **Table 7-24**. They represent a plausible range for which actual measurements may error and will be referenced later in the document.

Table 7-24. Probable error range in sample collection, storage, preservation, and analysis.

Measurement	Probable Error Range (±)	Source
Dissolved Oxygen	2% or 0.2 mg L^{-1}	YSI manual (2009)
pH	0.2 units	YSI manual (2009)
Temperature	0.15 °C	YSI manual (2009)
Conductivity	0.5% or 1 µS cm^{-1}	YSI manual (2009)
Chlorophyll-*a* - Phytoplankton	0.1% or 0.1 µg L^{-1}	YSI manual (2009)
Chlorophyll-*a* - Benthic algae	30%	DEQ (2011b)
Streamflow	10%	Harmel et al., (2006)[1]
TN	29%	Harmel et al., (2006)
NO_2+NO_3	17%	Harmel et al., (2006)
Ammonia	31%	Harmel et al., (2006)
TP	30%	Harmel et al., (2006)
SRP	23%	Harmel et al., (2006)
TSS/VSS/Detritus	18%	Harmel et al., (2006)

[1]Harmel et al., (2006) – Typical scenario average results.

8.0 SUPPORTING INFORMATION FOR THE CALIBRATION

A great deal of work went into model development. Supporting information for the calibration is found in this section. Included is a summary of sensitivity and rate coefficient estimates, and literature ranges expected for the model. These were used as an initial inference to guide calibration which was constrained by site-specific measurements (e.g., biomass, chemistry, water quality data, etc.).

8.1 SENSITIVITY ANALYSIS

A sensitivity analysis was completed to identify the most important (i.e., sensitive) model parameters [as recommended in the literature (Brown and Barnwell, 1987; Drolc and Koncan, 1999; Paschal and Mueller, 1991)]. We used QUAL2K-UNCAS (Tao, 2008) which is a re-write of the original QUAL2E-UNCAS (Brown and Barnwell, 1987). Parameter sensitivities were expressed as the normalized sensitivity coefficient (SC) (Brown and Barnwell, 1987) which reflects the ratio of change between model input and output (**Equation 8-1**),

(Equation 8-1)

$$SC = \frac{\Delta Y_o / Y_o}{\Delta X_I / X_I}$$

where ΔX_i = the change in the model input variable X_i and ΔY_o = change in the model output variable Y_o. Sensitivity was evaluated using a one-variable-at-a-time perturbation approach with an assigned magnitude of ±25%. Results are shown in **Table 8-1** for DO, pH and benthic algae and **Table 8-2** for TN and TP. Two locations of interest were evaluated, an element in the upper reach (km 150) and one in the lower (km 50). They reflect the different character of the river above and below the Powder River.

Boundary conditions yielded the highest sensitivities. This was expected as they directly influence mass flux in the modeled reach. However their influence subsides in the downstream direction (see Figure **8-1**). Indeed, parameter sensitivity becomes more important. Parameter sensitivities were interesting. With regard to DO and pH[32], stoichiometric parameters (STOCARB and STOCHLOR) were important which illustrate their significance on algal photosynthesis. Other sensitive rates included benthic algal subsistence quota (which is directly related to algal growth), CBOD oxidation rate (influences DO dynamics), and organic N hydrolysis rate (affects algal growth in soluble N deficient areas). Phytoplankton growth rate was also important due to its indirect influence on benthic algae.

For TN and TP, there were no rate coefficients of significance (<0.00). This is related to the fact that the rates do not change the total amount of nutrients in the system, only their form (i.e., organic or inorganic). The headwater boundary condition again was of importance (headwater TN and TP and phytoplankton internal N and P), as was point source load influent flow (for TP).

[32] This discussion focuses only on DO and pH, and later TN and TP. Many of the benthic algal rates directly influence the governing equation for algal mass balance and thus their significance relative to the other variables is misleading.

Table 8-1. Model sensitivities of the lower Yellowstone River Q2K model for DO, pH, and algae.
Evaluations completed at the ±25% level. The most sensitive parameters relative to DO, pH, and algae in bold.

Parameter[1]	Units	State-variable Sensitivity at km 150			State-variable Sensitivity at km 50		
		DO	pH	Benthic Algae	DO	pH	Benthic Algae
Rate Coefficients							
BALFACTP	Internal P half-sat. constant	0.02	0.01	0.25	0.00	0.00	0.02
BALG DET	Death rate	-0.06	-0.02	**-1.82**	-0.02	-0.01	**-1.84**
BALG GRO	Max Growth rate	0.04	0.02	0.50	0.01	0.01	0.50
BALG MAXN	Maximum uptake rate for N	0.00	0.00	0.01	0.02	0.01	0.85
BALG MAXP	Maximum uptake rate for P	0.07	0.02	0.90	-0.01	0.00	0.27
BALGQTAN	Subsistence quota for N	0.00	0.00	0.00	-0.04	**-0.03**	-1.01
BALGQTAP	Subsistence quota for P	**-0.08**	**-0.03**	-0.85	0.00	0.00	-0.50
BALLFACT	Light constant	-0.03	-0.01	-0.32	-0.01	-0.01	-0.42
BALNFACT	External N half-sat. constant	0.00	0.00	-0.01	-0.02	-0.01	-0.77
BALPFACT	External P half-sat. constant	-0.06	-0.02	-0.82	0.01	0.00	-0.07
FBODDECA	Fast CBOD oxidation rate	-0.05	-0.02	0.00	-0.04	**-0.03**	0.00
NH2 DECA	OrgN hydrolysis rate	-0.01	0.00	-0.01	**0.05**	**0.03**	**1.17**
PHYFACTN	Internal N half-sat. constant	0.00	0.00	-0.05	-0.02	-0.01	-0.72
PHYFACTP	Internal P half-sat. constant	-0.04	-0.01	-0.62	0.00	0.00	-0.02
PHYNFACT	External N half-sat. constant	0.00	0.00	0.03	0.02	0.01	0.75
PHYPFACT	External P half-sat. constant	0.04	0.01	0.55	0.00	0.00	0.02
PHYT GRO	Max Growth rate	0.03	0.01	-0.84	-0.02	0.00	**-2.86**
PHYT MAXN	Maximum uptake rate for N	0.00	0.00	-0.07	-0.03	-0.02	-0.98
PHYT MAXP	Maximum uptake rate for P	-0.04	-0.01	-0.65	0.00	0.00	-0.04
PHYTQTAN	Subsistence quota for N	-0.01	0.00	0.26	-0.01	-0.01	0.54
PORG HYD	Organic P hydrolysis rate	0.06	0.02	**0.95**	0.00	0.00	0.38
STOCARB	Carbon stoichiometry	**0.12**	**0.04**	0.00	**0.06**	**-0.05**	-0.01
STOCHLOR	Chlorophyll stoichiometry	**-0.12**	**-0.04**	0.12	**-0.06**	**0.05**	0.09
Boundary Conditions							
AIR_TEMP	Air temperature	**-0.08**	0.01	0.39	**-0.12**	0.00	0.25
HWTRALKA	Headwater alkalinity	0.00	0.01	0.00	0.00	**0.04**	0.01
HWTRBODF	Headwater CBODfast	-0.05	0.03	0.00	-0.03	0.03	0.01
HWTRDETR	Headwater detritus	-0.02	0.01	0.01	-0.01	0.01	0.07
HWTRDISP	Headwater dissolved P	0.03	0.01	0.35	-0.01	0.00	0.05
HWTRFLOW	Headwater flow	-0.01	0.02	0.28	0.00	0.01	0.26
HWTRFYTO	Headwater phytoplankton	-0.06	0.02	**1.19**	-0.02	0.02	1.13
HWTRNH2N	Headwater organic N	-0.01	0.00	0.01	0.04	0.02	0.88
HWTRNO3N	Headwater nitrate-N	0.01	0.00	0.04	0.01	0.01	0.21
HWTRPH	Headwater pH	0.00	**0.55**	0.00	-0.01	**0.40**	0.16
HWTRPINT	Headwater internal P	0.05	0.01	0.68	0.00	0.00	0.02
HWTRPORG	Headwater organic P	0.06	0.01	0.84	-0.01	0.00	0.04
HWTRTEMP	Headwater temperature	**-0.21**	0.00	0.62	**-0.08**	0.02	0.38
PH/PRESS	Partial pressure of CO_2	0.00	**0.04**	0.00	0.00	**0.04**	0.00
PTLDFLOW	Point load flow	0.00	0.00	0.06	-0.02	0.00	0.10

[1] BAL = benthic algae, PHYT = phytoplankton, PORG = organic phosphorus, FBOD = fast CBOD

Table 8-2. Model sensitivities of the lower Yellowstone River Q2K model for TN and TP.
Evaluations completed at the ±25% level. The most sensitive parameters relative to TN and TP in bold. All rate coefficients were insignificant.

Parameter[1]	Units	State-variable Sensitivity at km 150		State-variable Sensitivity at km 50	
		TN	TP	TN	TP
Rate Coefficients					
Insignificant		All sensitivities <0.00			
Boundary Conditions					
HWTRDISP	Headwater dissolved P	0.00	0.07	0.00	0.05
HWTRFLOW	Headwater flow	0.05	0.02	0.06	-0.16
HWTRNH2N	Headwater OrgN	**0.60**	0.00	**0.52**	0.00
HWTRNH3N	Headwater ammonia	0.02	0.00	0.02	0.00
HWTRNINT	Headwater internal N	**0.13**	0.00	**0.11**	0.00
HWTRNO3N	Headwater nitrate	**0.19**	0.00	**0.16**	0.00
HWTRPH	Headwater pH	-0.01	0.00	-0.02	0.00
HWTRPINT	Headwater internal P	0.00	**0.33**	0.00	**0.22**
HWTRPORG	Headwater OrgP	0.00	**0.52**	0.00	**0.36**
PTLDDISP	Point load dissolved P	0.00	0.01	0.00	0.03
PTLDFLOW	Point load flow	0.03	**0.08**	0.12	**0.25**
PTLDNH2N	Point load OrgN	0.01	0.00	0.05	0.00
PTLDNO3N	Point load nitrate	0.00	0.00	0.01	0.00
PTLDPINT	Point load internal P	0.00	0.01	0.00	0.01
PTLDPORG	Point load OrgP	0.00	0.03	0.00	0.11

Note: diffuse loads were insensitive and thus are not shown in the plots.

Longitudinal differences in sensitivity were also examined. These are shown in **Figure 8-1**. For both DO and pH, the upper river tends to be more sensitive to boundary conditions than the lower, with a declining importance in the downstream direction (with the exception of air temperature). In contrast, parameter (or rate coefficient) sensitivity increases in the downstream direction, with site-specific dependencies occurring due to river morphology, nutrient limitation, etc. It is concluded that initial condition error declines in the downstream direction whereas parameter sensitivity grows. The importance of this effect will be detailed further in the uncertainty analysis.

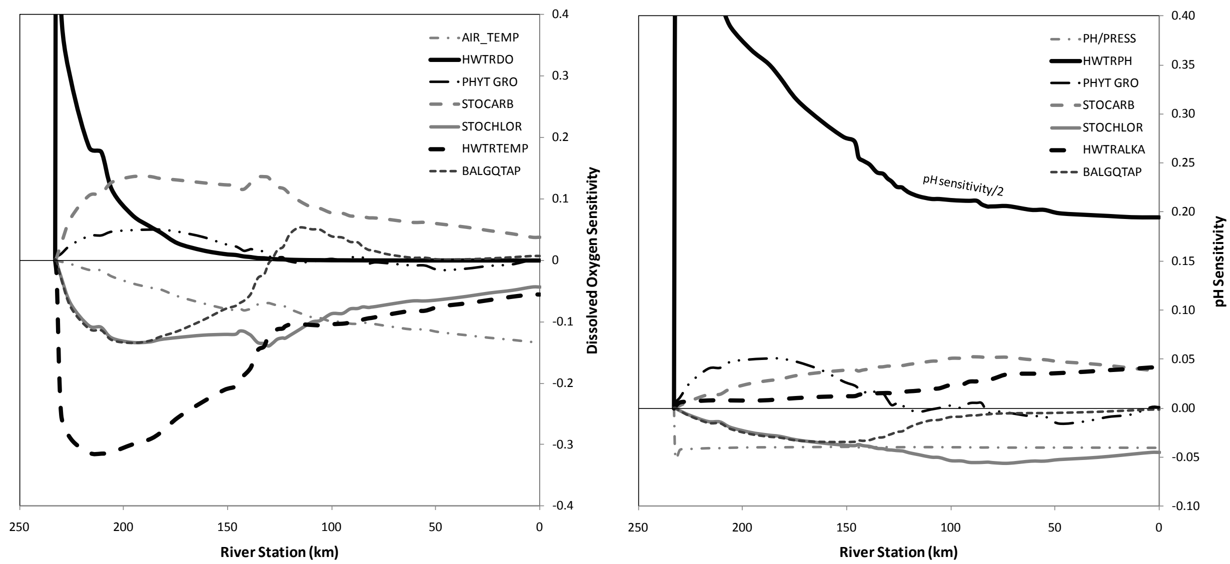


Figure 8-1. Longitudinal sensitivities of selected model rates and forcings for DO and pH.
(Left) Model sensitivities in relation to DO. (Right) Same but for pH.

8.2 Algal Taxonomy and Composition

Information on algal taxonomy was also acquired during 2007 to characterize species composition (i.e., diatoms versus filamentous algae), life cycle, mode of nutrient uptake (e.g., autotroph, heterotroph, nitrogen fixer, etc.), expected growth rates, and related information. The Yellowstone River has been well characterized in the past (Bahls, 1976b; Charles and Christie, 2011; Peterson and Porter, 2002), and through these efforts (and ours) we can make some general conclusions regarding the river.

First, algal assemblage differs longitudinally. In the upper regions of the river (i.e., from Billings upstream) benthic algae are the primary producers. Nuisance benthic algal accumulations have been observed numerous times in this vicinity, sometimes at concentrations greater than 800 mg Chl*a* m^{-2}. In contrast, phytoplankton are more abundant in the lower river (Peterson, 2009; Peterson and Porter, 2002) and tend to either dominate or co-dominate the river. The major shift between functional groups occurs below the Powder River marking transition between phytoplankton and benthic algal dominance.

Species composition is primarily in the division Bacillariophtya (diatoms), and *Cladophora* spp. (Bahls, 1976b). Diatoms dominate the net plankton of the river while filamentous *Cladophora* spp. and diatoms fairly evenly co-dominate the periphyton (Charles and Christie, 2011; PANS, 2008). Frequency of algal occurrence is shown in **Table 8-3** from net collections (Bahls, 1976b). From this, we conclude that little distinction can be made between the plankton and periphyton flora of the river. For example, suspended algae are primarily of benthic origin (scoured and resuspended by the current velocity of the river) and thus a solid understanding of benthic algae are required. In previous surveys, very few aquatic macrophytes were observed largely confirming that algae are the river's main primary producers. DEQ did not observe any macrophytes (i.e., vascular aquatic plants) during its work in 2006, 2007, or 2008.

Table 8-3. Percent frequency of algae taxa occurrence in the Yellowstone River.
From Bahls (1976).

Taxa[1]	Periphyton (% of taxa)	Plankton (% of taxa)
Bacillariophyceae (Diatoms)	44	56.6
Cladophora glomerata	47.5	29.2
Enteromorpha	3.6	0.9
Spirogyra	2.8	0.9
Hydrurus foetidus	0.7	1.8
Stigeoclonium spp.	0.7	0.9

[1]From 299 total periphyton and phytoplankton samples collected at 49 stations in the river.

8.3 Detached Drifting Filamentous Algae

Large amounts of detached and drifting filamentous algae were observed during 2007. These were mostly *Cladophora* spp. which were, according to field productivity experiments, still photosynthetically viable. To estimate the relative contribution of this detached drifting filamentous algae to areal benthic biomass, samples were collected with a fixed area screen. It was placed in the river perpendicular to flow for a known duration of time (area of screen was 0.3364 m^2, ≈4ft^2), and care was taken to not alter the oncoming velocity so that the approach was too fast that water would be shunted around the screen or so slow that algae wouldn't be in suspension. The experiment was halted before algae buildup on the screen occurred. Following the algal collection, velocity was measured at the center of the screen.

The net catch was then normalized to mg Chl*a* m^{-2} units using the screen area, accumulated or emigrated biomass, and the total water volume passing through the screen (by using the velocity, time, and screen area). In all instances (three different sites measured), the floating algae contribution was negligible. Measurements at the Highway 59 Bridge, Calypso Bridge, and Bell St. Bridge were all 0.02 mg Chl*a* m^{-2}. Consequently, detached, drifting filamentous algae was not considered in the modeling.

8.4 Stoichiometry of Algae

As shown in the sensitivity analysis, the stoichiometry of algae is an integral part of the carbon (C), nitrogen (N), phosphorus (P), and oxygen (O) mass balance. As algae die, hydrolytic bacteria quickly recycle nutrients into their respective pools at specified ratios and rates. In most modeling studies, the Redfield ratio (Redfield, 1958) is used due to a lack of site-specific data (Kannel et al., 2006; Turner et al., 2009). However for DEQ's purposes, site-specific estimates were preferable. Suspended seston samples were collected at a number of locations during both August and September 2007 to meet this need. Samples were analyzed for particulate C, N, P, ash free dry mass (D), and chlorophyll-*a* (Chl*a*).

Unfortunately raw river water contains both living and nonliving organic material. Hence detrital corrections were necessary to estimate the contribution from live algae. Corrections were made through linear regression of particulate organic C, N, P, and D (all in mg L^{-1}) with suspended Chl*a* (mg L^{-1}) where the ordinate of the best-fit line gives an estimate of the concentration not derived from phytoplankton (Hessen et al., 2003). This is shown in (**Equation 8-2**), where *x* = slope and *b* = y-intercept.

(Equation 8-2)

$$y = xChla + b$$

Estimates were made under the assumption that the slope of the regression line could be used to calculate individual ordinates for each *x*, *y* pair thereby providing a unique detrital estimate for each sampling site[33]. From this approach, detrital contributions for the Yellowstone River ranged from 35-57% in August and 63-85% in September. They averaged 47% for August and 73% for September (r^2=0.30-0.73, **Figure 8-2**), so there was more live algae in August (53%) than September (27%)[34].

Stoichiometry (by mass) for the river was therefore: 107 g AFDM: 43 gC: 4.7 gN: 1 g P: 0.8 gChl*a* for the August period, and 104 g AFDM: 42 gC: 4.5 gN: 1 gP: 0.8 gChl*a* for September. Values fall roughly into the range of C:N:P values reported in the literature for benthic algae, for example 61:8.1:1(Kahlert, 1998) and 46:7.7:1 (Hillebrand and Sommer, 1999). They are slightly lower than the Redfield ratio (40:7.2:1) (Redfield, 1958).

Several conclusions can be made from the stoichiometric estimates. First, the low N:P ratios (N:P<5.9 mass weight) and relatively high C:N ratios (C:N>8.6 mass weight) are suggestive of moderate nitrogen limitation in phytoplankton (Goldman et al., 1979; Hillebrand and Sommer, 1999). This interpretation is supported by the taxonomic findings of Peterson and Porter (2002) and Charles and Christie (2011) who note a large proportion of nitrogen fixers in the lower parts of the river (sometimes in excess of 30%). Conclusions by Klarich (1977) and Benke and Cushing (2005) suggest the same (that N is limiting).

In our case, conclusions are probably only valid for the floating algae. First off, analytical work was only for suspended algae and thus extending these estimates to benthic algae (i.e., those that are actually bottom-attached) may perhaps be a stretch. Second, soluble phosphorus levels in the river were very low in 2007 (2.6-3.7 ug L^{-1}), which suggests at least at some level of P limitation (Bothwell, 1985), or perhaps co-limitation. Additional discussions regarding nutrient limitation are found in the modeling results. Initial suggestions are provided only to give a general interpretation of the river.

[33] The ratio of ash-free dry mass (AFDM) and Chl*a* against other constituents (i.e., C, N, P) is the least reliable part of the detrital correction. For example, the C:N:P ratio remains fixed regardless of the detrital adjustment (i.e., it remains the same both before and after the correction), however the relation to Chl*a* and AFDM could vary. Therefore we feel that these ratios could be anywhere between the unadjusted and fully corrected ratios.

[34] The detrital contributions determined from this method were similar to those suggested by Bahls (1974) during non-productive conditions (e.g., comparing our September collections to his April analysis). During his study, he found 85-90% of the suspended seston in the river was unidentifiable pieces of organic detritus.

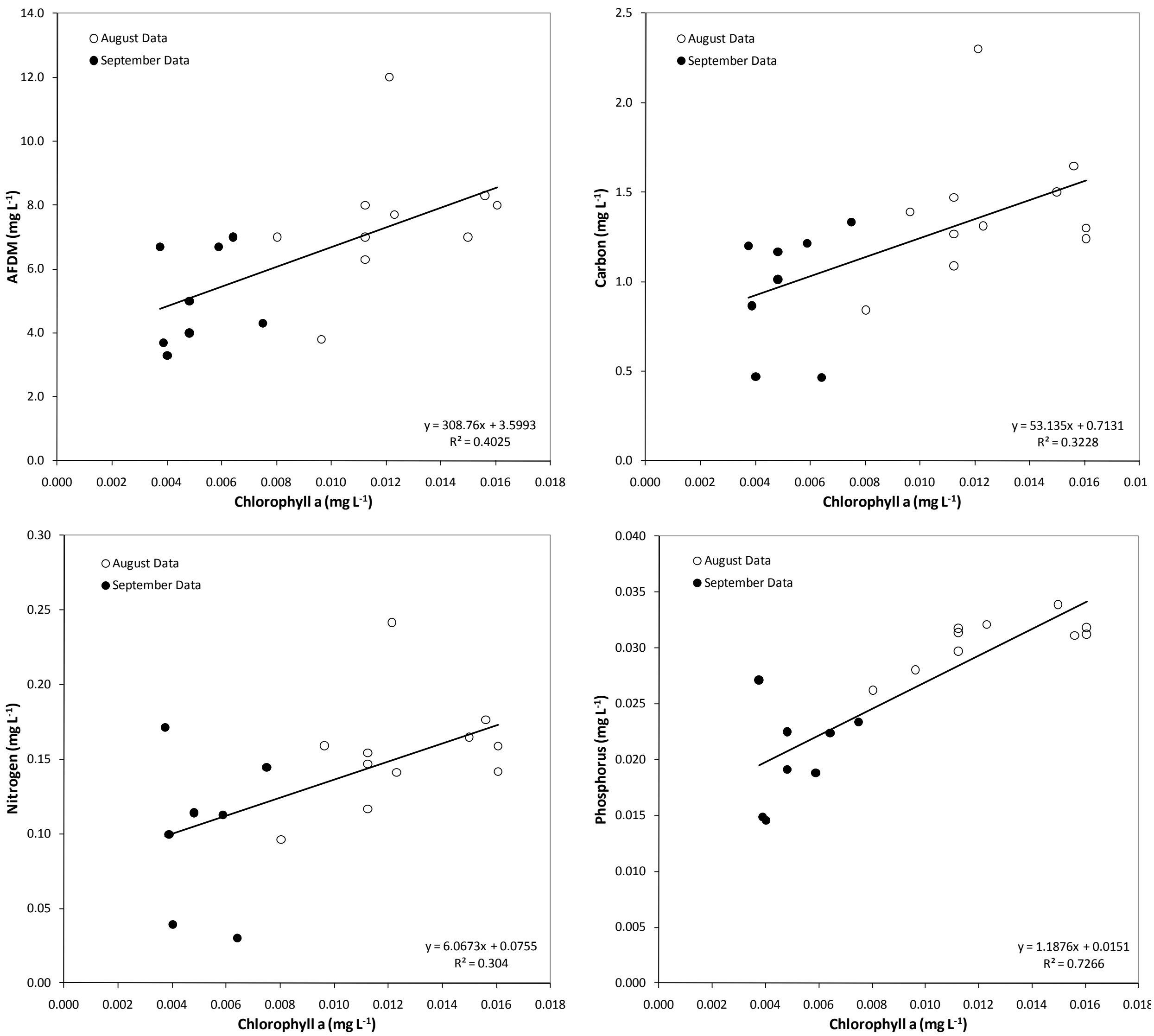


Figure 8-2. Stoichiometric C:N:P regression relationships for the Yellowstone River.

Note: One September data point was adjusted and one was removed due to inconsistent results. This was done as the carbon and AFDM values for this sample were anomalous compared to all of the other observations for the time period (e.g. ratio nearly double).

8.5 Algal Growth Rate Experiments (Light-Dark Bottles)

Field estimates of gross and net primary productivity and respiration were made in 2007 using light-dark bottles (Suplee et al., 2006b). Net specific growth rates were calculated according to Auer and Canale (1982) (**Equation 8-3**), where μ_{net} is the net specific growth rate (day^{-1}), P_{net} is the net photosynthetic rate (mg O_2 L^{-1} day^{-1}), C is the measured carbon content in the bottle (mg C L^{-1}), and P_q is the photosynthetic quotient. A photosynthetic quotient of 1.2 (i.e., 1 mole of C fixed per 1.2 mole of O_2 generated) was used (Wetzel and Likens, 1991). The gross specific growth rate is equal to the sum of the specific respiration rate and net specific growth rate.

(Equation 8-3)

$$\mu_{net} = \frac{P_{net}}{C} \bullet P_q$$

Gross specific growth rate measurements on the Yellowstone River were between 1.3-2.5 day^{-1} in August and 0.7-1.5 day^{-1} in September (**Figure 8-3**) which is within the expected range for phytoplankton (Chapra, 1997; Thomann and Mueller, 1987). Temperatures during these periods were 21°C and 16°C, respectively and adjustment to the standard temperature of 20°C [θ=1.07, from (Eppley, 1972)] yielded estimates ranging from 0.9-2.3 day$^{-1.}$ Consequently, the maximum unlimited growth rate in the model[35] was estimated to be 2.3 day^{-1}.

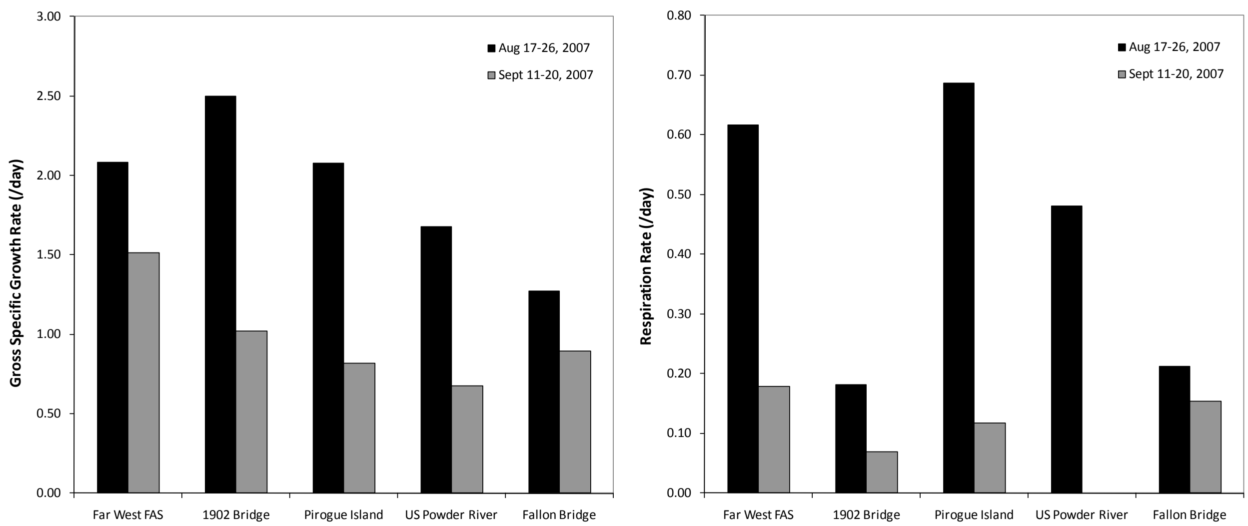


Figure 8-3. Primary productivity and respiration measurements on the Yellowstone River in 2007. There is a strong downstream decrease in productivity which is linked directly with river turbidity (and perhaps soluble nutrients to a lesser extent given the fact that nitrogen happened to be limiting phytoplankton). Respiration followed a similar trend, ranging from 0.2-0.7 day^{-1} in August and 0-0.2 day^{-1} in September (temperature corrected rates, 0-0.6 day^{-1}). Generally respiration rates were higher than expected (Chapra, 1997; Thomann and Mueller, 1987) and were believed to be at least partially due to the fact that they were not corrected for non-algal BOD (BOD decay rates in the river are believed to be on the order of 0.2 day^{-1}).

Rates of benthic algal growth could not be obtained but are believed to be lower than phytoplankton (Auer and Canale, 1982; Borchardt, 1996; Bothwell, 1985; Bothwell, 1988; Bothwell, 1989; Bothwell and Stockner, 1980; Horner et al., 1983; Tomlinson et al., 2010). Because of this, we used the literature to make an initial estimate of the maximum unlimited growth rate. According to Tomlinson et al., (2010), an upper limit of 1.5 day^{-1} is a reasonable estimate. Other literature suggests lower values could occur, but these are not always reflective of maximum unlimited growth conditions. Hence they are probably

[35] The maximum unlimited growth rate is the photosynthetic rate absent of any light or nutrient limitation (i.e., the fastest rate at which the algae could ever grow). Our field estimates are believed to be very close to the maximum unlimited growth rate for two reasons. First, the bottles were placed in ≈0.15 meters (0.5 feet) meters of water so they were absent of light limitation. Secondly, C:N:P measurements showed high internal P levels and associated concentrations of N were adequately high in the water column. Therefore nutrient limitation was not likely. Consequently, our field measurements seemed like a reasonable upper threshold for phytoplankton growth.

underestimates. Maximum unlimited rates identified by DEQ are shown in **Table 8-4** and show fairly consistent results when adjusted for temperature and photoperiod.

Table 8-4. Maximum unlimited first-order benthic algae growth rates from the literature.

Algae Type	Reported Growth Rate (day^{-1})	Temperature (˚C)	Adjusted to 24 hr lighting and 20˚C (day^{-1})[1]	Reference	Location
Diatoms	0.50	20	0.86	Klarich (1977)	Yellowstone River, MT
Diatoms	0.61	19.3	1.18	Bothwell and Stockner (1980)	McKenzie River, OR
Cladophora	1.08	19±2	1.08[2]	Auer and Canale (1982)	Lake Huron, MI
Green algae	0.76	17.5	0.89[2]	Horner et al. (1983)	Lab Flume
Diatoms	0.13	3.0	0.92	Bothwell (1985)	Thompson River, BC
Diatoms	0.54	17.9	1.14	Bothwell (1988)	S. Thompson River, BC
Diatoms	0.38	13.5	0.99	Biggs (1990)	South Brook, New Zealand
Diatoms	0.36	17	0.82	Stevenson (1990)	Wilson Creek, KY
Cladophora	1.53	19±2	1.53[2]	Tomlinson (1982)	Lake Huron, MI

[1]Data adjusted based on estimated photoperiod (range from 10-14 hours).
[2]No adjustment for lighting necessary (24 hr lighting used in experiment).

In summarizing this compilation, maximum unlimited growth rates range from 0.8 to 1.5 day^{-1}, with a mean of around 1.0 day^{-1}. Since most of these studies reflect the net specific growth rate (i.e., they are not corrected for the effects of respiration, death, or scour), estimates are likely low. Hence the initial estimate of 1.5 day^{-1} was believed to be a good starting point for calibration. This first-order maximum unlimited growth rate was then converted to a zero-order growth rate[36] to be consistent with the method used in the model. By approximating the slope of the exponential portion of the first-order growth model (**Figure 8-4**), 400 mg Chl*a* m^{-2} day^{-1} became our zero-order maximum unlimited growth rate estimate, similar to that identified by Turner, et al., (2009).

[36] First-order units were converted to zero-order units by approximating the exponential growth phase.

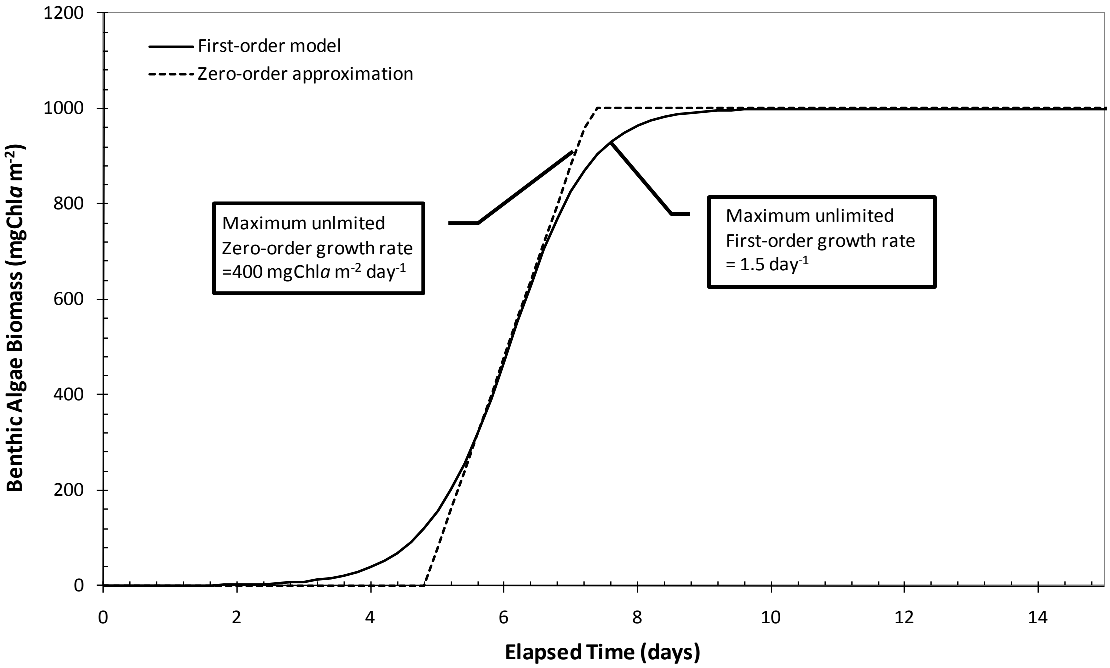


Figure 8-4. Comparison between zero- and first-order maximum unlimited growth estimates.
First-order growth modeled with an initial condition of 0.1 mg Chl*a* m^{-2} (**Equation 12-1**). The slowing of biomass accumulation over elapsed time is represented as a logistic function for implied space limitation (**Equation 12-2**, using 1,000 mg Chl*a* m^{-2} as the maximum biomass).

During this analysis, another consideration for the modeling was identified; the ability to reproduce maximum expected biomasses for the Yellowstone River. For diatoms (which were most abundant during 2007) maximum biomasses should be on the order of 300-400 mg Chl*a* m^2 (Stevenson et al., 1996). However, in the previous plot (which included no limitation terms) simulated equilibrium biomasses were nearer 1,000 mg Chl*a* m^{-2} (which was a user specified constraint otherwise biomass would grow infinitely). To verify that the model can indeed reflect the range of biomasses anticipated in the river, we did another algal growth simulation over time but with an assumed biomass loss (i.e., from respiration, death, scour, etc.)[37] of 50% based on Tomlinson et al., (2010) and Rutherford et al., (2000). This indicates that a maximum unlimited growth rate of 400 mg Chl*a* m^{-2} day^{-1} is a very appropriate value to reach the anticipated maximum biomass levels of 300-400 mg Chl*a* m^2 in the river for diatoms (**Figure 8-5**).

[37] In this instance we carried out the simulation using the zero-order algal growth model with assumed loss terms and incoming PAR at 100% (i.e., no light limitation).

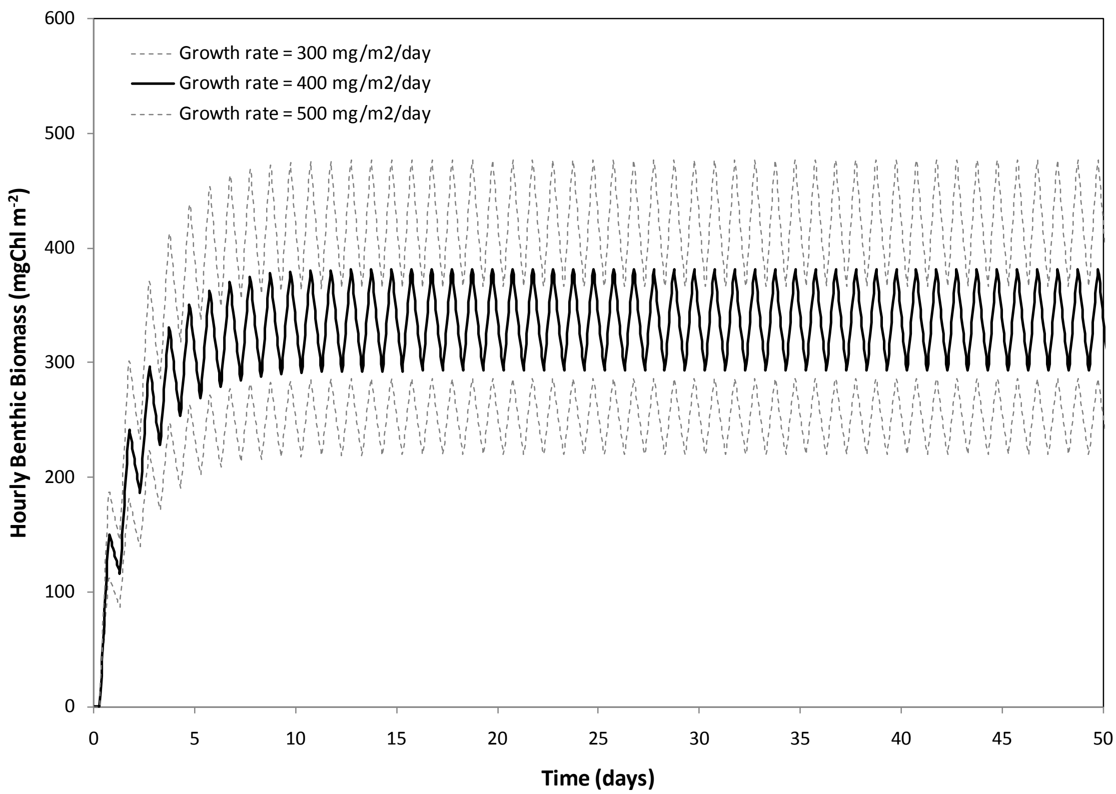


Figure 8-5. Estimated maximum biomasses with losses but no nutrient or light limitation.
Simulations show that a growth rate of approximately 400 mg Chl*a* m^{-2} day^{-1} is required to meet expected peak diatom biomass under unlimited growth conditions. The daily oscillations reflect the disparity between nighttime respiration and daytime photosynthesis (i.e., photosynthesis overcomes the respiration effect during the daytime and the opposite at night).

8.6 Minimum Cell Quota (Q_O) Estimates

Minimum cell quota (q_o) estimates were made and identify the minimum cellular concentration of N or P necessary for algal growth. According to Shuter (1978), q_o can be estimated for both N and P using cell biovolume (μm^3). From his regression analysis (data from more than 25 algal species), a log relationship exists between cell size and internal N and P concentration which suggests that larger algal cells have higher subsistence quotas and require more N and P than smaller ones. A very good correlation is observed across a wide range of alga species (r^2=0.9) and we used biovolume data collected by USGS during August of 2000 to make q_o estimates for the Yellowstone River.

For the broad spectrum of observations in the Yellowstone River during 2000 (i.e., the aggregate algal community), Shuter's (1978) regressions indicate that q_o should be on the order of 2.7 mgN mgChla^{-1} and 0.09 mgP mgChla^{-1}, with a range of 0.87-5.89 for N and 0.0-0.19 for P (according to the weighted average of cell sizes in the river during 2000 and the carbon to Chl*a* ratios found in 2007)[38]. The ratio of

[38] The conversion of units to mgA (Chl*a*) was completed with an assumed 43:0.4-0.8 ratio between carbon and chlorophyll.

q_o N and P values (30:1) is much larger than canonical Redfield (7.2:1 mass ratio), but is in agreement with Klausmeier et al., (2004) who indicate that the N:P ratio in autotrophic organisms shifts as they near the cell quota. For example, resource acquisition machinery (i.e., nutrient-uptake proteins and chloroplasts) are P-poor making the N:P ratio higher nearer the cell quota under nutrient deplete conditions (more like 20-30:1). In contrast, assembly machinery for exponential growth under optimal conditions (more like Redfield) is P-rich (ribosomes) and leads to lower N:P ratios.

It should be noted that the coefficients of variation (COV) from Shuter's work are quite low (COV=0.15) making most of the uncertainty associated with the C:Chl*a* ratio used in the analysis. Estimates are within the range reported by others (Reynolds, 1993; Shuter, 1978; Stevenson et al., 1996). For example, Reynolds (1993) suggests that q_o for N and P for phytoplankton should be on the order of 3.4-3.8 mgN mgChla^{-1} and 0.03-0.59 mgP mgChla^{-1} while Stevenson et al., (1996) indicate it should be 1.41-1.81 and 0.06-0.4 for N and P for benthic algae (although only one study was reported for N)[39]. Hence, we have a good initial estimate of the minimum cellular requirements of N and P in the river.

8.7 Algal Nutrient Uptake Estimates

Nutrient uptake estimates for algae were made solely through calibration. Since uptake is a function of both the internal and external nutrient concentrations, it is important to preserve the theoretical constructs during model calibration (Thomann, 1982). Calibration focused on uptake mechanics including assignment of maximum uptake rates, internal and external half-saturation coefficients, and observed data fits. Reviews of nutrient uptake kinetics can be found a number of places (Di Toro, 1980; Droop, 1973; Rhee, 1973; Rhee, 1978) and DEQ relied heavily on these constructs in model development.

Summarily, nutrient uptake depends on both internal and external nutrient concentrations where larger cells (or ones with higher growth rates) require more nutrients than those with smaller cells or lower growth rates. Counter intuitively, larger cells tend to have higher half-saturation constants than smaller cells and lower growth rates. Di Toro's (1980) work is particularly useful because he establishes constraints on parameter covariances of uptake factors. Suggested external half-saturation constants for phytoplankton range from 12-60 µg L^{-1} for P and around 4.2-42 µg L^{-1} for N, while internal half-saturation constants are an order of magnitude lower. A relationship between maximum unlimited uptake rate and maximum unlimited growth rate is also defined[40]. The dimensionless parameter β relates the maximum specific uptake rate to maximum growth rate suggesting the maximum possible variation in cell quota. Values for β are suggested to be on the order of 10 for N and 100 for P, reflecting a greater capacity of uptake for P as opposed to N. The internal half-saturation constant K_q can subsequently be estimated from q_o. We used a ratio of 1.0 for N and 0.5 for P which is recommended by Di Toro (1980) and others (Droop, 1973; Rhee, 1973; Rhee, 1978). Given the variability of these relationships, our estimates are at least a reasonable starting point in calibration.

[39] All literature conversions assumed a 100:1 ratio between sample AFDM and Chl*a* content (e.g., all original values were reported in N or P per unit dry weight).

[40] This ratio is defined as follows: $V_m = \beta(q_o \mu'_m)$, where V_m is the maximum unlimited uptake rate, β is the dimensionless ratio of V_m/q_o , and μ'_m is the maximum unlimited growth rate.

One last note regarding uptake kinetics should be made. We had the unexpected good fortune in 2007 of observing elevated nitrate at the headwater boundary condition at Forsyth. In this regard we got to observe longitudinal uptake/depletion which was a great benefit to model calibration. This condition was not present for P because at all times river P concentrations were very low, near or below the detection limit. Consequently, the literature was relied on heavily for P calibration which included calculated q_o from Shuter (1978) and internal and external half-saturation constants from Di Toro (1980).

8.8 REAERATION

Estimates of reaeration were made from the YSI sonde data using the procedures outlined in McBride and Chapra (2005). The approach is applicable to locations where the photoperiod is 10-14 hrs, primary production is well described by a half-sinusoid, and reaeration coefficients (k_a) are less than 10 day^{-1}. Other factors such as longitudinal gradients in stream temperature or water quality are assumed to be constant. The calculation for k_a is shown in **Equation 8-4** where η=the photoperiod correction factor[41] and φ=lag time between solar noon and the minimum DO (McBride and Chapra, 2005):

(Equation 8-4)

$$k_a = 7.5\left(\frac{5.3\eta - \phi}{\eta\phi}\right)^{0.85}$$

Measurements over a two week period at each datasonde location were used to make the reaeration estimate (i.e., the two weeks surrounding each sampling in August and September). The lag time between solar noon and the minimum dissolved oxygen deficit (i.e., the maximum DO concentration) was determined each day to calculate the temperature specific k_a. Photoperiod (f) and time of solar noon were taken directly from sunrise-sunset tables provided by the National Oceanic and Atmospheric Administration (NOAA, 2010b) using site latitude and longitude. The k_α was then calculated independently for a single day and results were averaged for the analysis period. Computed k_α values were adjusted to standard temperature (20˚C) and estimates for the August and September data collection episodes are shown in **Table 8-5**. A 95% confidence interval (CI) was also calculated from the mean of the observations.

[41] The photoperiod correction factor is defined as follows: $\eta = \left(\frac{f}{14}\right)^{0.75}$ (McBride and Chapra, 2005) where f=photoperiod and η is the photoperiod correction factor. The correction factor η was nearly at unity, as photoperiods for the Yellowstone River approximated 14 hours.

Table 8-5. Estimated reaeration coefficients for the Yellowstone River during 2007.

Sonde	Q2K Station (km)	August k_{a20} (day^{-1})	95% CI k_{a20}±	September k_{a20} (day^{-1})	95% CI k_{a20}±
10-Rosebud West FAS	232.9	2.4	1.24	2.5	1.20
20-US Cartersville Canal	184.3	2.6	0.94	2.7	0.86
30-1902 Bridge	147.5	3.4	0.60	5.1	0.93
35-RM 375	133.1	6.9	1.45	---	---
40-Kinsey Bridge FAS	124.2	3.9	1.71	5.9	1.57
50-US Powder River	87.9	3.1	0.76	4.6	1.46
60-Calypso Bridge	80.5	3.6	1.19	3.6	1.29
70-US O'Fallon Creek	55.3	2.9	1.17	5.1	1.48
80-Bell St. Bridge	0	1.9	0.94	2.3	0.80

8.9 Sediment Oxygen Demand (SOD)

Sediment oxygen demand (SOD) measurements were made in 2006 using sediment cores incubated at ambient river temperatures. SODs from 2006 are shown in **Table 8-6**. Duplicates ranged from 0.06-0.78 g O_2 m^{-2} d^{-1}, with an overall mean of 0.5 g O_2 m^{-2} d^{-1}. Our measured values are fairly typical for unpolluted rivers (Bowie et al., 1985; Edburg and Hofsten, 1973; Uchrin and Ahlert, 1985) and low relative to rivers with heavy pollution (Uchrin and Ahlert, 1985). Results suggest that the Yellowstone River has fairly low organic content and low SOD.

Table 8-6. SOD from the Yellowstone River measured via core incubations.

Location	SOD (g O^2 m^{-2} d^{-1})	Mean (g O^2 m^{-2} d^{-1})	COV (%)	ApproximateRange	
				Min	Max
Roche Jaune FAS	0.66	0.51	43		
Duplicate	0.35				
Fallon Bridge	0.78	0.67	22	0.2[1]	1.0
Duplicate	0.57			0.1[2]	10
Richland Park	0.43	0.24	105		
Duplicate	0.06				

[1] For sand bottoms (Bowie et al., 1985).
[2] Approximate range (Bowie et al., 1985).

SOD measurements were also attempted during 2007 using *in situ* benthic SOD chambers after the design of Hickey (1988). Unfortunately, we were unable to derive any useful data from the chambers because we could not get a good seal between the chambers and the river bottom (due to the coarse nature of the Yellowstone River's gravel/cobble substrate). DO levels increased inside the darkened chambers, even after they were in place during the morning-long DO increase in the river from algal photosynthesis. Thus water from outside the chamber was evidently leaking in. Consequently, we were only able to use the 2006 sediment core SODs as our field-measured values for the modeling work.

To go along with the SOD measurements, percent SOD coverage was visually estimated in the field to provide areal estimates for the model. We observed sediment at 11 locations within each sampling transect and used particle size (i.e., fine grained) as a surrogate for SOD generating material. In all cases, <5% of the channel substrate would qualify as SOD responsive (**Table 8-7**). Admittedly, our *n* was small, but observations did generally fit our conceptual understanding of the river, i.e., a well-armored cobble/gravel bed with high flow velocities devoid of organics or other SOD generating material.

With regard to algal cover percentage we used field observations. Admittedly, the water was too deep to make a visual assessment in several instances (noted as not visible on the field form), but the presence of Chl*a* was verified analytically at nearly all transect sites (even on sands/clays). Lastly, the average of sites was used in the modeling. Given the river variability, values were rounded to the nearest five percent, at 5% for SOD and 90% for benthic algae coverage.

Table 8-7. SOD and algal coverage estimates for Yellowstone River.

Location	Mean substrate Size (mm)	Class	Estimated cover by SOD (%)	Estimated cover by algae (%)
Far West FAS	59	gravel	0	90
1902 Bridge	38	gravel	5	90
Pirogue Island	53	gravel	0	100
Kinsey Bridge FAS	84	cobble	0	80
Fallon Bridge	49	gravel	5	100
Averages	56	gravel	5	90

8.10 Light Extinction and Suspended Particles

Light extinction and the influence of suspended particles were also evaluated using the Beer-Lambert law (**Equation 3-1**). The primary variable of interest was the extinction coefficient (k_e), which reflects the collective absorption and scattering of particles in the water column. Chemistry and PAR data from Peterson (2009) were used to identify k_e through rearrangement of **Equation 3-1**[42], where k_e is the slope of the best fit line, z is water depth, and $PAR_{surface}$ is the y-ordinate. Fitted extinction coefficients are shown in **Figure 8-6** and were found to range between 1.3-2.5 m^{-1} (r^2=0.85-0.99). They generally increase in the downstream direction.

Net k_e can also be approximated linearly as the sum of several partial extinction coefficients reliant on the concentrations of particles in suspension and their optical attributes (Blom et al., 1994; Di Toro, 1978; Van Duin et al., 2001). **Equation 8-5** illustrates this where k_{eb} reflects the extinction due to colloidal color and water (m^{-1}), α_i, α_o, α_p, and α_{pn} are unique to the suspended particle type ($m^2\ g^{-1}$), and m_i, m_o, and a_p are the concentrations of inorganic suspended solids (m_i, mg L^{-1}), detritus (m_o, mg L^{-1}), and phytoplankton (a_p, µg L^{-1}) respectively.

(Equation 8-5)

$$k_e = k_{eb} + \alpha_i m_i + \alpha_o m_o + \alpha_p a_p + \alpha_{pn} a_p^{2/3}$$

[42] $\ln(PAR_z) = -k_e z + \ln(PAR_{surface})$

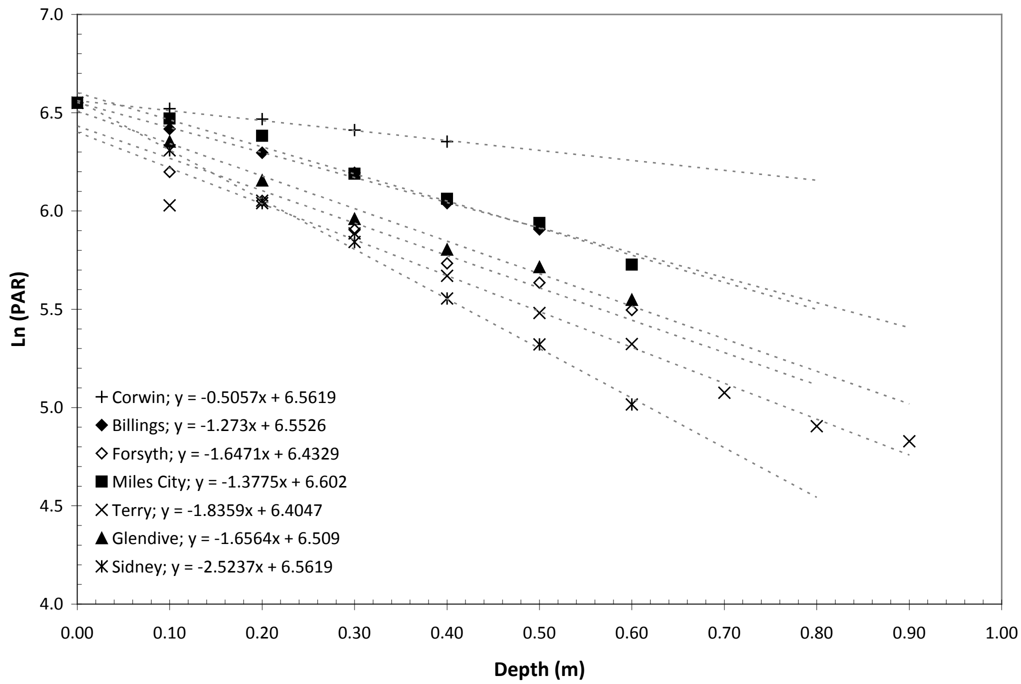


Figure 8-6. Light extinction coefficients calculated for the Yellowstone River.
Extinction values range from 1.3-2.5 m^{-1} in the lower river (Billings to Sidney). Measurements taken in late August or early September by Peterson (2009).

Partial extinction coefficients were determined according to **Equation 8-5**, where the effect of water and color (k_{eb})[43] was first determined using the sum of the partial extinction coefficient of pure water (k_w) and color (k_{color}). The value for k_w was assumed to be that of pure water 0.0384 m^{-1} (Lorenzen, 1972; McPherson and Miller, 1994; Phlips et al., 2000) and the partial attenuation coefficient for color (k_{color})[44] was calculated using the relationship of 0.014 m^{-1} per platinum-cobalt unit (Pt-Co, a measure of color) (McPherson and Miller, 1994; Phlips et al., 2000). Historical color measurements on the river were used to define the overall color effect in the model. Based on an *n*=5 and *n*=11 for the two gages of interest the estimated true color under low flow conditions is 5.38 Pt-Co units or a k_{eb} estimate[45] of 0.114 m^{-1}. Tabulated historical color measurements for the river are shown in **Figure 8-7**. Measurements were consistent (standard deviation=1.4 Pt-Co units) for the most part.

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[43] $k_{eb}=k_w + k_{color}$
[44] A water's color changes based on dissolved aquatic humus, i.e., gilvin or yellow substance, see (Davies-Colley, 1992; Kirk, 1994).
[45] Calculation is as follows: 5.38 mg L^{-1} Pt-Co x 0.014 m^{-1} L mg^{-1} (color) + 0.0384m^{-1} (water).

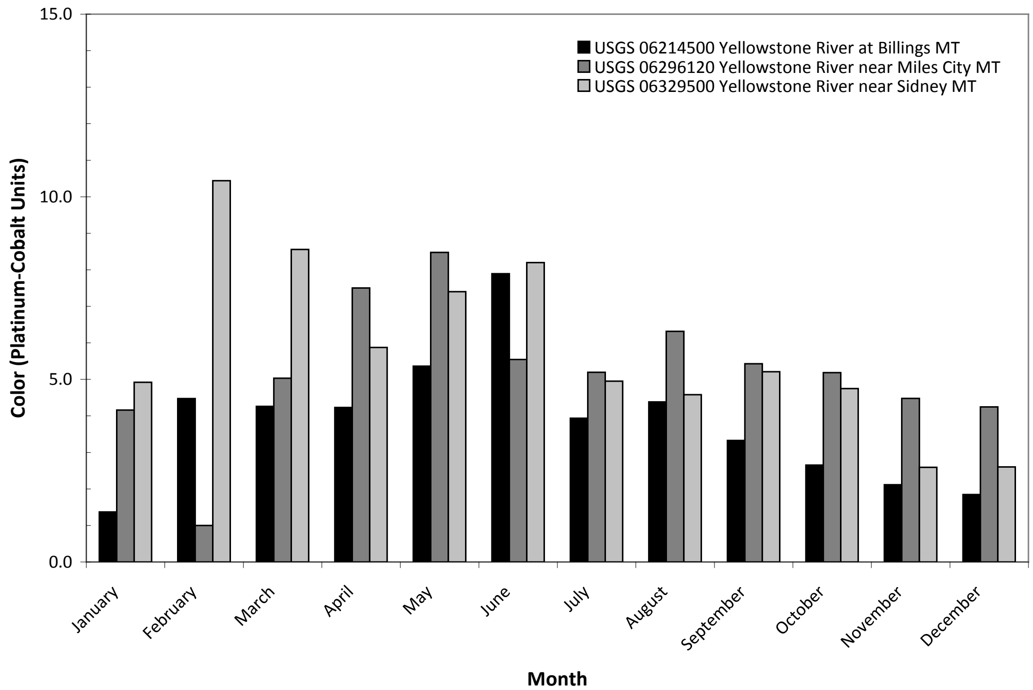

Figure 8-7. Monthly mean true color measurements for the lower Yellowstone River (1963-1970).
All months had at least 2+ observations. Observations vary from year to year, however, during low-flow conditions remain relatively consistent. Billings gage shown for reference only.

Given the uncertainty in the estimate, results were verified through an independent measure developed by Cuthbert and Giorgio (1992). In this method the spectrophotometric absorption coefficient at 440 nanometers (g_{440}) was obtained and related to color by the following, g_{440} = (visual color + 0.43) / 15.53, and was then integrated over the spectrum of 400-700 nm according to the spectral dependence of light absorbance[46] and the reference solar spectral irradiance from the American Society of Testing and Materials (ASTM) G173-03 (ASTM, 2011) (**Figure 8-8**). The irradiance weighted absorption coefficient was 0.128 m^{-1} yielding an average of the two methods of k_{eb}=0.121 m^{-1}. Thus the two methods were very similar.

[46] The spectral dependence of light absorbance is defined by the following equation (Cuthbert and Giorgio, 1992); $g_\lambda = g_{440}\, e^{[-S(\lambda-440)]}$, where g_λ=light attenuation coefficient (m^{-1}) at a specified wavelength (nm), g_{440}=the light attenuation coefficient at 440 nanometers, and the S=slope which falls in a fairly narrow range of values reported in the literature (0.01688 nm^{-1} used).

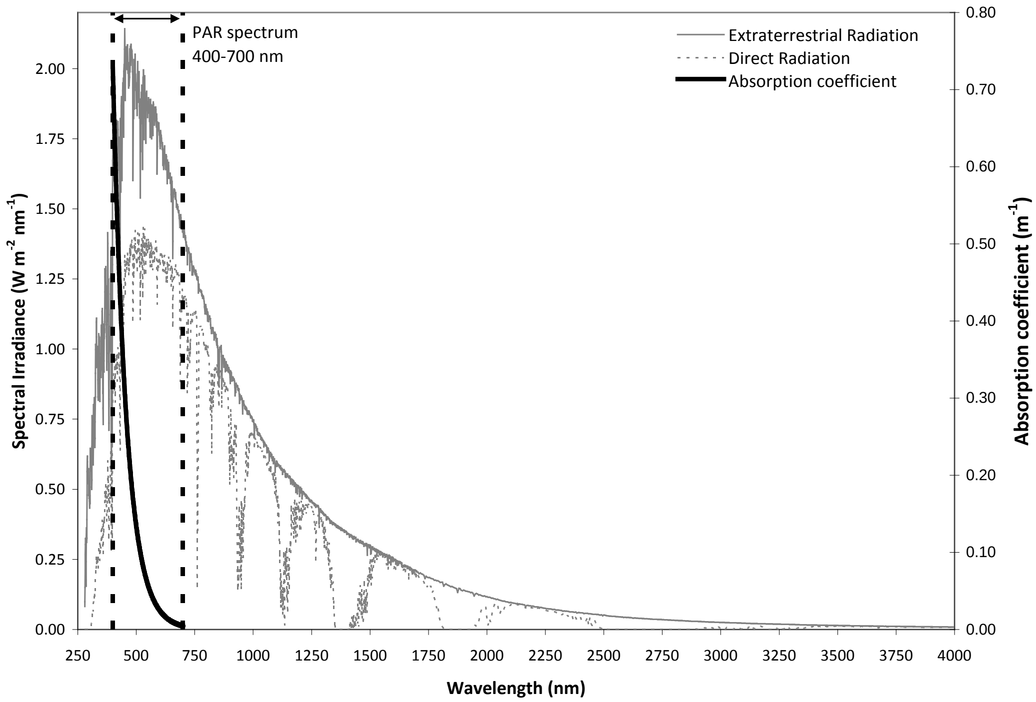

Figure 8-8. ASTM reference spectra used to evaluate the net absorption coefficient for color.

The other partial attenuation coefficients were determined according to theoretical and empirical considerations. The estimate of partial light attenuation for inorganic suspended solids (ISS) was based on the relationship from Blom, et al., (1994) where α_i is roughly proportional to fall velocity (m day^{-1}). In their work, values of 0.0064-0.059 (m^2 g^{-1}) were reported, and particles with the smallest size (e.g., settling velocity) or alternatively highest organic content yielded the highest light attenuation. Values of 0.019-0.137 m^2 g^{-1} have been reported elsewhere (Van Duin et al., 2001) citing (Bakema, 1988; Blom et al., 1994; Buiteveld, 1995; Di Toro, 1978).

In the Yellowstone River, we estimated α_i from low flow suspended sediment fall measurements from USGS (August 1 – April 30). Only five different size classes were characterized in their work ranging from 0.004 to 0.25 mm (from clay particle sizes to sands), and size classes, not actual velocity measurements were reported. Thus fall velocities were back-calculated using Stokes' law (**Equation 8-6**) (Chapra, 1997) where, v_s= settling velocity [m s^{-2}], α = Corey shape factor (assumed to be 1 in this application), g = acceleration of gravity [m s^{-1}], $\rho_{s,w}$ = densities of sediment and water [kg m^{-3}] (assume silt, 2,650), μ = dynamic viscosity of water at 20˚C [kg m^{-1} s^{-1}], and d = effective particle diameter [m].

(Equation 8-6)

$$v_s = \alpha \frac{g}{18} \frac{(\rho_s - \rho_w)}{v} d^2$$

From the Stokes relationship, mean fall velocity was estimated to be 0.012 m d^{-1} with a mean (D_{50}) sediment diameter of 0.0004 mm (**Figure 8-9**). Most of the particles in suspension in the Yellowstone River are therefore quite small (e.g., clays).

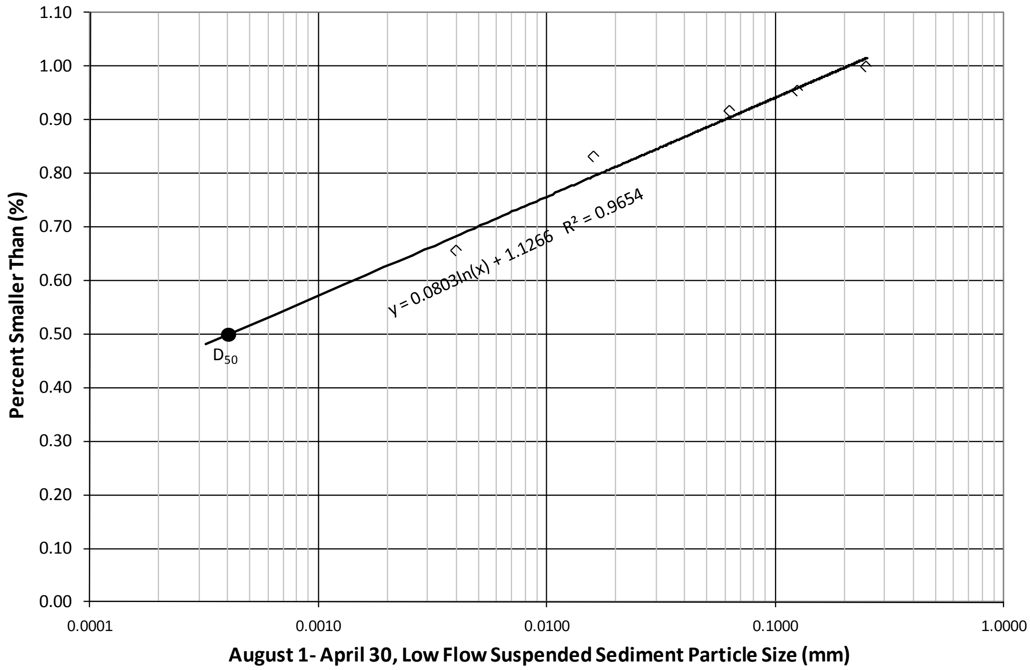

Figure 8-9. Suspended sediment particle size (mm) in Yellowstone River during low-flow conditions. Data taken as average of fall diameter measurements at USGS 06295000 Yellowstone River at Forsyth MT and USGS 06329500 Yellowstone River near Sidney MT.

Based on the computed fall velocity, α_i from Blom et al. (1994) was reconstructed (**Figure 8-10**). Using a very simple linear regression DEQ estimated α_i to be 0.05-0.06 $m^2\ g^{-1}$ which is near the mid to upper end of the literature (Van Duin et al., 2001). Given that the estimate is very near the upper range of Blom et al., (1994), and also very similar to the 0.052 $m^2\ g^{-1}$ reported by Di Toro (1978), the Q2K default value was used (which happens to be from Di Toro).

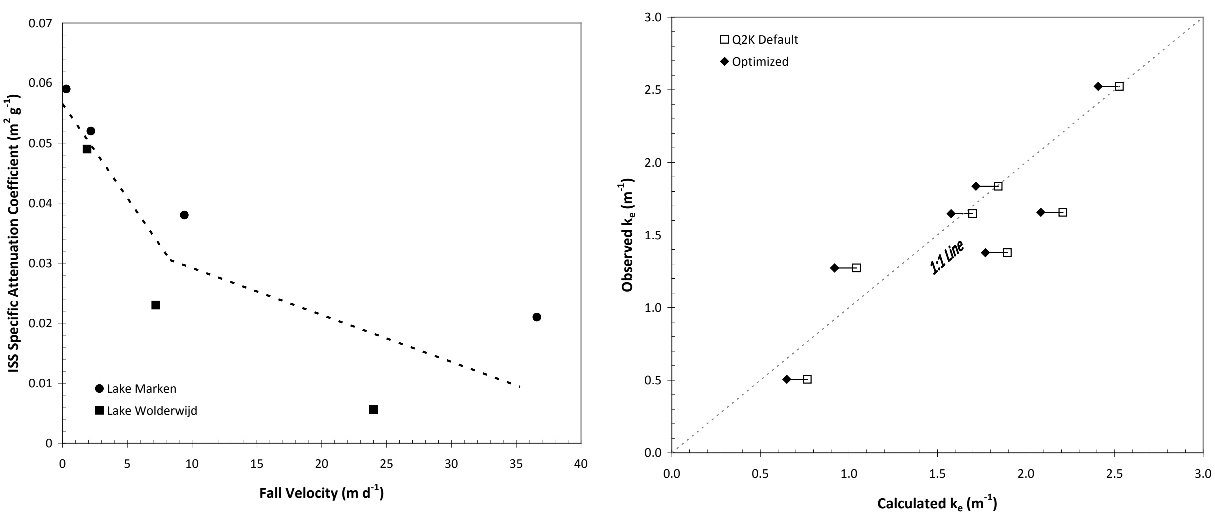

Figure 8-10. Optimized partial extinction coefficients for the remaining particles (αi, αo, αp & αpn). (Left) Relationship between fall velocity and inorganic suspended solids partial attenuation coefficient ($m^2\ g^{-1}$) based on data from Blom et al., (1994). (Right) Optimization of remaining partial attenuation values for detritus and phytoplankton showing the relative shift toward the 1:1 line.

For the remaining partial attenuation coefficients (i.e., detritus[47] and phytoplankton), water quality and k_e measurements were used to find optimal values. The non-linear part of the chlorophyll equation was set to zero to be consistent with recent optics literature (Van Duin et al., 2001). Then from evaluation of the remaining terms, it was established that the default recommendations in Q2K are quite good (**Table 8-8**). The greatest overall improvement resulted in root mean squared error (RMSE) of 0.32 to 0.27 m^{-1}.

Table 8-8. Final optic coefficients for Yellowstone River Q2K model.
Final/optimized values are essentially from Di Toro (Di Toro, 1978).

Suspended Material	Parameter	Units	Q2K Default	Range	Final/Optimized Value
Water & Color	k_{eb}	m^{-1}	0.20	0.02-6.59[1]	0.12
Inorganic Solids	α_i	$m^2\ g^{-1}$	0.052	0.019-0.137[2]	0.052
Detritus	α_o	$m^2\ g^{-1}$	0.174	0.008-0.174[2]	0.174
Phytoplankton	α_p	$m^2\ g^{-1}$	0.0088	0.0088-0.031[2]	0.031
Phytoplankton	α_{pn}	$m^2\ g^{-1}$	0.054	n/a	not used

[1] Range of inland waters reported by (Kirk, 1994) at 440 nm, adjusted to irradiance from 400-700
[2] From Van Duin et al. (2001) which includes a review of the followings studies (Bakema, 1988; Blom et al., 1994; Buiteveld, 1995; Di Toro, 1978).

8.11 Settling Velocities

The last thing considered was settling velocities. These were detailed to some extent in **Section 8.10**. Recall that inorganic settling velocity was 0.012 m d^{-1} (based on a D_{50} of 0.0004 mm). However, it is unclear whether particulate settling would actually occur (Hjulstrom, 1935)[48]. Since turbulence tends to advect sediment both downward and upward uniformly (Whiting et al., 2005), the calculated settling velocity of 0.012 m d^{-1} was used directly in the modeling without adjustment.

Phytoplankton settling rates were calculated in a similar fashion by assuming dynamic equilibrium between re-suspension and deposition (i.e., such that the net effect is represented). The algal biovolumes detailed previously were used to determine the particle size of algae (≈8 μm[49])[50] which according to Stoke's law was 0.086 m day^{-1}. This appears to be a very reasonable first estimate based on Bowie et al., (1985). Since detritus data were not available, it is reasonable to believe detritus particles in suspension are a similar size during low-flow conditions. Thus 0.086 m day^{-1} was used for that as well.

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[47] Detritus was estimated from observed particulate organic carbon (POC) data (mg L^{-1}) using the SOC:VSS and AFDM:Chl*a* ratio during 2007 (4.3 mg L^{-1} VSS: 1 mg L^{-1} SOC).
[48] Analysis of critical shear stress (τ_c) indicates that incipient motion requirements are greatly exceeded (the actual shear stress of 6.3 N m^{-2} is several orders of magnitude above the τ_c of 0.005 N m^{-2}).
[49] Geometric mean of phytoplankton biovolumes taken (307 μm^3) and particle diameter estimated using the volume of a sphere where $d = 2\sqrt[3]{\mu m^3 \frac{3}{4\pi}}$. Density of phytoplankton from (Chapra, 1997) as 1027.
[50] Particle sizes were actually for benthic algae (not phytoplankton). However, it is believed that much of the algae in suspension are of benthic origin (Bahls, 1976b).

9.0 Model Calibration and Validation

Details regarding the model calibration are detailed in this section. Supporting information is found in **Section 8.0**.

9.1 Approach

The approach towards calibration and validation for the Yellowstone River is shown in **Figure 9-1**. It consisted of iterative adjustment of rate coefficients until the criteria identified in **Table 9-1** were met. Validation tests were then performed to confirm whether or not the model was acceptable for use. The approach is typical of classic split sample calibration-validation methodology where one dataset is used solely for model calibration and a second independent dataset is used for model validation[51].

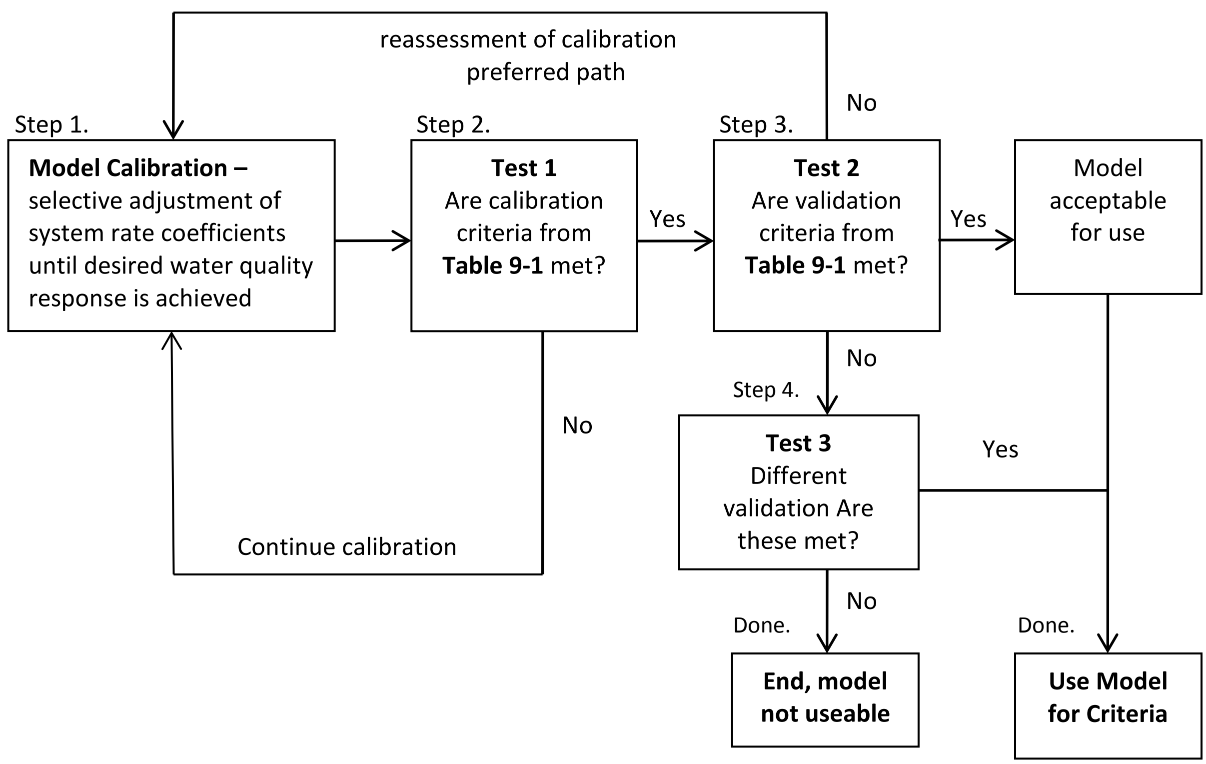

Figure 9-1. Model calibration and validation approach for the Yellowstone River.
Calibration was completed iteratively until acceptability criteria in **Table 9-1** were achieved. Validation or confirmation tests were then performed to identify whether the model was acceptable for use.

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[51] Also termed corroboration, confirmation, or verification. Two independent low flow datasets were used in model evaluation. The first was August 17-26, 2007 (warm-weather) for calibration and (2) September 11-20, 2007 (cooler-weather) for validation. We also had a third independent dataset for use which was another warm weather dataset collected by USGS in August of 2000. Similarity of environmental conditions (e.g., light, temperature, etc.) is not necessarily required in mechanistic studies as process-based models explicitly account for such variation (see Chapra, 2003).

9.2 Calibration and Validation Time-Period

The calibration and validation periods were constrained to two 10-day periods over which conditions were approximately steady-state and water quality sampling was completed. These were:

- **Calibration:** August 17-26, 2007
- **Validation**[52]**:** September 11-20, 2007

Each period was believed to be appropriate in minimizing streamflow and climatic variability, reducing the possibility of YSI sonde interference (i.e., from biofouling), and meeting the travel time requirements of the river. The time-frame was also similar conditions used in wasteload allocation studies (EPA, 1986b).

9.3 Evaluation Criteria for Calibration and Validation

Two statistical tests were selected to assess the sufficiency of the Yellowstone River model calibration. These include relative error (RE) and the root mean squared error (RMSE). RE is a measure of the percent difference between observed and predicted ordinates (**Equation 9-1**), where *RE* = relative error, Obs_i = observed state variable, Sim_i = Simulated state variable. Overall system RE should approach 0% (on average) and recommendations for specific model state-variable are shown in **Table 9-1**.

(Equation 9-1)

$$RE = 100 \times \sum_{i=1}^{n} \left(\frac{Sim_i - Obs_i}{Obs_i} \right)$$

Root mean squared error (RMSE) was also used which is a common objective function for water quality model calibration (Chapra, 1997; Little and Williams, 1992). It compares the difference between modeled and observed ordinates and uses the squared difference as the measure of fit. Thus a difference of 10 units between the predicted and observed value is 100 times worse than a difference of one unit. Squaring the differences also treats both overestimates and underestimates as undesirable. The root of the averaged squared differences is then taken as RMSE. Calculation of RMSE is shown in **Equation 9-2** (Diskin and Simon, 1977), where *n*=is the number of observations being evaluated.

(Equation 9-2)

$$RMSE = \sqrt{\frac{1}{n} \sum_{i=1}^{n} [Obs_i - Sim_i]^2}$$

The utility of RMSE is that error is expressed in the same units as the data being evaluated. Thus by decreasing RMSE, model error is inherently reduced.

[52] The USGS dataset detailed previously (**Section 6.4**) was reserved for additional validation as described later in the document.

Table 9-1. Recommended relative or standard errors for water quality model simulations.

State Variable	QAPP criterion ± (%)	Literature Recommendation ± (%)
Temperature	5 (or 1°C)	5[1]
Dissolved Oxygen	10 (or 0.5 mg L^{-1})	10[1], ≤10[2]
Chlorophyll-*a* – Phytoplankton	10	40[1], 30-35[3], 30[2], (0.5 µg L^{-1})[2]
Chlorophyll-*a* – Bottom Algae	20	10-28[4]
Nitrate	Not specified	30[1], (25 µg L^{-1})[2a]
Ammonia	Not specified	50[1], (5 µg L^{-1})[2a]
Dissolved orthophosphate	Not specified	40[1],(2 µg L^{-1})[2a]

[1]Arhonditsis and Brett (2004), 153 aquatic modeling studies in lakes, oceans, estuaries, and rivers.
[2]Thomann (1982), studies on 15 different waterbodies (rivers and estuaries). [2a]Lake Ontario only.
[3]Håkanson (2003), coefficient of variation for River Danube (days to weeks).
[4] Biggs (2000c), for 3 rivers with varying algae densities (high, medium, low) and *n*= 10 replicates per location (very close to cross-section *n*= 11 in the present study).

9.4 Data for Calibration

Data for calibration comes primarily from the field program described in **Section 6.0** which we have summarized in **Table 9-2**.

Table 9-2. Data used in calibration and validation of Q2K for the lower Yellowstone River.

Data type	Measurement	Increment
Water chemistry/algae	1. EWI samples of nutrients, suspended solids, etc. 2. Benthic/suspended algae collections	Instantaneous
Diurnal water quality	1. YSI sonde deployments (DO, pH, temperature, etc.)	sub-hourly
Others described in **Section 9.0** Algae, kinetics, sediment/benthics	1. Filamentous floating algal characterization 2. Academy of natural science taxonomic evaluations 3. C:N:P stoichiometry 4. Productivity/respiration experiments 5. Minimum cell quota estimates 6. Reaeration from sonde DO delta 7. Sediment oxygen demand measurement	n/a n/a n/a n/a n/a n/a n/a

Of primary importance were the water chemistry and YSI sonde diurnal data which are shown in great detail in **Section 10.0**[53]. Since the datasonde variables are not described anywhere else in this document (and were condensed into single point values by DEQ to form the data in **Section 10.0**), they are briefly reviewed here.

YSI time-series were 83% and 74% complete for the calibration and validation, which is sufficient for practical evaluation of river conditions. Procedures to identify missing or erroneous data (e.g., due to biofouling including snagged drifting filamentous algae interference) were identified in the SAP addendum (Suplee et al., 2006a), and an example of a time-series for one of DEQ's sites is shown in **Figure 9-2**. A number of issues[54] were identified at this and other sites, and standard procedures such as

[53] There was a great deal of ancillary information also used in calibration as detailed in **Section 8.0**.
[54] Turbidity and Chl*a* florescence were most routinely affected. Spikes in turbidity (errant data as shown in the figure) or the suppression or "quenching" of chlorophyll fluorescence (also shown from approximately 8:00 a.m. to 6:00 p.m., when the sun was at higher zeniths) were the primary problems identified. Suppression is the process

the average of the prior and following observation, or parallel estimation procedures with adjacent stations (Linacre, 1992) were used to synthesize or reconstruct errant data. Data was condensed into mean repeating day series as required in Q2K which forms the basis for the analysis shown elsewhere in the document.

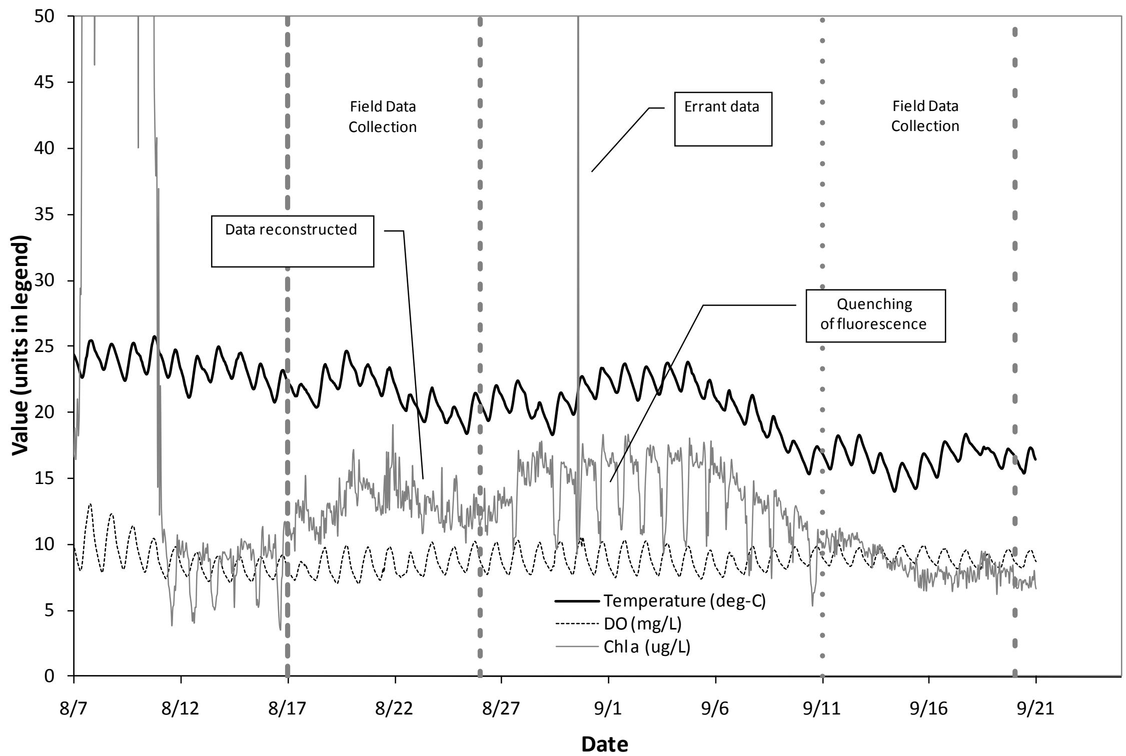

Figure 9-2. Example of YSI sonde data from the Yellowstone River in 2007.
Temperature, DO, and Chl*a* shown for YSI-20, upstream of Cartersville canal.

9.5 Water Chemistry Relationships with Model State-variables

Information presented previously requires further explanation for context within the model, in particular, the relationship between water chemistry and Q2K model state-variables. These are shown in **Table 9-3** for those that are not obvious.

whereby algae change their fluorescence when absorbed light energy exceeds their capacity for utilization (Müller et al., 2001; Vaillancourt, 2008). It can change fluorescence by a factor of 10 with no change in Chl*a* concentration.

Table 9-3. Relationship between Q2K state-variables water chemistry collections.
Definitions shown at the bottom of the table or are defined within the table.

Model State Variable	Symbol	Water Chemistry Relationship & Calculation [as taken from Chapra et al., (2008)][1]
Benthic/phytoplankton Chl*a*	a_p	Chlorophyll-*a* (Chl*a*)
Detritus	m_i	TSS - VSS
Inorganic suspended solids	m_o	$VSS – r_{da}\ a_p$
Total suspended solids	Calculated	$m_i + m_o + r_{da}\ a_p$
Nitrate nitrogen	n_n	$NO_2^- + NO_3^-$ (nitrate plus nitrite)
Ammonium nitrogen	n_a	NH_4^+ - (ammonia)
Organic nitrogen	n_o	$TN - NO_2^- + NO_3^- - NH_4^+ - r_{na}\ a_p$
Total nitrogen	Calculated	$n_n + n_a + n_o + r_{na}\ a_p$
Inorganic phosphorus	p_i	SRP (soluble reactive phosphorus)
Organic phosphorus	p_o	$TP – SRP – r_{pa}\ a_p$
Total phosphorus	Calculated	$p_i + p_o + r_{pa}$Chl*a*
CBOD ultimate	Calculated	$c_f + r_{oc}\ r_{ca}\ a_p + r_{oc}\ r_{cd}\ m_o$
Total organic carbon (TOC)	Calculated	DOC + POC

[1]TSS = total suspended solids, VSS = volatile suspended solids, r_{da} = ratio of ash-free dry weight to phytoplankton Chl*a*, r_{na} = ratio of nitrogen to Chl*a*, r_{pa}= ratio of phosphorus to Chl*a*, c_f = fast oxidizing carbon, DOC = dissolved organic carbon, POC = particulate organic carbon, r_{oc} = ratio of oxygen to carbon, r_{oc} = ratio of carbon to Chl*a*, r_{cd} = ratio of carbon to ash-free dry weight.

9.6 Model Configuration and Solution

Model time-step, runtime, and element sizing were completed according to courant stability and critical segment sizes identified in Chapra (1997). A time step of 0.1 hours was needed to ensure stability for some of the shorter model elements, and critical element size (Δx) balanced with dispersion and other stability requirements. The Euler and Brent solution methods were used as they are computationally efficient. During this work it was identified that steady condition boundary conditions induce oscillatory behavior when using shorter element lengths (due to advection being much greater than dispersion). Use of correctly timed diurnal variation was found to remedy this problem, but was not efficient, and so instead we used a reduced number of elements to correct this issue (i.e., through the addition of numerical dispersion). We also considered initial condition effects in the model in regard to benthic algal biomass (recall that initial conditions for the algae are fixed in the model, 0.1 mg Chl*a* m^{-2}) and thus require time to grow to steady state conditions). A run time of 60-90 days was found to be necessary to ensure algal biomass had achieved maximum levels by the end of the simulation for existing conditions (**Figure 9-3**). A simulation length of 90 days was obligatory to ensure that initial conditions do not influence the final model output[55]. It will be shown later that as nutrient conditions in the model increase (thus reflecting a higher growth rate), the time to achieve equilibrium biomass will decrease. Thus the run time required to reach equilibrium conditions in model development should not be

[55] This computational necessity is an artifact of starting biomass in the model being assumed to be 0.1 mg Chl*a* m^{-2}. In the river, algal densities would start about 1 order of magnitude higher and 0.1 mg Chl*a* m^{-2} would rarely occur. Thus the 60-90 day response time is likely a significant overestimate. Note that this only applies to existing conditions (un-enriched nutrient levels) and that response time varies as a function of level of enrichment.

confused with times to achieve nuisance biomass as discussed in design flow specification in **Section 12.2**, or in the scenarios outlined in **Section 13.0**.

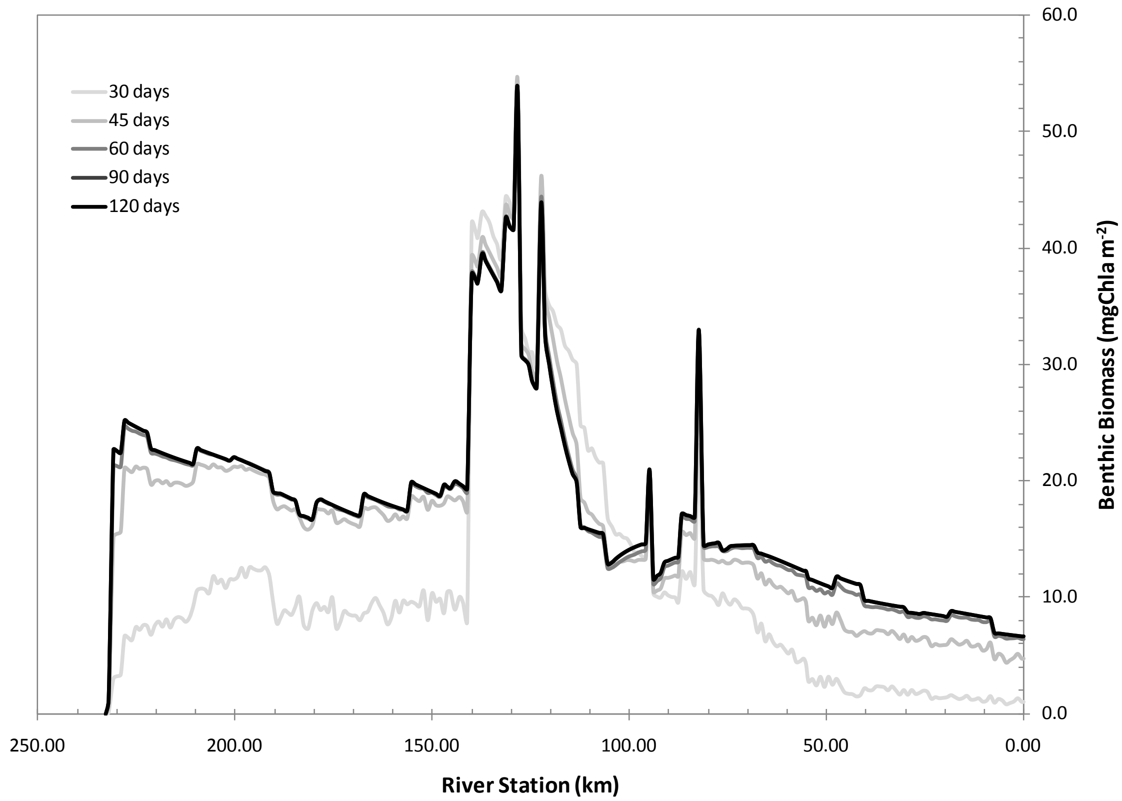

Figure 9-3. Evaluating model runtime requirements for the Yellowstone River.
Existing condition model run reflecting time required to reach steady state algal biomass. A run of approximately 90 days is required to reach convergence.

9.7 Model Calibration

Two different methods were used for model calibration: manual calibration and autocalibration. Each is briefly described below.

9.7.1 Manual Calibration

The manual calibration relied primarily on knowledge of system coefficients and river response, field observations, past modeling experiences, and nutrient work elsewhere in Montana. Primary consideration was given toward preservation of the theoretical constructs of the model, not just curve fitting. We relied on the following indicators to complete our calibration:

- Diurnal state-variables such as temperature, dissolved oxygen and pH. These were thought to encompass eutrophication and algal photosynthetic response.

- Water chemistry measurements, which are suggestive of water quality kinetics of the river including algal uptake, death and decomposition, settling, etc.

- Algal biomass measurements, which characterize algal growth rates, loss mechanisms (death, respiration, etc.), and the effect of light.

- Field rate measurements, which provide direct estimation of some of the model kinetics [e.g., sediment oxygen demand (SOD), primary productivity, etc.].

- Other indicators as described in **Section 8.0**.

Over forty rate coefficients were calibrated. General information on model calibration can be found in the literature (ASTM, 1984; Reckhow and Chapra, 1983; Thomann, 1982) and explicit detail regarding the methodology is not presented here.

9.7.2 Autocalibration

An automated calibration was also employed using a genetic algorithm from Tao (2008). Shuffled-complex evolution (SCE) was used to optimize model parameters towards global optimality. After several implementations of the automated procedure, however, it was found that very little improvement could be made over that of the initial manual calibration. Furthermore, the necessity of co-calibration with AlgaeTransect2K (AT2K) largely negated any applicability of the automated method (i.e., the two models are independent of each other and should be optimized together). Hence the autocalibration was abandoned.

9.8. Calibrated Rates for Q2K on the Yellowstone River

Calibrated rates for the Yellowstone River Q2K model along with recommended literature ranges are shown in the following pages (**Tables 9-3, 9-4, 9-5, and 9-6)**. They encompass much of the information detailed previously, and are supported by information in **Section 8.0**. Literature ranges for the calibrated values are also shown, and are taken from a compilation of studies, including several that were directly applicable to QUAL2E/K. For each table, a brief overview is provided for how the final calibrated values were determined. At all times, the calibration was within the specified literature range.

Light and heat parameters are shown in **Table 9-4**. They were calibrated through evaluation of solar radiation and water temperature as noted in the table. Sediment parameters were found to be relatively insensitive and were identified in our field measurements according to bed consistency.

Table 9-4. Light and heat parameters used in the Yellowstone River Q2K model.
Based on calibration and literature review.

Parameter Description	Symbol	Units	Initial Estimate	Literature Range[1]		Final Calibrated Value
				Min	Max	
% of radiation that is PAR	n/a	dimensionless	0.47	n/a	n/a	0.47
Extinction from light/color	k_{eb}	m^{-1}	0.2	0.02	6.59	0.121[1]
Linear Chl*a* light extinction	α_p	$(\mu g\ A\ L^{-1})\ m^{-1}$ *	0.031	0.009	0.031	0.031[1]
Nonlinear Chl*a* extinction	α_{pn}	$(\mu g\ A\ L^{-1})\ m^{-1}$	not used	n/a	n/a	not used (0)[1]
ISS light extinction	α_i	$(mg\ D\ L^{-1})\ m^{-1}$	0.052	0.019	0.137	0.052[1]
Detritus light extinction	α_o	$(mg\ D\ L^{-1})\ m^{-1}$	0.174	0.008	0.174	0.174[1]
Atmospheric solar model	n/a	n/a	Bras	n/a	n/a	Bras[2]
Bras solar parameter	n_{fac}	dimensionless	2	2	5	2.8[2]
Atmospheric emmisivity model	n/a	n/a	Brutsaert	n/a	n/a	Brutsaert[3]

Table 9-4. Light and heat parameters used in the Yellowstone River Q2K model.
Based on calibration and literature review.

Parameter Description	Symbol	Units	Initial Estimate	Literature Range[1]		Final Calibrated Value
				Min	Max	
Wind speed function	n/a	n/a	Adams 1	n/a	n/a	Adams 2[3]
Sediment thermal thickness	H_s	cm	10	n/a	n/a	10
Sediment thermal diffusivity	α_s	$cm^2\ s^{-1}$	0.0	0.002	0.012	0.009[4]
Sediment density	ρ_s	$g\ cm^{-3}$	2.2	1.5	2.7	2.1[4]
Water density	ρ_w	$g\ cm^{-3}$	1.0	1.0	1.0	1.0
Sediment heat capacity	C_{ps}	$cal\ g^{-1}\ °C^{-1}$	0.2	0.19	0.53	0.21[4]
Water heat capacity	C_{pw}	$cal\ g^{-1}\ °C^{-1}$	1.0	1.0	1.0	1.0

[1]As determined in Section 8.10 from Van Duin et al. (2001) for optics which includes a review of the following studies (Bakema, 1988; Blom et al., 1994; Buiteveld, 1995; Di Toro, 1978) and from Chapra et al., (2008) for sediment.
[2]As determined in **Section 7.4**.
[3]Calibrated using observed water temperature data.
[4]Determined from field estimates [95% gravel (rock) and 5% clay] from tables in Chapra et al., (2008).
*Unit abbreviation, A=Chl*a*.

Calibrated rate coefficients for carbon, nitrogen, and phosphorus are shown in **Table 9-5**. They were determined with the assistance of the information presented in **Section 8** as well as the literature review. A reference regarding each of the calibrated values is provided along with the suggested literature range. It should be noted that initial parameter estimates are based on previous recommendations or initial data evaluations, which must be adjusted on a per-system basis through model calibration. Thus the magnitude of change from the initial parameter estimate is not a factor of whether a calibration is suitable or not, rather the fit between the observed and simulated data is.

Calibrated parameters for benthic and planktonic algae in the lower Yellowstone River are shown in **Table 9-6** and **Table 9-7**. The kinetics between the two algal types varied as a function of growth rate. Since growth rate is a function of cell size or volume (Harris, 1986), we assumed that algae in suspension (phytoplankton) would be smaller and grow faster (therefore having lower subsistence quotas, higher uptake rates, and lower half-saturation constants) than larger (benthic) algae. In regard to final calibrated algal rates, they were well within the specified literature range and do not differ greatly from past studies completed elsewhere in the state (Knudson and Swanson, 1976; Lohman and Priscu, 1992; Peterson et al., 2001; Watson et al., 1990).

Table 9-5. C:N:P rate coefficients used in the Yellowstone River Q2K model.
Based on calibration and literature review.

Parameter Description	Symbol	Units	Initial Estimate	Approximate Range[1]		Final Calibrated Value
				Min	Max	
Stoichiometry:[2]						
Carbon	gC	grams	40	25	60	43
Nitrogen	gN	grams	7.2	4	20	4.7
Phosphorus	gP	grams	1	1	1	1
Dry weight	gD	grams	100	65	130	107
Chlorophyll	gA	grams	1	0.4	3.5	0.4
Carbon:						

Table 9-5. C:N:P rate coefficients used in the Yellowstone River Q2K model.
Based on calibration and literature review.

Parameter Description	Symbol	Units	Initial Estimate	Approximate Range[1]		Final Calibrated Value
				Min	Max	
Fast CBOD oxidation rate	k_{dcs}	d^{-1}	0.2	0.005	5.0	0.2
Temp correction	θ_{dc}	dimensionless	1.05	1.02	1.15	1.05
Nitrogen:						
Organic N hydrolysis rate	k_{hn}	d^{-1}	0.2	0.001	1.0	0.1[3]
Temp correction	θ_{hn}	dimensionless	1.08	1.02	1.08	1.08
Organic N settling velocity	v_{on}	$m\ d^{-1}$	0.1	0	0.1	0[3]
Ammonium nitrification rate	k_{na}	d^{-1}	1.0	0.01	10	2.5[3]
Temp correction	θ_{na}	dimensionless	1.08	1.07	1.10	1.08
Nitrate denitrification rate	K_{dn}	d^{-1}	0	0.002	2.0	0.1[3]
Temp correction	θ_{dn}	dimensionless	1.05	1.02	10.9	1.05
Sediment denitrification trans.	v_{di}	$m\ d^{-1}$	0	0	1	0
Temp correction	θ_{di}	dimensionless	1.05	1.02	1.08	1.05
Phosphorus:						
Organic P hydrolysis rate	k_{hp}	d^{-1}	0.2	0.001	1	0.1[3]
Temp correction	θ_{hp}	dimensionless	1.05	1.02	1.09	1.05
Organic P settling velocity	v_{op}	$m\ d^{-1}$	0.1	0	0.1	0.012[3]
SRP settling velocity	v_{ip}	$m\ d^{-1}$	0	0	0.1	0
SRP sorption coefficient	k_{dpi}	$L\ mg\ D^{-1}$	0	n/a	n/a	0
Sed P oxygen attenuation	K_{spi}	$mg\ O^2\ L^{-1}$	20	n/a	n/a	not used (20)
Suspended Solids:						
ISS settling velocity	v_i	$m\ d^{-1}$	0.1	0	30	0.012[5]
Detritus dissolution rate	k_{dt}	d^{-1}	0.5	0.05	3	0.25
Temp correction	θ_{dt}	dimensionless	1.05	1.04	1.08	1.05
Detritus settling velocity	v_{dt}	$m\ d^{-1}$	0.1	0	1	0.05[5]

[1] According to the literature (Bowie et al., 1985; Chapra, 1997; Chaudhury et al., 1998; Cushing et al., 1993; de Jonge, 1980; Drolc and Koncan, 1999; Fang et al., 2008; Kannel et al., 2006; Ning et al., 2000; Park and Lee, 2002; Turner et al., 2009; Van Orden and Uchrin, 1993).
[2]From the sestonic C:N:P analysis detailed in **Section 8.4.**
[3]From calibration.
[4]From settling velocity estimates in **Section 8.11**.

Table 9-6. Bottom algae Q2K parameterization for the lower Yellowstone River.
Based on calibration and literature review.

Parameter Description	Symbol	Units	Initial Estimate	Approximate Range[1]		Final Calibrated Value
				Min	Max	
Max growth rate	C_{gb}	$mg\ A\ m^{-2}\ day^{-1}$*	400	15	500	400[2]
Temp correction	q_{gb}	dimensionless	1.07	1.01	1.2	1.07
Respiration rate	k_{rb}	day^{-1}	0.3	0.02	0.8	0.2[3]
Temp correction	q_{rb}	dimensionless	1.07	1.01	1.2	1.07
Excretion rate	k_{eb}	day^{-1}	0.0	0.00	0.5	0
Temp correction	q_{db}	dimensionless	1.07	1.01	1.2	1.07
Death rate	k_{db}	day^{-1}	0.0	0.00	0.8	0.3[4]
Temp correction	q_{db}	dimensionless	1.07	1.01	1.2	1.07
External N half-sat. constant	k_{sPb}	$\mu g\ N\ L^{-1}$	350	10	750	250[4]
External P half-sat. constant	k_{sNb}	$\mu g\ P\ L^{-1}$	100	5	175	125[4]

Table 9-6. Bottom algae Q2K parameterization for the lower Yellowstone River.
Based on calibration and literature review.

Parameter Description	Symbol	Units	Initial Estimate	Approximate Range[1]		Final Calibrated Value
				Min	Max	
Inorganic C half-sat. constant	k_{sCb}	mole L^{-1}	1.30E-05	n/a	n/a	not used (0)
Light model	n/a	n/a	Smith	n/a	n/a	Half saturation
Light constant	K_{Lb}	langley day^{-1}	100	30	90	60[4]
Ammonia preference	k_{hnxb}	µg N L^{-1}	15	5	30	20[4]
Subsistence quota for N	q_{0Nb}	mg N mgA^{-1}	0.7	0.5	5.0	3.20[5]
Subsistence quota for P	q_{0Pb}	mg P mgA^{-1}	0.1	.05	0.5	0.13[5]
Maximum uptake rate for N	r_{mNb}	mg N mgA^{-1} day^{-1}	70	5	100	35[4]
Maximum uptake rate for P	r_{mPb}	mg P mgA^{-1} day^{-1}	10	1	15	4[4]
Internal N half-sat. constant	K_{qNb}	mg N mgA^{-1}	0.9	0.25	5.0	3.20[4]
Internal P half-sat. constant	K_{qPb}	mg P mgA^{-1}	0.13	0.025	0.5	0.09[4]

[1]According to the literature (Auer and Canale, 1982; Biggs, 1990; Borchardt, 1996; Bothwell, 1985; Bothwell, 1988; Bothwell and Stockner, 1980; Bowie et al., 1985; Chapra, 1997; Chaudhury et al., 1998; Cushing et al., 1993; Di Toro, 1980; Drolc and Koncan, 1999; Fang et al., 2008; Hill, 1996; Horner et al., 1983; Kannel et al., 2006; Klarich, 1977; Lohman and Priscu, 1992; Ning et al., 2000; Park and Lee, 2002; Rutherford et al., 2000; Shuter, 1978; Stevenson, 1990; Tomlinson et al., 2010; Turner et al., 2009; Van Orden and Uchrin, 1993).
[2]From discussion in **Section 8.5**.
[3]From light-dark bottle experiments in **Section 8.5**.
[4]Calibrated.
[5]Initial estimate from **Section 8.6**.
*Unit abbreviation, A=Chl*a*.

Table 9-7. Phytoplankton parameter Q2K parameterization for the lower Yellowstone River.

Parameter Description	Symbol	Units	Initial Estimate	Approximate Range[1]		Final Calibrated Value
				Min	Max	
Max growth rate	C_{gp}	day^{-1}	2.5	0.5	3.0	2.3[2]
Temp correction	q_{gp}	dimensionless	1.07	1.01	1.2	1.07
Respiration rate	k_{rp}	day^{-1}	0.3	0.02	0.8	0.2[2]
Temp correction	q_{rp}	dimensionless	1.07	1.01	1.2	1.07
Excretion rate	k_{ep}	day^{-1}	0.0	0.00	0.5	0
Temp correction	q_{dp}	dimensionless	1.05	1.01	1.2	1.07
Death rate	k_{dp}	day^{-1}	0.0	0.00	0.5	0.15[3]
Temp correction	q_{dp}	dimensionless	1.07	1.01	1.2	1.07
External N half-sat. constant	k_{sPp}	µg N L^{-1}	70	5	50	40[3]
External P half-sat. constant	k_{sNp}	µg P L^{-1}	10	10	60	12[3]
Inorganic C half-sat. constant	k_{sCp}	mole L^{-1}	1.30E-05	n/a	n/a	0.00E+00
Light model			Smith	n/a	n/a	Half saturation
Light constant	K_{Lp}	langley day^{-1}	100	30	90	60[3]
Ammonia preference	k_{hnxp}	µg N L^{-1}	15	5	30	20[3]
Subsistence quota for N	q_{0Np}	mgN mgA^{-1}*	0.7	0.5	5.0	2.50[4]
Subsistence quota for P	q_{0Pp}	mgP mgA^{-1}	0.1	.05	0.5	0.10[4]
Maximum uptake rate for N	r_{mNp}	mgN mgA^{-1} day^{-1}	70	5	100	40[3]
Maximum uptake rate for P	r_{mPp}	mgP mgA^{-1} day^{-1}	10	1	15	27[3]
Internal N half-sat. constant	K_{qNp}	mgN mgA^{-1}	0.9	0.25	5.0	2.50[3]
Internal P half-sat. constant	K_{qPp}	mgP mgA^{-1}	0.13	0.025	0.5	0.05[3]
Settling velocity	v_a	m day^{-1}	0.1	0	1	0.05[5]

[1]According to the literature (Auer and Canale, 1982; Biggs, 1990; Borchardt, 1996; Bothwell, 1985; Bothwell, 1988; Bothwell and Stockner, 1980; Bowie et al., 1985; Chapra, 1997; Chaudhury et al., 1998; Cushing et al., 1993; Di Toro, 1980; Drolc and Koncan, 1999; Fang et al., 2008; Hill, 1996; Horner et al., 1983; Kannel et al., 2006; Klarich, 1977; Lohman and Priscu, 1992; Ning et al., 2000; Park and Lee, 2002; Rutherford et al., 2000; Shuter, 1978; Stevenson, 1990; Tomlinson et al., 2010; Turner et al., 2009; Van Orden and Uchrin, 1993).
[2]From the light dark bottle experiments in **Section 8.5**.
[3]Calibrated.
[4]Initial estimate from **Section 8.6**.
[5]From settling velocity estimates in **Section 8.11**.
*Unit abbreviation, A=Chl*a*.

In review of the calibration coefficients described previously, several things should be noted. First, N and P half-saturation constants required for calibration may seem high in comparison with other work (e.g., Bothwell; 1985, Borchardt, 1996; Rier and Stevenson, 2006). However, Bothwell (1989) shows that low saturating levels are probably only valid during the cellular growth, at a time when nutrient supply is high and is not impeded by diffusion through the algal mat. Thus when algal biomasses are larger (or detrital accumulation is significant), it is possible that the nutrient supply and associated gradient is diffusion limited which may explain why higher values are needed to calibrate the model to a natural river. It is also important to realize that the Droop (1974) internal stores model is being used and thus to frame the coefficients in a simple Michaelis-Menton or Monod saturation form is not correct. Rather the actual model response must be considered. By doing so we found that peak algal biomass saturated at around 152 µg/L soluble inorganic nitrogen (SIN) and 48 µg/L SRP (other factors non-limiting) which is well within reason given the literature on the subject . This line of evidence provides additional confidence in the model's predictions.

Similarly, with respect to the algal parameterization, it is commonly misconceived that subsistence quotas scale at Redfield ratio (7:1 by mass). However, Shuter (1978) provides a compilation of minimum cell quota data for N and P vs. biovolume (for phytoplankton) that seem to disprove this. From data on more than 25 algal species it is shown that N to P ratios deviate substantially from Redfield near the minimum cell quota. Recent work by Klausmeier et al., (2004) supports this assertion. They suggest resource acquisition machinery (i.e., nutrient-uptake proteins and chloroplasts) are P-poor, making the N:P ratio higher (ca. 20-30:1 by mass) when algae are nearer to the cell quota. Conversely, under nutrient replete conditions (more like Redfield) P-rich ribosome assembly machinery for exponential growth is more prevalent leading to lower N:P ratios. All of these findings are consistent with the classic work by Goldman et al., (1979) where it is shown that algal cellular N:P ratios are strongly influenced by the alga's growth rate. At very low growth rates (i.e., those approaching the minimum cell quota) cellular N:P ratios increase greatly, up to 45:1 (by mass).

Finally, many of the rate coefficients for nitrogen, phosphorus, or carbon transformations are difficult to evaluate. We can only suggest that they are within the recommended range of the modeling literature (see references in each of the prior tables for specific examples) and result in reasonable modeling outcomes (as shown in **Section 10.0**). However, simulated vs. observed measures are never foolproof and can result in an apparently correct responses for the wrong reason (i.e., multiple parameter sets can satisfy a calibration, albeit in an incorrect way). As a consequence, we took addition steps to evaluate model parameter uncertainty as described in **Section 14.0** using Monte Carlo error propagation methods. Please refer to these sections for addition discussions regarding the utility of the model calibration and associated parameter selection.

10.0 REVIEW OF MODEL OUTPUT AND COMPARISON TO FIELD DATA

The results of the modeling are contained in this section. To assist readers, a statistical summary has been presented first so that quick conclusions can be made (**Table 10-1**). In all but a few cases, (i.e., benthic and phytoplankton algae) we met our Quality Assurance Project Plan (QAPP) criteria or literature recommended acceptance criteria. This required a second validation to do so. Results follow, and a complete discussion about the use of a second validation is described in **Section 11.0**.

Table 10-1. Statistical summary of Q2K model simulations for Yellowstone River.

State-variable	August 2007 (calibration)			September 2007 (validation)			August 2000 (2nd validation)		
	RMSE (units)	RE (%)	met	RMSE (units)	RE (%)	met	RMSE (units)	RE (%)	Met
Streamflow ($m^3 s^{-1}$)	0	0	n/a	0	0	n/a	0	0	n/a
Width (m)	26	3.6	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Depth (m)	0.5	-22.2	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Travel-time (days)	0.01	-0.6	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Reaeration (day^{-1})	0.87	-12.4	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Temperature (°C)	0.24	0.0	yes	0.38	-0.7	yes	1.4	-0.2	yes
TSS (mg D L^{-1})	3.5	9.9	n/a	9.0	-8.9	n/a	2.1	-7.0	n/a
ISS (mg D L^{-1})	2.2	3.3	n/a	9.2	-19.3	n/a	---	---	---
Detritus (mg D L^{-1})	0.9	4.1	n/a	1.5	22.0	n/a	---	---	---
Total N (µg L^{-1})	37	7.3	n/a	67	13.8	n/a	44[3]	7.5	n/a
Organic N	22	-0.6	n/a	79	23.2	n/a			
NO_2+NO_3	9	215	n/a	29	-63.2	n/a			
NH4	8	-36.4	n/a	17	-47.8	n/a			
Total P (µg L^{-1})	9	-11.7	n/a	6	8.9	n/a	5	-11.7	n/a
Organic P	8	-11.5	n/a	5	11.9	n/a	---	---	---
SRP	nd	nd	n/a	nd	nd	n/a	---	---	---
Benthic Algae (mg Chl*a* m^{-2})									
Q2K	4	10.3	yes	23	86.7	***no**	n/a	n/a	n/a
AT2K[1]	22	51.9	***no**	24	-0.8	yes	---	---	---
Phytoplankton[2] (µg Chl*a* L^{-1})	1.9	-2.0	yes	1.1	-3.0	yes	1.8	18.5	no
Dissolved Oxygen (mg O_2 L^{-1})	0.59	-2.5	yes	0.63	0.21	yes	0.36	1.8	yes
CBOD (mg O_2 L^{-1})	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
pH (pH units)	0.16	0.9	n/a	0.18	-1.4	n/a	0.07	-0.2	n/a
Alkalinity (mg $CaCO_3$ L^{-1})	1.5	0.0	n/a	2.5	1.5	n/a	2.9	-0.9	n/a
TOC (mg C L^{-1})	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Conductivity (µS cm^{-1})	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a

n/a = not applicable or not assessed.
nd = not determined (analytical data below reporting limit).
[1]Using alternative growth rate.
[2]Based on YSI sonde data.
[3]With the assumptions detailed in **Section 11.3.5.1**.

10.1 STREAMFLOW HYDROLOGY

Simulated and observed streamflow for the August and September evaluation period is shown in **Figure 10-1**. Due to the fact that the water balance is constrained by gage observations (**Section 7.2**), no deviation occurs between simulated and observed flows (i.e., RMSE=0 $m^3 s^{-1}$ and RE=0%). Simulated flows ranged from 100-135 $m^3 s^{-1}$, with the primary difference being incoming flow at the headwater boundary condition and spatial and temporal variability in irrigation. Flow in September is 15-30% greater than in August, which is attributed to a 15% increase in headwater flow and an equal decrease in diversion rates. Estimates fit well with an independent mass balance model based on evapotranspiration (ET) and crop water use requirements for the region[56].

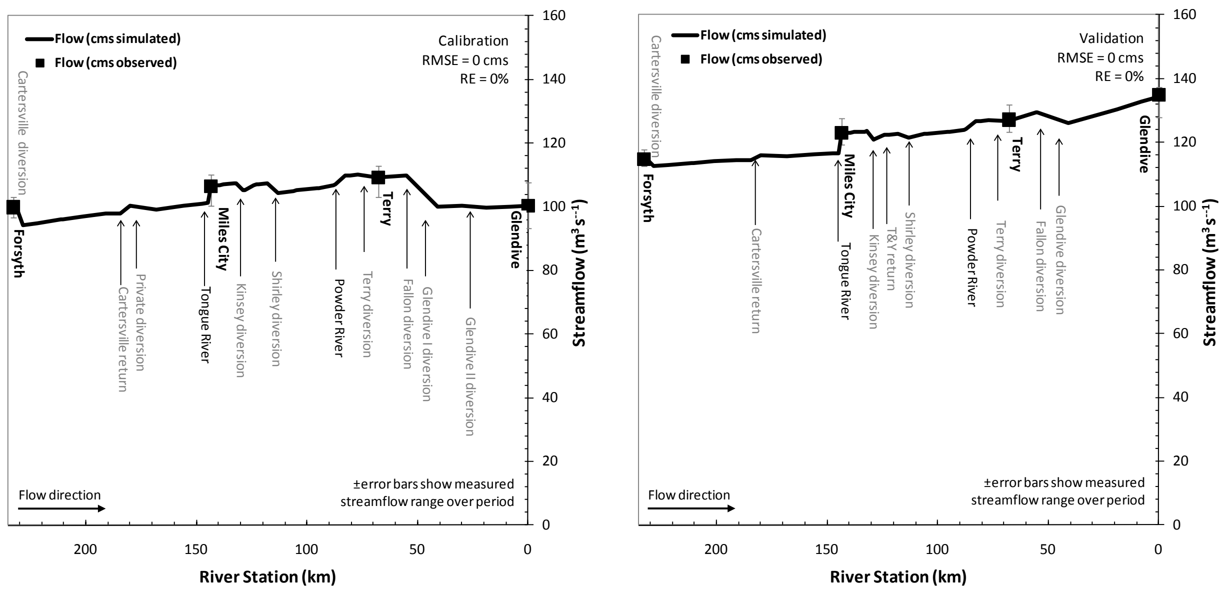

Figure 10-1. Simulated and observed streamflow for the Yellowstone River during 2007.
(Left panel) August calibration. (Right panel) September validation. Flows during August remained relatively constant due to irrigation depletion whereas in September the streamflow profile shows a longitudinal increase in flow due to reductions in irrigation pumping rates. The water balance is typical of irrigated watersheds in Montana where irrigation plays a major role in the surface water balance.

10.2 Mass Transport, Travel-time, and Reaeration

Mass transport reflects the movement of water and pollutants downstream in the river. Both advection and dispersion are calculated and included in the model. Three methods were used to evaluate the hydraulics of the Yellowstone River. These included: (1) a review of simulated river widths and depths, (2) examination of time of travel or residence time, and (3) appraisal of reaeration.

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[56] The ET model was based on peak alfalfa at the Terry AgriMet site which consumed 0.6 and 0.5 cm day^{-1} (0.23 and 0.20 inch day^{-1}) of water in the August and September periods respectively. The mass balance was determined as follows: Diversion = Crop ET + Return Flow + Ditch loss, which for August calculations were 27.81 = 13.64 + 7.38 + 6.79, (all in $m^3 s^{-1}$). Crop ET was based on the NLCD irrigated area, return flow was measured, and ditch loss was assumed to be 24% (Schwarz, 2002). For late September, some of the acres were not irrigated, thus, net crop ET was unknown. It was back-calculated using our diversion estimate of 15.55 = Crop ET + 7.63 + 3.73 which resulted in crop ET of 0.25 cm day^{-1} (0.10 inches day^{-1}), or half of all the irrigated acres not being irrigated.

As shown in **Figure 10-2**, the model reasonably represents river widths (RMSE=26 meters and RE=3.6%) and marginally reflects depths (RMSE=0.5 meter and RE=-22.2%). The simulation error is somewhat misleading, however, as many of the field measurements were made at bridges. Bridges are believed to be slightly deeper than normal which is apparent from review of bathymetric data [(Sadak, 2005), also shown]. Consequently, we feel the model reflects the general river character including: (1) the deep and slow moving water upstream of the Cartersville Diversion Dam (km 232.9-231.4), (2) the shallow and wide areas of the river near Miles City (km 150-100), (3) the deepening and widening in the lower reaches near Glendive (km 50-0), and (4) several of the rapids detailed in **Section 7.4** (km 128.9, 122.9, 95.4, 82.9). In this context, we feel we have an adequate representation of the river in Q2K.

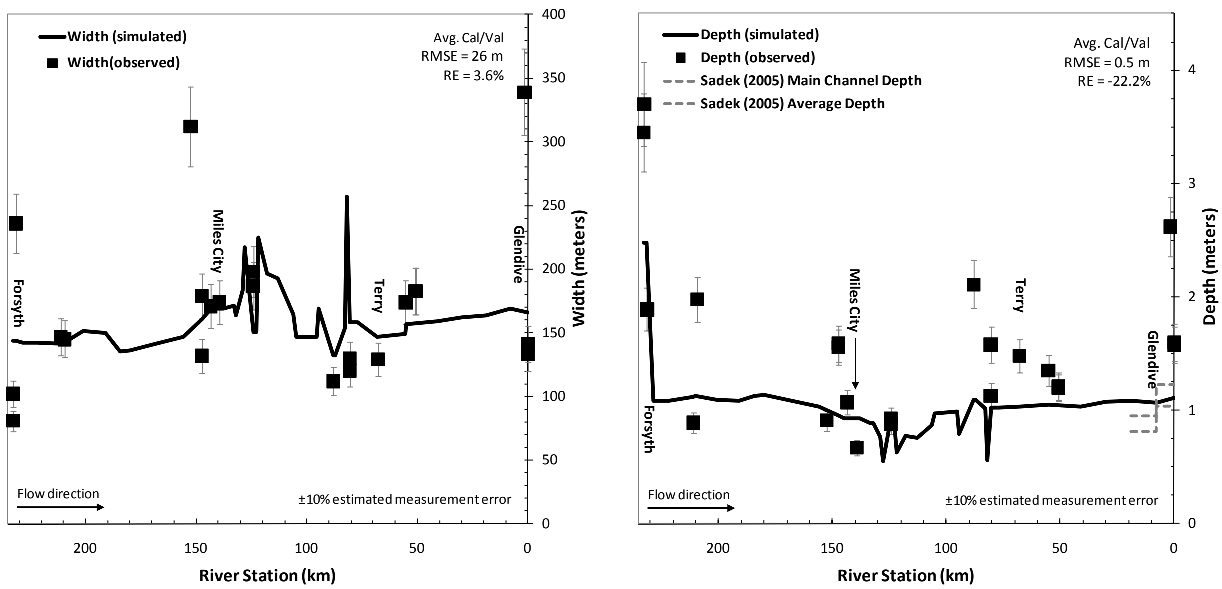

Figure 10-2. Simulated and observed river widths and depths for the Yellowstone River during 2007. (Left panel) Simulated river width with associated error statistics. (Right panel) Same but for river depth. Comparisons shown for the average of the calibration and validation periods as very little change occurred between the two periods (i.e., approximate 4 m change in width and 0.08 m change in depth).

A more reliable estimate of mass transport is system volume and residence time. In Q2K, residence time is determined as a function of streamflow and element volume. These are then summed to form the overall travel time for the river. The following data sources were used to make travel-time comparisons: (1) the travel-time calculator estimates detailed in McCarthy (2006), (2) field-measurements from a cooperative USGS/DEQ dye tracer study (McCarthy, 2009), and adjusted field measurements. Results of each are shown in **Figure 10-3** (Left panel).

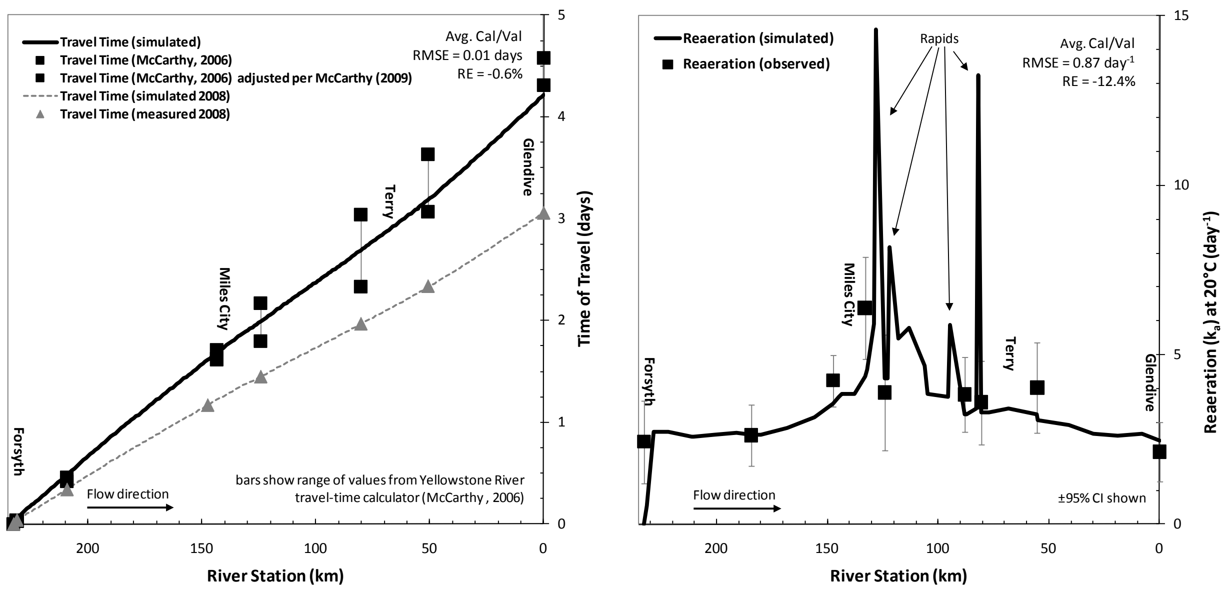

Figure 10-3. Verification of travel-times for the Yellowstone River during 2007.
(Left panel). Simulated and observed travel-time for 2007 and 2008 flow conditions. (Right panel) Simulated and observed reaeration for the Yellowstone River during 2007. It should be noted that reaeration from the Cartersville Diversion Dam is not shown in the plots. It is computed separately as a function of the height of the drop of the structure in the model. Error bars are based on the 95% confidence interval.

RMSE for the travel-time simulation was 0.01 days and RE= -0.6% based on a model run for flow conditions during 2008[57] (i.e., when the dye tracer study took place). Results from 2007 are bounded by the ranges reported in the previously referenced studies[58]. A tabular comparison of results is shown in **Table 10-2**. For all of the different years and flow conditions, there was very little difference between simulated and observed values.

Reaeration is a final plausible check on the model and is shown in **Figure 10-3** (right panel). It is computed as a function of depth and velocity in Q2K. Reaeration rates very closely approximate estimated field reaeration using the delta method (described in **Section 9.8**). As a result, the physical basis of the model appears sound. RMSE was 0.87 day^{-1} and RE=-12.4% and rates were higher in the wide shallow regions near Miles City (i.e., higher velocities) and lower elsewhere. The effect of the four rapids (mentioned previously) is also apparent.

[57] The 2008 simulation was based on the 2008 flow condition which was derived from the operational gages on the river and tributaries (mainstem sites and Tongue and Powder Rivers). Other information was not available. Consequently the effort focused on ensuring flows matched the USGS gages sites.
[58] This includes the McCarthy (2006) travel-time calculator which was based on the ratio of flood wave velocity to most probable velocity and then adjusted dye velocities according to McCarthy (2009) field tracer studies. The velocities in 2007 were slower making the travel-time estimates larger. These were as follows: Cartersville Diversion Dam =+24%, Rosebud Bridge = +8%, Keough Bridge (1902) = +5%, Kinsey Bridge = +17%, Calypso Bridge = +23%, Fallon Bridge = +15%, and Glendive Bell St. Bridge = +6%.

Table 10-2. Comparison of various travel-time estimates for Lower Yellowstone River.

	2007			2008	
	McCarthy (2004)	**McCarthy (2004) adjusted for dye**	**Modeled in Q2K[1]**	**McCarthy (2009)**	**Modeled in Q2K**
Flow ($m^3 s^{-1}$)	94-135	same	same	221-225	same
Travel-Time: Forsyth to Miles City (days)	1.7	1.6	1.6	1.2	1.2
Travel-Time: Forsyth to Glendive (days)	4.6	4.3	4.1	3.1	3.0

[1]Estimates shown as the average of the August and September simulations. These were within 0.1 days at Miles City and 0.3 days at the end of the project reach (at Glendive).

10.3 Water Temperature

Water temperature simulations are shown in **Figure 10-4** (Top panel). They represent the cumulative interaction of air, water, and sediment boundaries and their importance lies in the fact that they govern all kinetic processes in the model[59]. Modeled minimum, maximum, and mean temperatures show very good agreement over the August and September period with RMSE and RE of 0.24 and 0.38°C and 0.0 and -0.7% respectively. Diurnal temperatures at the 1902 Bridge and Kinsey Bridge FAS were also quite good (near ⅓ and ⅔ along the project reach) and are shown in **Figure 10-4** (bottom panel). In all cases, simulated temperatures were within the criteria specified in the QAPP (±5°C) and are satisfactory to DEQ for our model development purposes.

In general, consistent trends occur in the longitudinal temperature profile where water is cooler both in the upper and lower reaches of the river (from groundwater and climatic gradients) and then warms near Miles City (km 150-100). The only notable difference between these locations was widening and shallowing of the river and slight climatic variation and thus the change is primarily a physical occurrence. A change also occurred between the calibration and validation which was seasonally induced. The river was 5°C warmer in August than it was in September due to longer daylength and a warmer mean air temperature over the period of interest.

Changes in diurnal flux (maximum – minimum temperature) were not that different in either case. The daily range in both August and September was approximately 2-3°C which consisted of a minimum shortly after daybreak (around 8:00 a.m.), daily averages at both midday and midnight, and a nighttime maximum around 6:00 p.m. (**Figure 10-4**, Bottom panel). Overall, the two profiles are very consistent short of the shift in mean daily temperature.

[59] The Arrhenius equation (Chapra et al., 2008) is used to adjust all biogeochemical rate coefficients in the Q2K model.

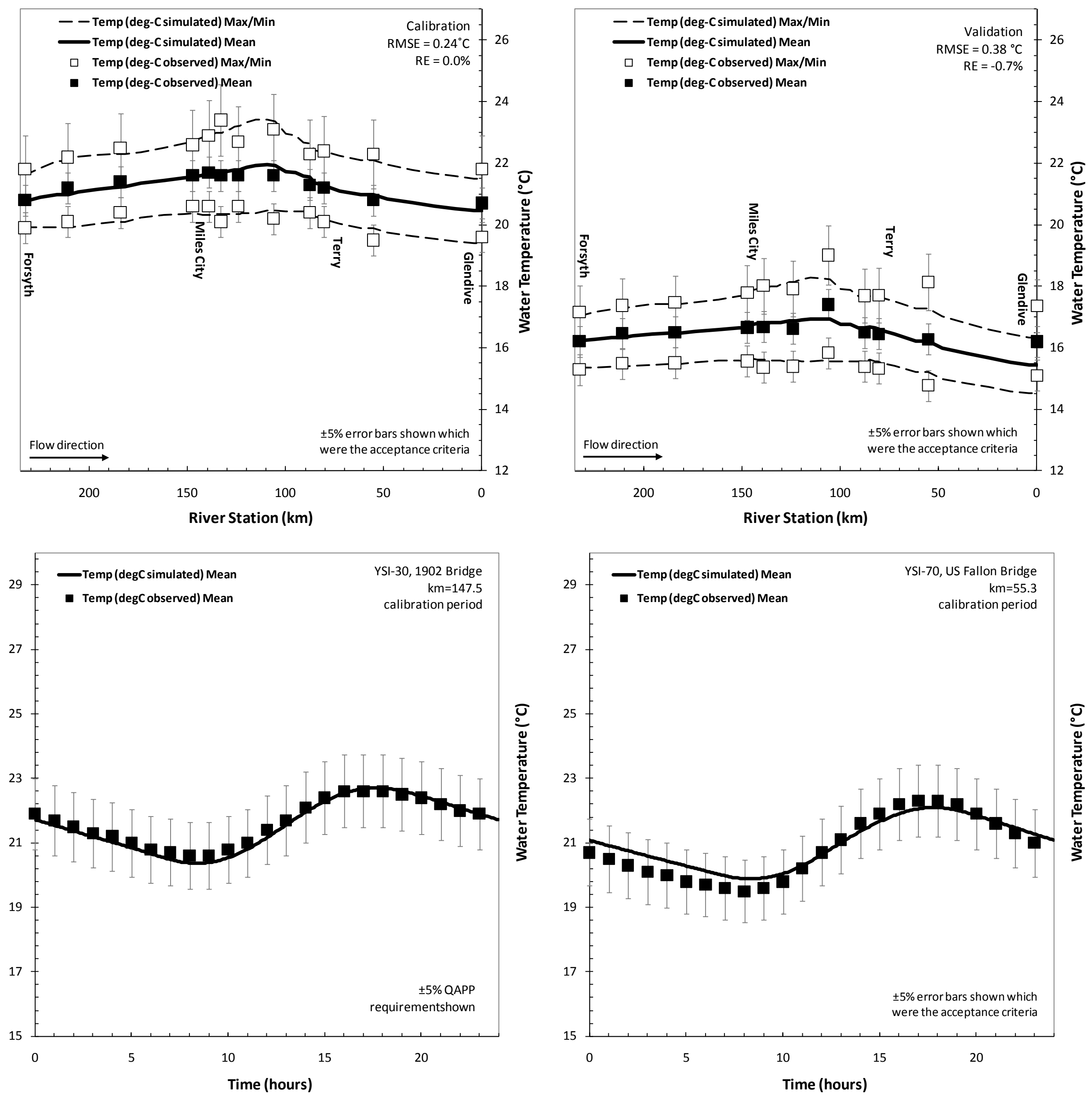

Figure 10-4. Water temperature simulation for the lower Yellowstone River during 2007.
(Top left/right panel) Simulated and observed water temperature for the August calibration and September validation periods. (Bottom left/right panel) Diurnal simulations for km 147.5 (1902 Bridge) and km 55.3 (upstream of O'Fallon Creek). Diurnal plots are from the calibration period.

10.4 Water Chemistry and Diurnal Water Quality Simulations

Water chemistry simulations represent the bulk of the work in model development and are of primary importance for the criteria development process. Output for the modeling has been grouped into functional categories so that results are better organized. Included are the following:

- Suspended particles, including total suspended solids (TSS), inorganic suspended solids (ISS), and detritus, excluding phytoplankton.
- Nutrients, both nitrogen (N) and phosphorus (P)

- Algae (both benthic algae and phytoplankton)
- Carbon [including pH, alkalinity, CBOD, and total organic carbon (TOC)]

Results are presented in the remaining sections[60].

10.4.1 Suspended particles

Suspended particles consist of both organic and inorganic materials in suspension and collectively form total suspended solids (TSS). TSS increases from external loads or resuspension and is lost via settling. The inorganic fraction of TSS is called inorganic suspended solids (ISS) is comprised of materials such as clays, sand, and silica that are derived from inorganic materials. The organic fraction includes both living and non-living material such as phytoplankton and detritus[61]. Materials that are combustible at 550°C in a muffle furnace are considered organic, while those that are not are inorganic. In the Yellowstone River, a large fraction of the suspended particles were inorganic (roughly 70-80%).

Model simulations of suspended particles are shown in **Figure 10-5** [reported as ash-free dry mass (AFDM), mg D L^{-1}]. From review of the simulations, particulate matter in the model is reasonably represented during both calibration and validation with RMSE and RE of 3.5 mg L^{-1} and 9.9% and 9.0 mg L^{-1} and -8.9% for TSS (each period respectively) and 2.2 mg L^{-1} and 3.3% and 9.2 mg L^{-1} and -19.3% for ISS. Hence the calibration performs better than the validation, but both are within the expected uncertainty limits of the field data making it acceptable to DEQ.

In review of the simulation, there is an apparent longitudinal trend in TSS and ISS with relatively consistent concentrations for the first 150 km (km 232.9-80) and then noticeable increases thereafter. Much of this is coincident with the Powder River, but is not directly ascribed to it as its flow was minimal at the time of monitoring. Since the increase could not be linked to other inflows (e.g., other tributaries, irrigation return flows), the contribution was believed to originate directly from within from the Yellowstone River itself (autochthonous). The source is likely previously deposited material from the Powder River that is now in intermittent resuspension from shear stress in the Yellowstone River. Approximately 130 tonnes day^{-1} of ISS load was needed to make up the difference. A line accretion was added to the model to reflect this increase[62].

[60] Throughout the water chemistry and diurnal water quality simulation, attempts are made to characterize measurement uncertainty of our observed data. This is not meant to take away from or add to the apparent reliability of the model. Rather, it is to show potential ranges for the purpose of assessing model usability. These were taken directly from our monitoring instrumentation or from Harmel et al., (2006) as described in **Section 7.7**. The typical collection scenario error was used.

[61] Detritus consists of dead and decaying (non-living) organic matter. It can be lost from dissolution and increase from algal death. To separate detritus from phytoplankton mass in OSS measurements, the corrections detailed in **Section 9.4** were used.

[62] The Powder River actually enters at river km 87.3. However, to be consistent with the water balance (which was completed to the Terry gage), the diffuse accretion term was extended slightly upstream.

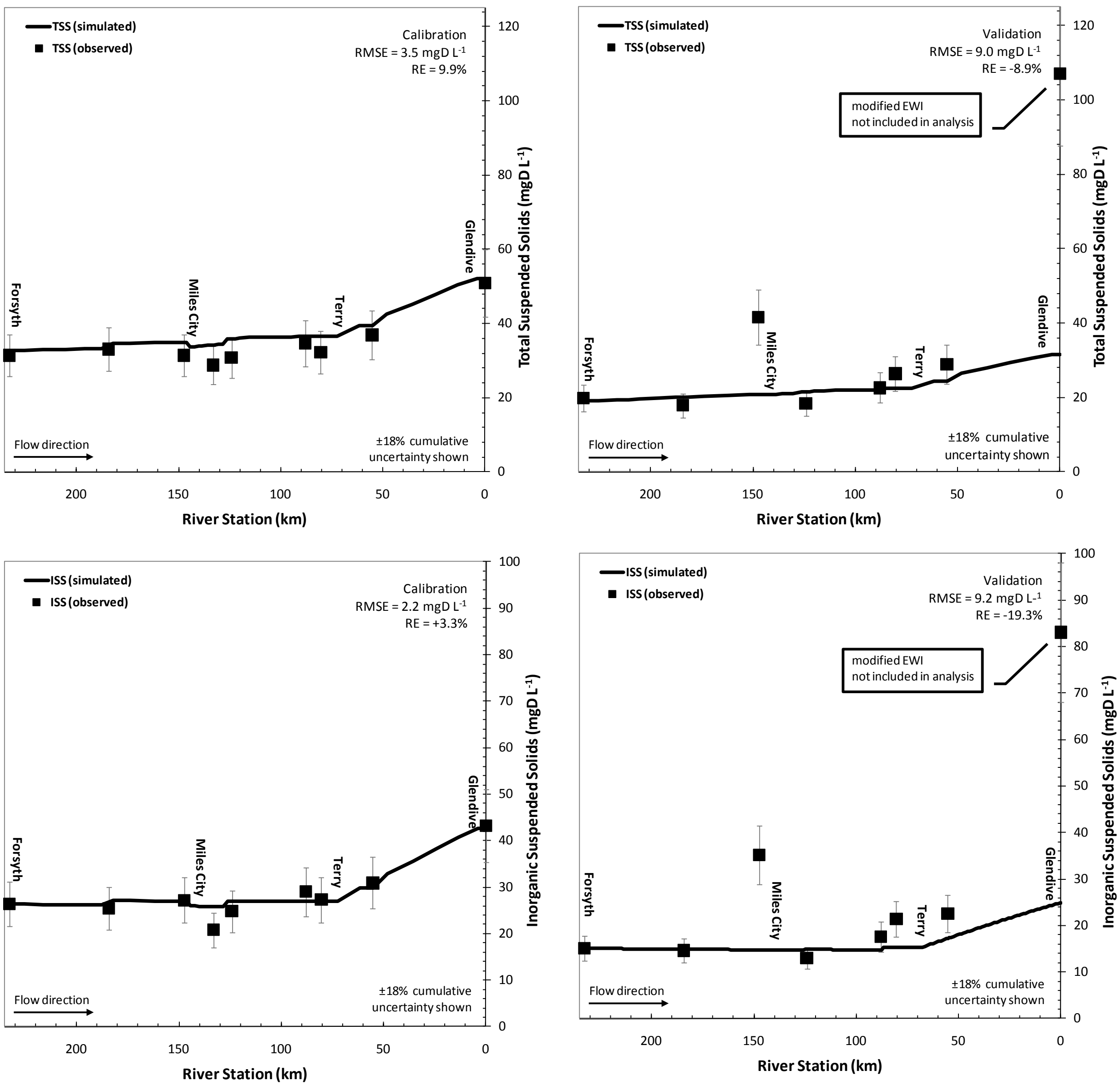

Figure 10-5. TSS and ISS simulations for the lower Yellowstone River during 2007.
(Top left/right panel) Simulated and observed TSS values for the August and September evaluation period. (Bottom left/right panel) Same but for ISS. The ±18% cumulative uncertainty is based on Harmel, et al., (2006).

Detritus is another suspended component and follows a pattern different than TSS/ISS. For example, it increases greatly in the first 150 km due to greater algal biomass recycling whereas it declines in the lower reaches due to settling and reductions in productivity (**Figure 10-6**). RMSE and RE for detritus were 0.9 and 1.5 mg L^{-1} and 4.1 and 22% for calibration and validation, which are within the uncertainty limits.

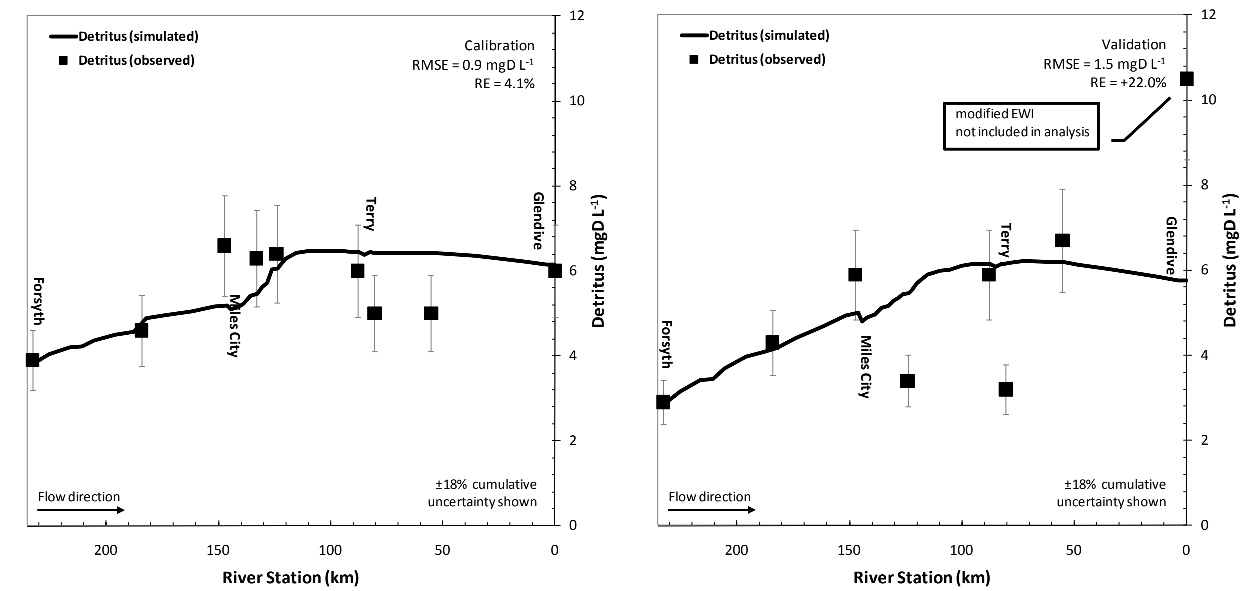

Figure 10-6. Detritus simulation for the lower Yellowstone River during 2007.
(Left panel) Simulated and observed detrital values for the August calibration period. (Right panel) Same but for the validation. The ±18% cumulative uncertainty is from Harmel, et al., (2006) values for TSS.

10.4.2 Nutrients

The nutrient results, both nitrogen and phosphorus, are included in this section.

10.4.2.1 Nitrogen

All forms of nitrogen are reflected in the total nitrogen (TN) measurement which includes soluble inorganic N (NO_2^- + NO_3^- + NH_4^+), organic N (OrgN), and intracellular N in phytoplankton, which are sequentially linked through death → hydrolysis → nitrification → denitrification reactions. NO_2^- is not modeled. TN and OrgN increase due to plant death and excretion, and are lost (converted) due to hydrolysis and settling. OrgN hydrolysis produces ammonia N (NH_4^+) which is lost due to nitrification (i.e, increases NO_3^-), and both NO_3^- and NH_4^+ are lost due to plant uptake, while NO_3^- can also decrease from denitrification.

TN and OrgN simulations are shown in **Figure 10-7**. Ambient values range from 400-500 µg L^{-1} and generally decrease in the downstream direction. OrgN was roughly 75% of the total N contribution (i.e., 300-400 µg L^{-1}) and RMSE and RE for the calibration and validation for each constituent were 37 and 67 µg L^{-1} and 7.3 and 13.8% and 22 and 79 µg L^{-1} and 0.6 and 23.2% for the calibration and validation respectively. TN was lowest in the middle reaches (near km 150) due to low soluble nutrients, while OrgN was highest at this same location (due to higher biomasses, productivity, and algal death). Our simulations were very close to the suggested measurement error in Harmel et al., (2006) and TN showed a small disparity in the downstream direction. This is an artifact of inflation of internal N within phytoplankton which is believed to occur at least partially because of the uncertainty in the soluble N load contributions from irrigation waste-drains (meaning we could have overestimated these values). Case in point, the model actually performed better without them[63].

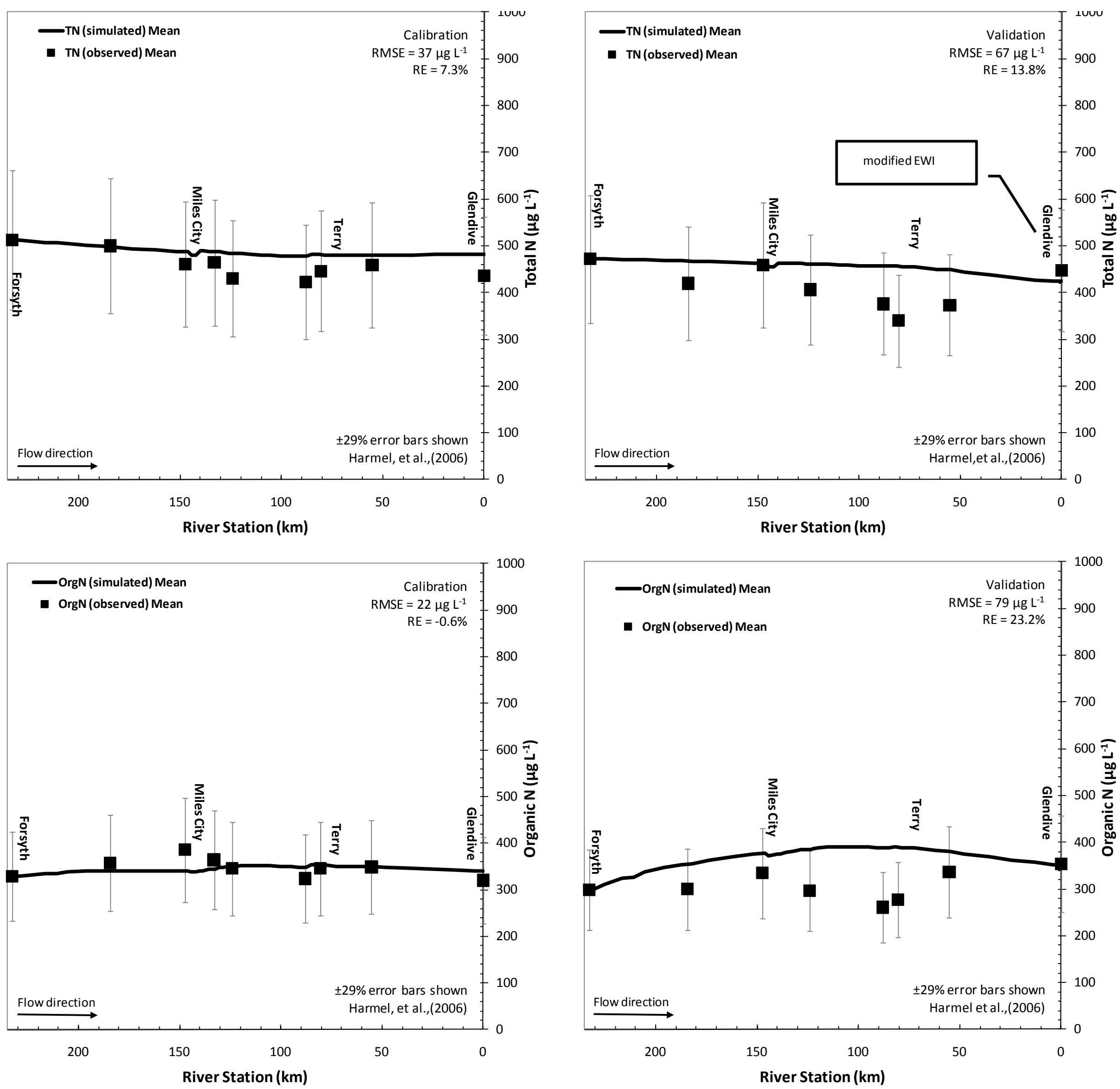

Figure 10-7. Total and organic N simulations for the Yellowstone River during 2007.
(Top left/right panel). Simulated TN for the August and September 2007 calibration and validation periods. (Bottom left/right panel). Same but for OrgN. The system appears to perhaps behave differently than the model suggests during the validation period.

From review of the TN and OrgN simulation, it appears as if there is difficulty in model validation. This is evidenced by greater RMSE and RE, as well as the visual departure of the model from observed values. It

[63] Estimated waste-drain loads were calculated as identified in **Section 7.0** but were calibrated down from our original estimates due to the fact that they increased the nutrient load beyond expected levels.

will be shown later that this is related to a shift in river trophic condition, which becomes a recurring theme throughout the remaining discussion. The reasons for this change will be expounded upon more in **Section 11.0**.

Dissolved forms of nitrogen are shown in **Figure 10-8**. Nitrate (actually NO_2^- +NO_3^-) showed reasonable agreement in the calibration but not the validation. RMSE's for NO_3^- were 9 and 29 µg L^{-1} (calibration and validation respectively), and relative errors were 215 and -63.2%. The magnitude of the RE is misleading due to the low concentrations in the river (e.g., RMSE was only 9 µg L^{-1}). We chose to focus our calibration on NO_3^- in the upper parts of the river, where values were well above detection, however, it should be noted that minor reductions in the nitrification rate perhaps would improve the calibration of NO_3^- in the lower river (thereby increasing NH_4-N and). In any regard, nitrate uptake is very rapid and high concentrations near the upper study limit were depleted to non-detect levels near the midpoint of the reach (km 150). Ammonia concentrations were quite low in 2007 (ranging from non-detect to 20 µg L^{-1}) and were characterized by a slight decline in the most productive reaches of the river near Miles City (km 150) and higher concentrations elsewhere. RMSE and RE for NH_4^+ were 8 and 17 µg L^{-1} and -36.4 and -47.8% for the calibration and validation. These were reasonable given the low concentrations found in the river.

Again, there were problems with the validation. Primarily, this was related to overestimation of soluble N uptake which slowed greatly between August and September (as suggested by the change in the NO_3^- concentration longitudinal curve[64]). It is important to distinguish a change in uptake versus a change in nutrient supply. That is, the load to the river did not change between the two periods, just the uptake capacity. Since uptake is biologically mediated, we completed experimental perturbation of model rate coefficients to characterize the reason for this change. Accordingly, we found the shift in river productivity was likely related to a change in benthic algae rather than other model rate coefficients (i.e., all others remained consistent during the two periods). A reduction in growth rate of 50% was necessary to pattern the change in uptake and other diurnal indicators such as DO and pH which will be described in subsequent sections.

Given the difficulty described in the prior paragraph, an alternative parameter set was proposed (for validation) which reflects the change in benthic productivity. Mechanisms could be attributed to a number of things including physical changes in the growth rate due to changes in river taxa, changes in algal light use efficiency, or changes in growth rate with temperature outside that described by Arrhenius. We have chosen to show this as a separate model run entitled "alternative growth rate" in all subsequent plots. Please note that this has not been done to alter the validation statistics, but to better illustrate an understanding of relational processes in the model. We address and elaborate on this validation deficiency in later sections.

[64] The model simulation actually showed an increase in N uptake in September which was a function of more light (i.e., less turbidity). This was slightly offset by the differences in water temperature 21°C in August compared to 16°C in September).

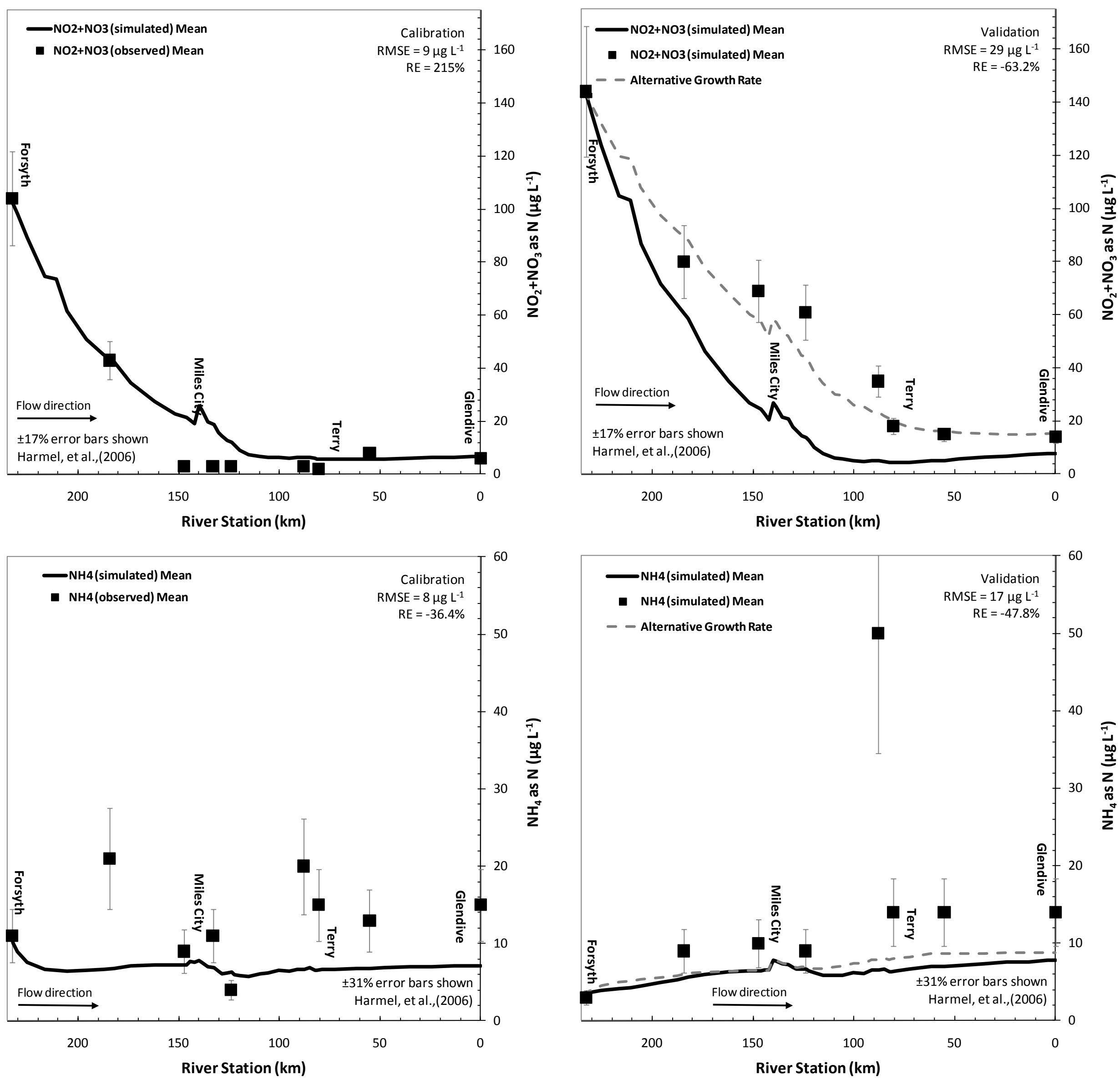

Figure 10-8. Dissolved nitrogen simulations for the Yellowstone River during 2007.
(Top left/right panel) Nitrate simulations for the August calibration and September period. (Bottom left/right panel) Same but for ammonium. The alternative growth rate illustrates the change in benthic algal growth rate to bring the model into agreement with the observed data.

10.4.2.2 Phosphorus

Similar to TN, TP represents all P in the system including organic and inorganic forms. Organic P (OrgP) increases due to plant death and excretion, and is lost via hydrolysis and settling. Inorganic P (SRP) increases from OrgP hydrolysis and excretion, and is lost through plant uptake. For the purpose of our work, SRP is considered 100% bioavailable. This assumption seems reasonable, but has been questioned by some (Li and Brett, unpublished 2011).

Simulations of TP and OrgP for 2007 are shown in **Figure 10-9**. Overall there is fairly good agreement as RMSE for the calibration and validation were 9 and 6 µg L^{-1} and 8 and 5 µg L^{-1} for TP and OrgP respectively, while RE was -11.7 and 8.9% and -11.5 and 11.9%. A majority of the TP was in organic form (≈70%) and was closely related to ISS (r^2=0.82). A large shift in both TP and OrgP occurred downstream of the Powder River (km 90) which is related to the concomitant increase in ISS. We used the TSS-TP relationship presented by Miller et al., (2004) to estimate this increase in the model.

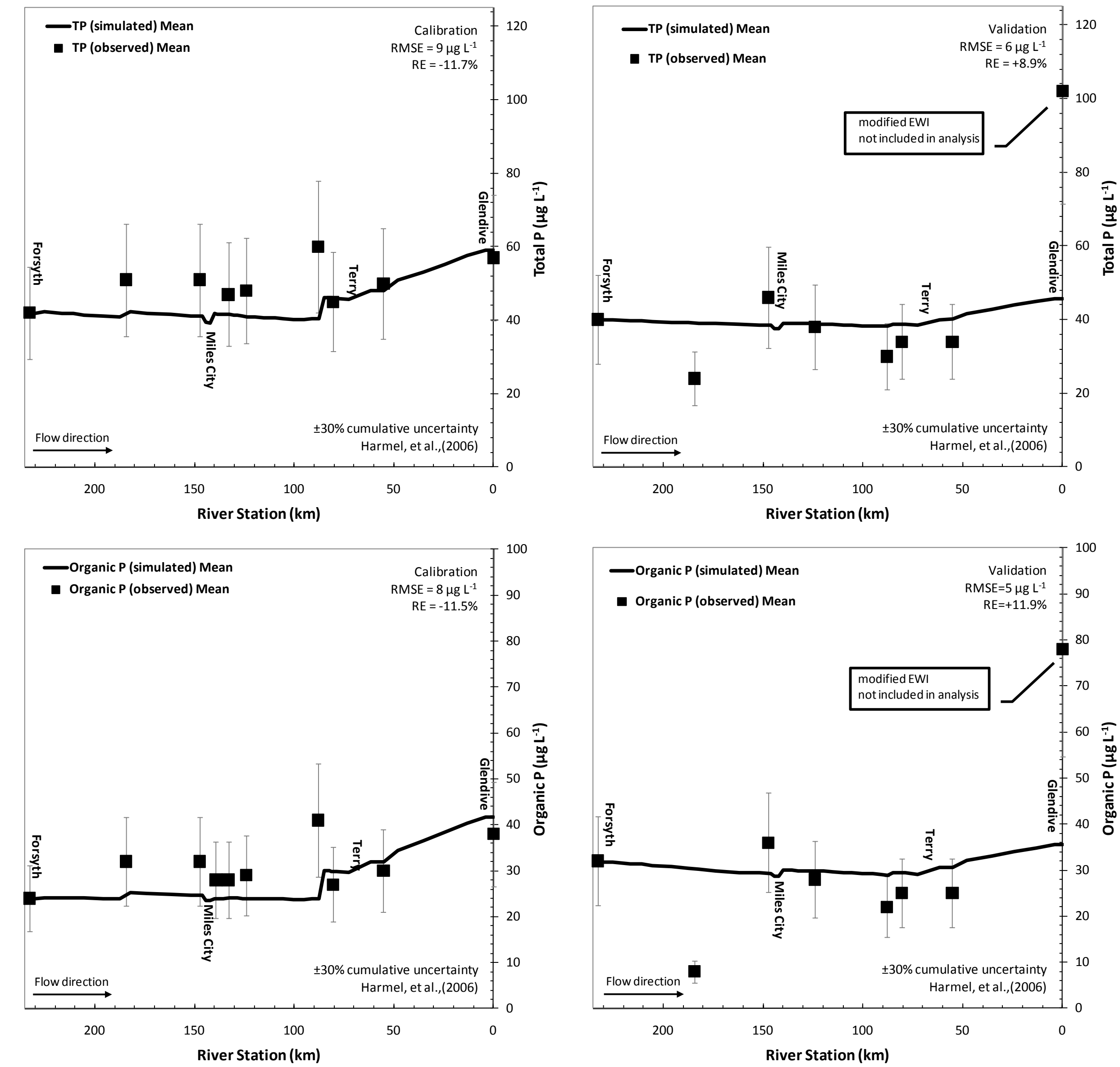

Figure 10-9. Total and organic phosphorus simulations for the Yellowstone River during 2007. (Top left/right panel) Simulated TP for the August and September 2007 calibration and validation periods. (Bottom left/right panel) Same but for OrgP. The problems shown in previous validation plots (i.e., for nitrogen) are less apparent for P due to N:P stoichiometric ratios.

Soluble reactive phosphorus (SRP) fits were also completed, but should only be used anecdotally as values were below the laboratory reporting limit of 4 µg L^{-1}. Comparisons are made with estimated quantitative values (flagged by DEQ, not provided in STORET) which ranged from 2.6-3.6 µg L^{-1}. All were near the threshold of analytical noise (i.e., the actual method detection limit) and were also affected by poor laboratory QA blanks (false detections of 2.1 µg L^{-1}, standard deviation of 0.3 µg L^{-1}, n=3). Consequently, there is uncertainty in the observations. However, it was still of use in calibrating the model (only after due consideration) given that structure in the data is apparent. Simulations are shown in **Figure 10-10** and primary drivers of SRP on the Yellowstone River were found to be the Forsyth and Miles City WWTP as evidenced by the slight increase at each location. Statistical model efficiencies were not determined due to the reasons mentioned previously.

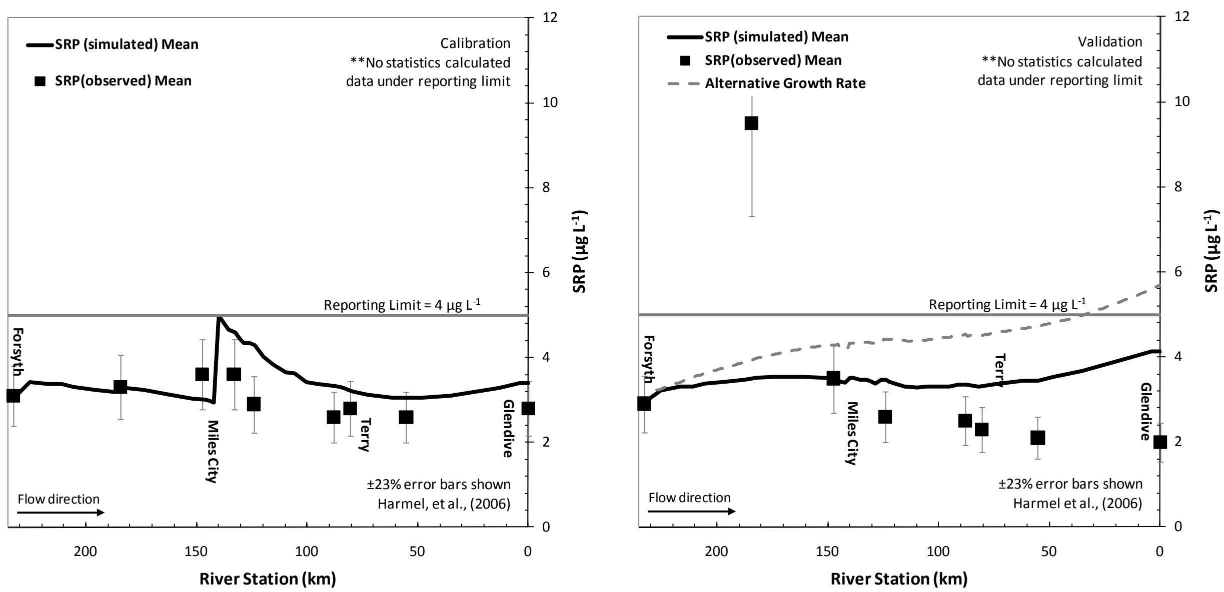

Figure 10-10. Soluble phosphorus simulations for the Yellowstone River during 2007.
(Left panel) Simulated SRP for the August calibration period. (Right panel) Same but for the September validation. Very little SRP was discharged by Miles City in September of 2007. No statistics were computed for SRP given the concerns described previously.

10.4.2.3 Nutrient Limitation During 2007

Nutrient limitation can be calculated according to Droop (1973) (**Equation 10-1**) for both the N and P where ϕ_N is the nutrient limitation factor, and q_{oN}, q_{oP}, q_N, and q_P are the subsistence quotas and cell quotas for nitrogen and phosphorus respectively.

(Equation 10-1)

$$\phi_N = \min\left[1 - \frac{q_{oN}}{q_N}, 1 - \frac{q_{oP}}{q_P}\right]$$

Limitation is determined according to a single limiting nutrient (Liebig's law of the minimum), where the most limiting nutrient in supply attenuates algal growth. The growth attenuation factor ranges from 0-1 and is multiplied by the maximum unlimited growth rate to yield the net specific growth rate. A nutrient limitation factor of 0 would be indicative of no growth, while a factor of 1 would yield maximum growth.

Using such an approach, limitation during 2007 varies along the longitudinal extent of the river and differs between benthic algae and phytoplankton (**Figure 10-11**). In the case of benthic algae, P-limitation occurs over approximately half of the study reach (km 232.9-150) until a switch to N-limitation occurs near Miles City (from WWTP phosphorus additions). The river then ultimately goes on to co-limitation in the lower reaches. The mechanics near Miles City are most complicated and are not well understood by DEQ. Phytoplankton are more stable because they are not tied to site-specific nutrient conditions and thus their internal nutrient pools are less variable (because they advect through the water column and experience longitudinal variation in nutrient conditions). In 2007, phytoplankton were N-limited according to our seston stoichiometry measurements, e.g., 4.7:1 N to P mass ratio, and stayed that way throughout the project reach. A slight shift occurred near the upper end of the study reach from the high soluble N (NO_3-) levels, but in general, phytoplankton were unresponsive to site-specific environmental conditions and more responsive to the overall trend of N or P in the river. This is largely believed to occur from luxury uptake and the ability regulate internal cell quotas.

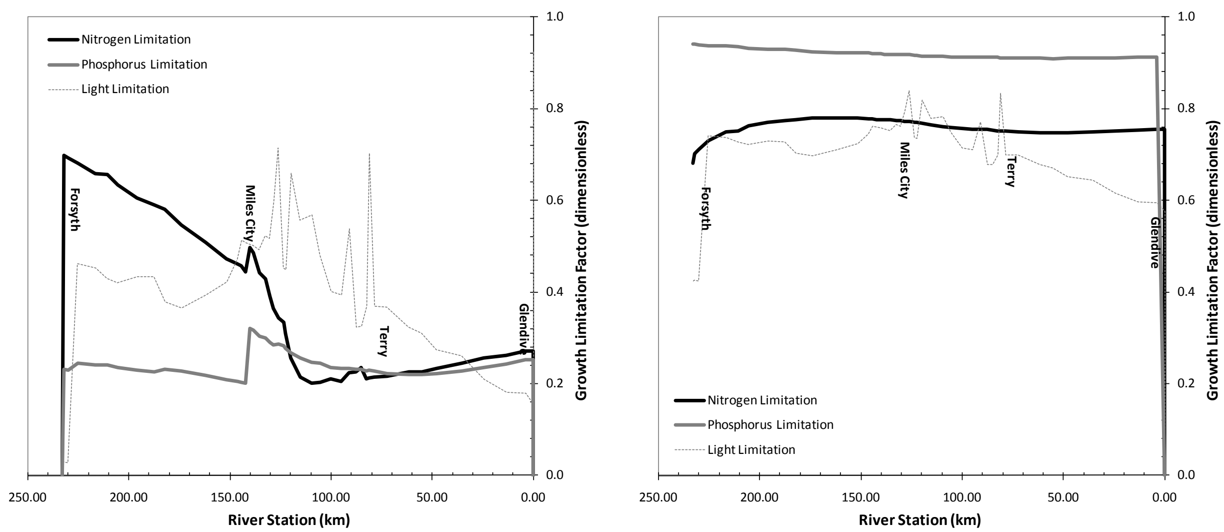

Figure 10-11. Nutrient and light limitation factors for the Yellowstone River during 2007.
(Left panel) Benthic algae nutrient and light limitation for the Yellowstone River. The lowest ordinate reflects the most limiting factor in any given case. (Right panel) Same but for phytoplankton. Benthic algae are P limited from Forsyth to just downstream of Miles City (km 125) and then switch to N limitation for a short period. The river is then essentially co-limited thereafter. This is consistent with Charles and Christie (2011) who indicate a high percentage of N-fixing diatoms occur at km 125. Phytoplankton enter the reach in N-limitation and stay that way throughout. Light limitation is also shown as discussed in subsequent sections. Bottom algae are least light limited near Miles City (km 150) and are strongly light limited in the lower river. There is a consistent decline in available surface light throughout the river. The effect is less pronounced on phytoplankton as they are able to re-circulate through the water column.

10.4.3 Algae

10.4.3.1 Benthic Algae

Benthic algae are of primary importance in the Yellowstone River as evidenced from our sensitivity analysis and from DO and benthic biomass relationships presented in Charles and Christie (2011). Consequently, we did considerable work understanding their importance. To characterize the lateral distribution of algae in the river, we collected 11 discrete samples at each river transect in both the wadeable and non-wadeable regions. We then reduced these data into a single cross-sectional biomass average[65] for Q2K modeling. For AT2K analysis, the original discrete data were used.

Collections were also made to identify benthic algal taxa using standard DEQ protocols. In brief: at each point, after having collected a benthic Chl*a* sample, we scraped/scrubbed material from the same river substrate and composited it with similar material from the remaining transect points. The composite sample was preserved with formalin (2-3 % final concentration) and later analyzed for soft-bodied and diatom algae species including density and taxa identification (DEQ, 2011a).

Benthic algae reflect the net balance between photosynthesis and respiration and death. Model simulations of algal biomass (mg Chl*a* m^{-2}) for the August and September calibration and validation periods are shown in **Figure 10-12**. RMSE was 4 and 23 mg Chl*a* m^{-2}, and RE 10.3 and 86.7% each period respectively. Again there were problems with the validation. Consequently, we met our QAPP criteria of ±20% for the calibration (not for the validation), but only when hydraulic depth in the model was adjusted to the exact depth of the field transect[66]. This illustrates the importance of depth on site-specific algal measurements and is just one of the many difficulties that one could encounter when modeling benthic algae in large rivers.

The most productive region of the river was found to be near Miles City (km 150) where the river is wide, shallow, and nutrient replete due to soluble nutrient additions from the Miles City WWTP. Spatially, algal biomasses tended to be higher in the upper study reach (km 232.9-80.7) than the lower river (km 80.7-0) primarily because of differences in light. This translates into an induced shift in algal dominance from benthic alga to phytoplankton as evidenced by the continued downstream decline of bottom biomasses and increase in phytoplankton (see **Figure 10-12** and **Figure 10-16** for further support of this statement). It should also be noted that a number of elevated algal peaks occur at rapids (e.g., shallow and wide). It is unclear if such biomasses would actually occur. These locations perhaps are limited by high shear velocities but were included regardless of the case.

[65] Areal weighting procedures were used to determine equivalent cross-sectional average. In other words, the 11 biomass measurements were averaged based on equivalent area between measurements to provide a mean cross-sectional average.

[66] In Q2K, depth is modeled as the mean over an entire reach to meet the expected productivity response for that segment (e.g., DO, pH, etc.). However, our periphyton measurements reflect an actual site measurement (and depth), which may vary greatly from the overall average. For example, at station 150 km, hydraulic depth in the model was 0.96 vs. 1.56 meters in the field (0.6 meter difference). A similar case was noted at Pirogue Island (km 135) and O'Fallon Bridge (km 55). Thus to make a representative comparison of biomass, depth was adjusted as shown in the plots.

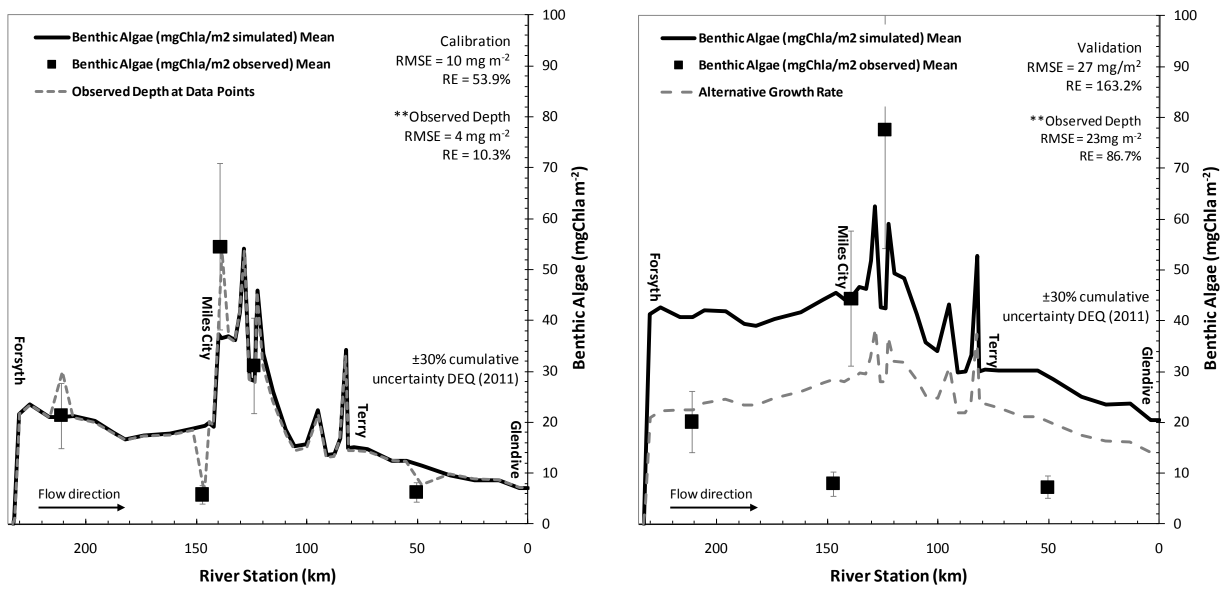

Figure 10-12. Simulated benthic algae Chl*a* for the Yellowstone River during 2007.
(Left panel) Simulated and observed benthic algae Chl*a* for the August 2007 calibration period. (Right panel) Same but for the September 2007 validation.

As mentioned previously, the validation was problematic and algal biomasses were overestimated over the entire length of the river. This suggests an overall decline in river trophy between August and September 2007. The use of the alternative growth rate improved the simulation slightly. However, over-simulations were still higher than desired. As a result, we believe that the biomasses during September were more a function of residual growth in August than as a result of the river's biogeochemical conditions in September. Perhaps senescence had begun or a shift in algal taxa occurred. Further discussion regarding each hypothesis is provided in **Section 11.1**.

10.4.3.2 Benthic Algae – Lateral Simulation

The lateral distribution of algae was evaluated using AT2K. AT2K functionally represents the same light and nutrient processes as Q2K, with the exception that algal growth is evaluated laterally as opposed to longitudinally. Simulations were carried out using the observed water quality data from each site[67] and ATK was calibrated in a joint fashion with Q2K. Similar to the longitudinal discussion, modeled lateral benthic algae distributions were relatively good for the calibration and poor for the validation (**Figure 10-13**). The average RMSE and RE (across the 5 cross-sections) were 22 and 35 mg Chl*a* m^{-2} and 51.9% and 98.4% respectively each period. Individual cross-section RMSE ranged from 8 to 48 mg Chl*a* m^{-2} and RE from -41.9 to 189%. By using the alternative growth rate (as detailed previously), the model yielded slightly better results. RMSE was 24 mg Chl*a* m^{-2} and RE -0.8%. Algal biomasses were still over-predicted, but captured the underlying trend of variation with respect to depth. Near-shore regions had highest biomasses (50-100 mg Chl*a* m^{-2}) while deeper areas (e.g., 1-2.5 meters) contributed very little, typically

[67] This was done so that representative water chemistry/optics were applied to each transect. Also, one site location (Far West FAS) did not have any water quality data. In this instance we used simulated values from Q2K. In all locations, simulated diurnal water temperatures were used.

less than 10 mg Chl*a* m^{-2}. Hence, we can conclude that the shallow regions of large rivers are of primary importance in eutrophication management.

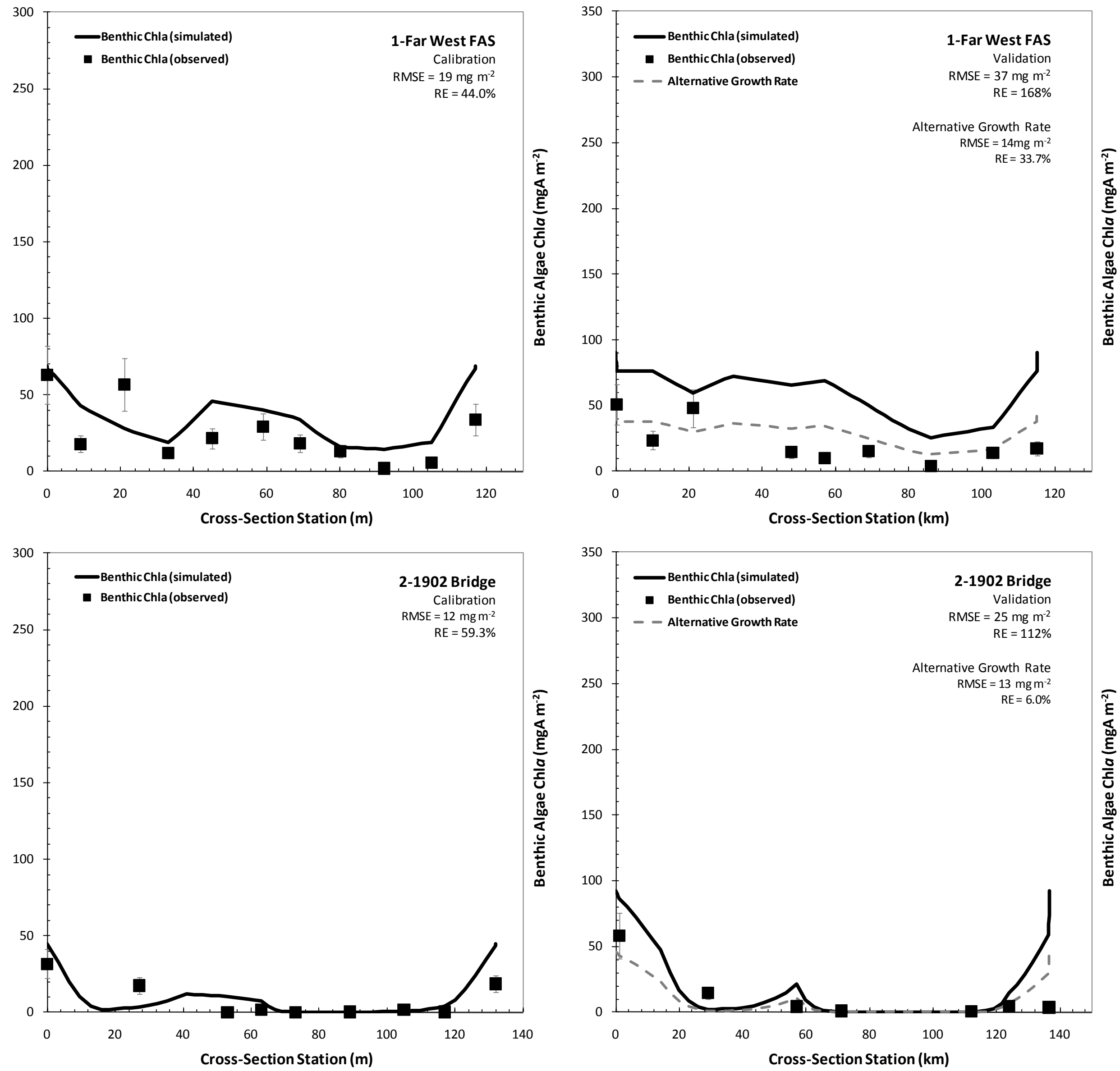

Figure 10-13. AT2K simulations of lateral algal distribution in Yellowstone River.
(Left panel) Simulated and observed values for each of the transects evaluated in August 2007. (Right panel) Same but for the validation. The alternative growth rate reflects the benthic algae growth rate determined previously to meet the productivity response of the river (see previous discussions). Field and lab variability estimated to be ±30% (DEQ, 2011).

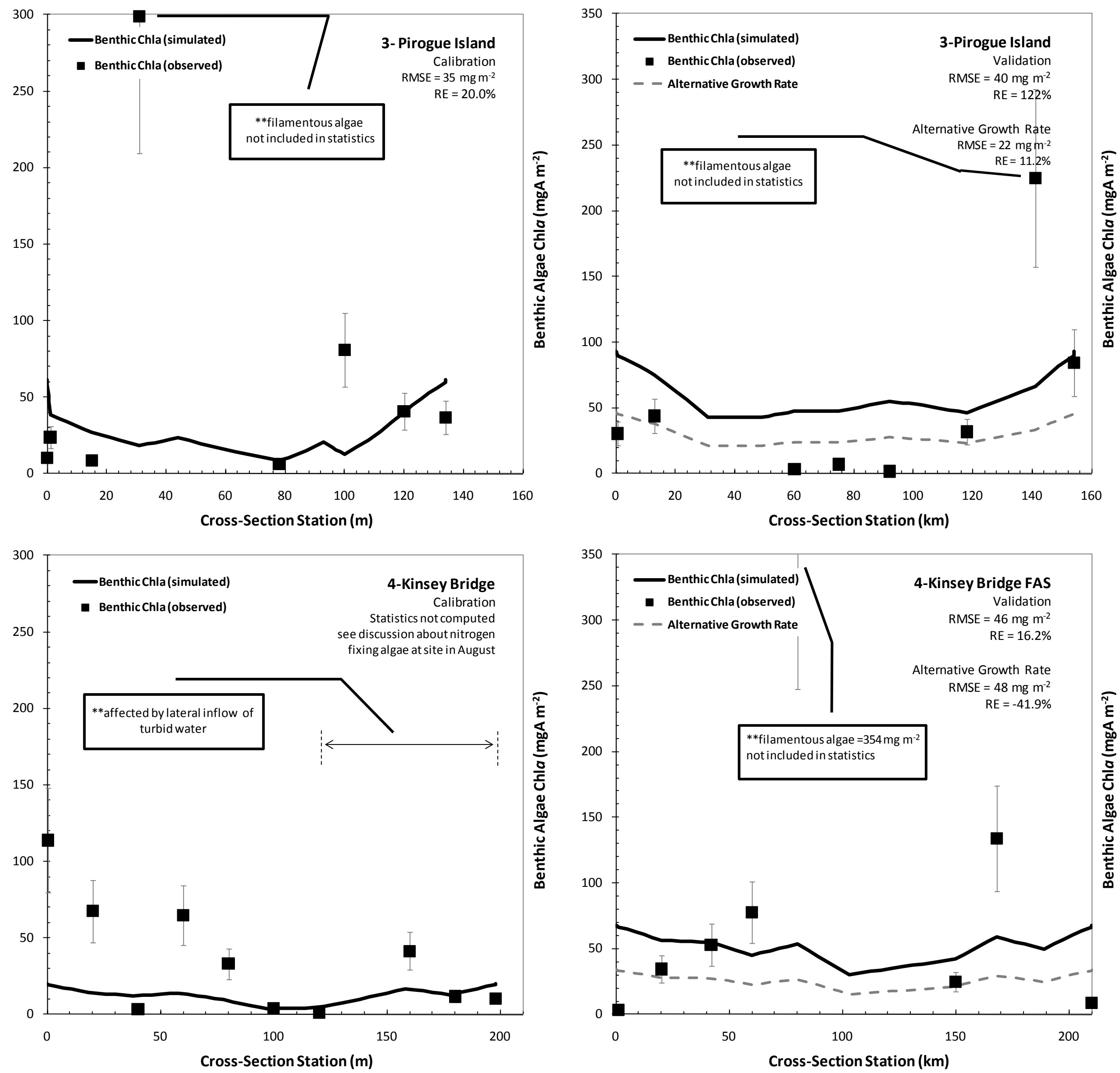

Figure 10-13 (cont.). AT2K simulations of lateral algal distribution in Yellowstone River.
(Left panel) Simulated and observed values for each of the transects evaluated in August 2007. (Right panel) Same but for the validation. The alternative growth rate reflects the benthic algae growth rate determined previously to meet the productivity response of the river (see previous discussions). Field and lab variability estimated to be ±30% (DEQ, 2011).

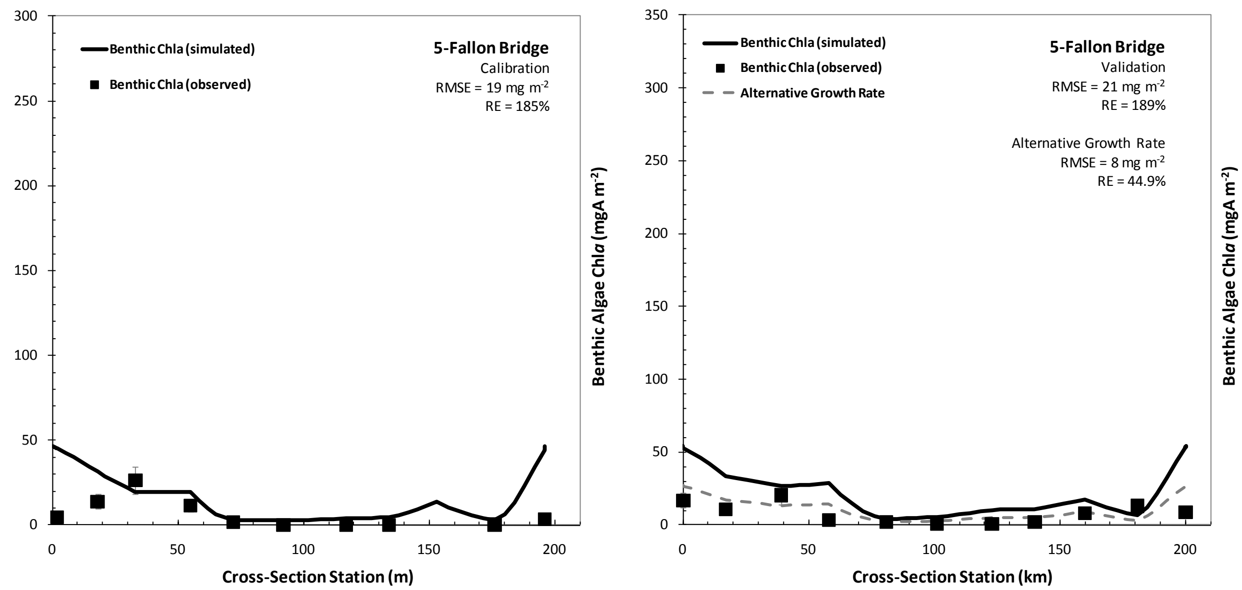

Figure 10-13 (cont.). AT2K simulations of lateral algal distribution in Yellowstone River
(Left panel) Simulated and observed values for each of the transects evaluated in August 2007. (Right panel) Same but for the validation. The alternative growth rate reflects the benthic algae growth rate determined previously to meet the productivity response of the river (see previous discussions). Field and lab variability estimated to be ±30% (DEQ, 2011).

It should be noted that in **Figure 10-13,** AT2K does not adequately simulate Chl*a* levels when the Chl*a* measurement was unusually high, especially when samples were dominated by long filamentous algae. In these places, long filamentous streamers of *Cladophora* spp. were present, which were sampled by the hoop method[68] (Freeman, 1986; Watson and Gestring, 1996). Such measures consider areal biomass that extends up into the water column which results in biomasses several times greater than periphyton. In contrast, diatom algae (which were most prevalent in this reach of Yellowstone River) consist of a 0.5-5 mm thick film on rocks that are sampled by scraping a small template of known substrate area.

As a consequence, long streamers of *Cladophora* in isolated locations in the Yellowstone River present a difficult problem to model. Q2K is limited to a single state-variable for simulating benthic growth in one-dimension (i.e., longitudinally) which is normalized to a 2-dimensional space (areal units, mg Chl*a* m^{-2}). Diatoms and short filaments essentially fit this scenario. In contrast, long *Cladophora* streamers protrude up into the water column in 3-dimensions (the volumetric space of the water) attaching to rocks via small holdfasts and growing out into the boundary layer under optimal conditions (Dodds and Gutter, 1992). As such, *Cladophora* streamers can develop considerably higher levels of biomass than algae growing strictly attached to the substrate because their space limitation term is less limited. Neither Q2K nor AT2K can currently address this condition.

[68] Briefly, a metal hoop with interior area of 710 cm^2 is placed over the river bottom and only the algae streamers (or segments thereof) within the confines of the hoop are collected. Only 3% of the 2007 benthic algae samples on the Yellowstone River were collected via the hoop. This was because the vast majority of sampling locations encountered (97%) were dominated by diatom algae or mixes of diatoms and very short filaments of green algae (including short *Cladophora*).

In critique of our newly developed AT2K model, good agreement is seen at most locations in our cross-sections (at least for the calibration). Since the vast majority of algal growth encountered during 2007 field sampling (97%) was closely attached to the bottom (i.e., diatom-like), we believe the model is suitable to represent the typical diatom-like conditions that would be observed in the lower river at low-flow. One exception would be the Kinsey Bridge FAS, which was much more productive than nutrient levels would suggest. According to Charles and Christie (2011), nitrogen fixing *Epithemia sorex*[69] were prevalent at this site and made up nearly 50% of the overall periphyton community (**Figure 10-14**). Frustules of *Epithemia sorex* contain nitrogen-fixing endosymbiotic cyanobacteria which enable them to become abundant in N-deplete microhabitats, even those with low N/P ratios. It is not surprising then that even at low N levels, algal biomass was still sufficiently high. In this instance, N-fixation likely provided the necessary N for algal growth. A very large percentage of N-fixers were observed at this same site in August of 2000 (Peterson and Porter, 2002). The Q2K model does not include the N-fixation process, nutrient exchange from epiphytic diatoms with cyanobacterial endosymbionts, or mat self-sustainment processes.

To put the final AT2K simulation reliability into context, data from all cross-sections (excluding the validation) were compiled and plotted on the 1:1 line (**Figure 10-15**, Left panel). By doing this, we find that simulations tend to roughly be within ±20 mg Chl*a* m^{-2} (Kinsey FAS excluded), despite notable dispersion in the data. This gives an estimate of model reliability, i.e., ±20 mg Chl*a* m^{-2} (the RMSE) which generally tends to over simulate lower biomasses and under simulate higher ones. The qualification of this error is based on the assumption that the error is attributed to the model, not field data. However in all reality it could be either, as previous work by DEQ has shown that the variability in Chl*a* averages due to field variation can be as high as ±30% (DEQ, 2011b). Hence error could just as easily be attributed to sampling noise as opposed to model uncertainty.

To assess which one of these it might be, cumulative frequency plots were constructed (again for the calibration only) as shown in **Figure 10-15** (Right panel). Model simulations appear to represent the data reasonably well, with the exception of less frequent higher biomasses. This perhaps suggests that the model has difficulty in predicting very high biomasses even for diatoms (filamentous algae excluded from this analysis), or that field observations were spurious and not entirely representative. An appropriate margin of safety perhaps should be included to address quantification limits, which is addressed in later sections in discussions regarding the nutrient criteria.

In lieu of previous discussions, we feel that the lateral benthic algae simulations in AT2K are sufficient to answer the questions in which we are interested, are in line with expectations from the literature (see NSTEPS comments in **Appendix D**), and are useful for regulatory management provided that the assumptions and limitations of the model are taken in proper context. Specifically the model allows us the ability to gain better information about spatial relationship of biomasses across a river transect, and in particular evaluate algal densities in the wadeable or near-shore environments.

[69] *Epithemia sorex* is frequently found as an epiphyte on Cladophora and other coarse filamentous algae in western rivers. It is most common in N-limited habitats. Due to their ability to directly fix nitrogen from N_2 gas, they do not need aqueous nitrogen to maintain their biological metabolism.

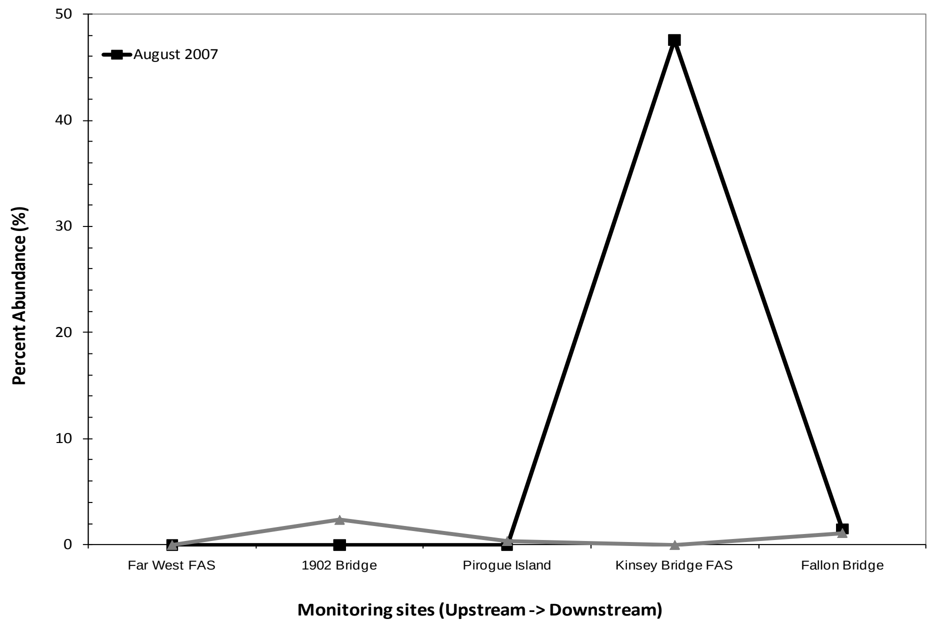

Figure 10-14. Percent abundance of nitrogen fixing *Epithemia sorex* at Kinsey Bridge FAS in 2007. The Kinsey FAS site was the only site with a large percentage of nitrogen fixing algae, which is suggestive of very strong nitrogen limitation. It also explains the deviation between the model simulations and observed algal biomasses at this site. Data reproduced with permission from Charles and Christie (2011).

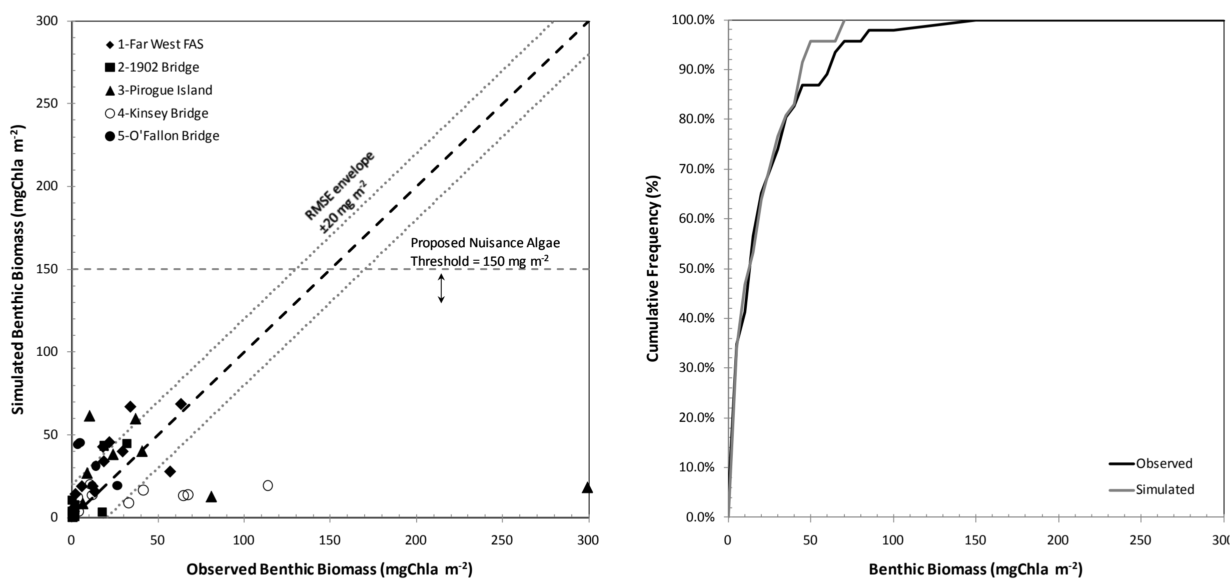

Figure 10-15. Benthic algal biomass lateral simulation reliability.
(Left panel) 1:1 plots of model simulations for the calibration period. A RMSE envelope is shown which represents a plausible margin of safety to account for simulation error as discussed previously. (Right panel) Cumulative frequency plot of simulated and observed benthic algal biomasses for the August 2007 period indicating relatively consistent simulation of lower biomasses and tendency to underestimate higher biomasses.

10.4.3.3 Phytoplankton

Phytoplankton simulations reflect algae that are in suspension, primarily plankton or displaced benthic algae. Plankton increase due to photosynthesis and are lost via respiration, death, and settling. DEQ has two different lines of data to evaluate phytoplankton simulations: (1) integrated EWI field samples and (2) continuous fluorescence measurements from our YSI datasondes[70].

As shown in **Figure 10-16**, concentrations of phytoplankton differed significantly between the August and September (8 to 16 µg L^{-1} in August, and 4 to 12 µg L^{-1} in September), but were well represented both periods. In both instances we met our QAPP requirement of ±10% (using the sonde data) and RMSE and RE were between 1.9-3.2 µg L^{-1} and -16.3 to -2.0% during calibration and 1.1-3.0 µg L^{-1} and -3.0 to -17.3% in validation (depending on which data source was considered). Phytoplankton tended to increase in the downstream direction with an apparent plateau near the downstream end of the study reach. This reflects the point where light and nutrient limitation is near maximum.

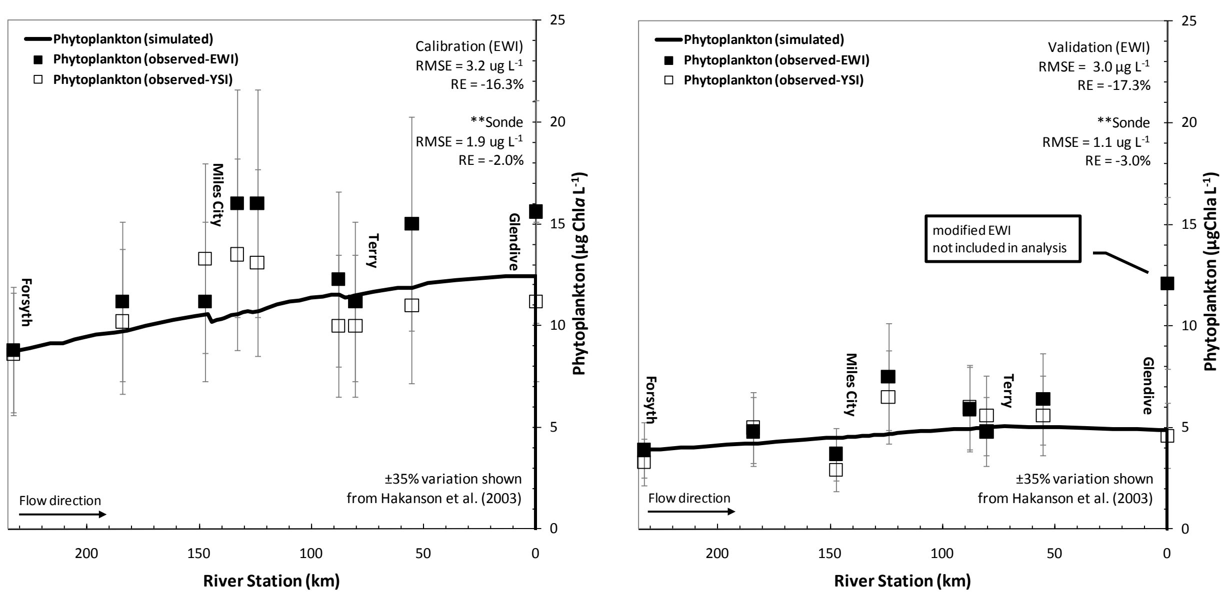

Figure 10-16. Phytoplankton simulations for the Yellowstone River during 2007.
(Left panel) Simulated and observed phytoplankton Chl*a* simulations for the August 2007 calibration period. (Right panel) Same but for the September validation. Both the EWI and sonde data are shown.

10.4.4 Oxygen

Dissolved oxygen (DO) is a very important indicator of river productivity and was heavily relied upon in model calibration[71]. Oxygen is gained from photosynthesis and lost via CBOD oxidation, nitrification, plant respiration, and sediment oxygen demand. Depending on saturation, it can also be gained or lost

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[70] The sondes were calibrated to field Chl*a* measurements. A simple 1:1 empirical adjustment was made at each site for each specific collection period. For example, if the sonde recorded a Chl*a* value of 10.0 µg L^{-1} and the measured Chl*a* value was 8.0 µg L^{-1}, the entire time-series for each sonde was adjusted by a factor of 0.8 for the period of interest (e.g., August or September). Due to the daily variation in sonde data (especially suppression of fluorescence) these calibrations were completed over a 1- or ½-day average period after the sonde was cleaned.
[71] We feel that DO is a good indicator because it reflects net community photosynthetic response and is reliably measured. We used the YSI 6600 sondes extended deployment system (EDS) which has an optical probe and provides some of the best field measurements possible (accurate calibration, minimal long-term drift). Additionally, we quantified many of the sources and sinks of DO in the field.

through reaeration. In review of our DO simulations, the model performed reasonably well for the calibration and poorly for the validation (**Figure 10-17**, Top left/right panel). Despite the latter, we still met the QAPP DO requirements (±10) with RMSE=0.59 and 0.63 mg O_2 L^{-1} and RE=-2.5 and 0.21%. In the validation, diurnal DO was somewhat questionable. As indicated in previous sections, the alternative growth rate resulted in better simulations (RMSE=0.42 mg O_2 L^{-1} and RE=-2.5%).

The Yellowstone River exhibited several distinctive areas with site-specific DO response. The effects of the Cartersville Diversion dam (located just downstream of the Forsyth site) are clearly shown at km 231.5 pushing the minimum and maximum diurnal DO prediction towards saturation. Diel DO patterns then quickly recover and are fairly consistent until reaching Miles City (km 150) where river productivity increases due to wastewater contributions and changes in river morphology. The change only occurs for a short period and then productivity declines from that point downstream. In essence, this marks the point where benthic algae dominance ceases and the river becomes a turbid, phytoplankton-dominated system. Thereafter, the DO signal grows weaker more closely approximates saturation.

We were unable to capture the full magnitude of the daily diurnal DO variation near Miles City. At least some of our apparent inability could be related to incomplete lateral mixing[72] of wastewater effluent in the observed area. Calculated lateral mixing length below the WWTP was considerably longer than the distance to the sonde (at station km 133), as well as the next sonde downstream[73]. This suggests that the wastewater effect might have been larger on one side of the river than the other and was causing deviation between the simulated and apparent[74] observed data. Due to this possibility, we are not overly concerned about the deviation between the model and the field data at this location.

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[72] The sonde at km 133 (the one with the large diurnal variation) was on the right bank which was the same side as the wastewater discharge while the next downstream sonde (km 124) was on the opposite bank and had a more moderated diel swing.

[73] Lateral mixing length in meters (L_m) is calculated according to Chapra (1997) using the Fischer, et al., (1979) formula, where $L_m = 0.4U\, B^2 / E_{lat}$ and *U*=velocity (m s^{-1}), *B*=channel width (m), and E_{lat}=the lateral dispersion coefficient. The lateral dispersion coefficient can be calculated as: $E_{lat} = 0.6HU^*$, where U^*= is the critical shear velocity (m s^{-1}) and *H*=depth (m). Critical shear can be calculated as $U^* = \sqrt{gHS}$, where *g*=acceleration of gravity (m s^{-2}), and *S* = slope (m m^{-1}), and for this reach, U^* was estimated to be 0.077 m s^{-1} (*S*=0.00081, *H*=0.75). Given *B* = 150 m and *U*=0.7 m s^{-1}, L_m would be very large (181 km).

[74] The word "apparent" was used due to concern about lateral mixing described previously and the fact that the sonde data may not have been reflective of the overall condition of the river.

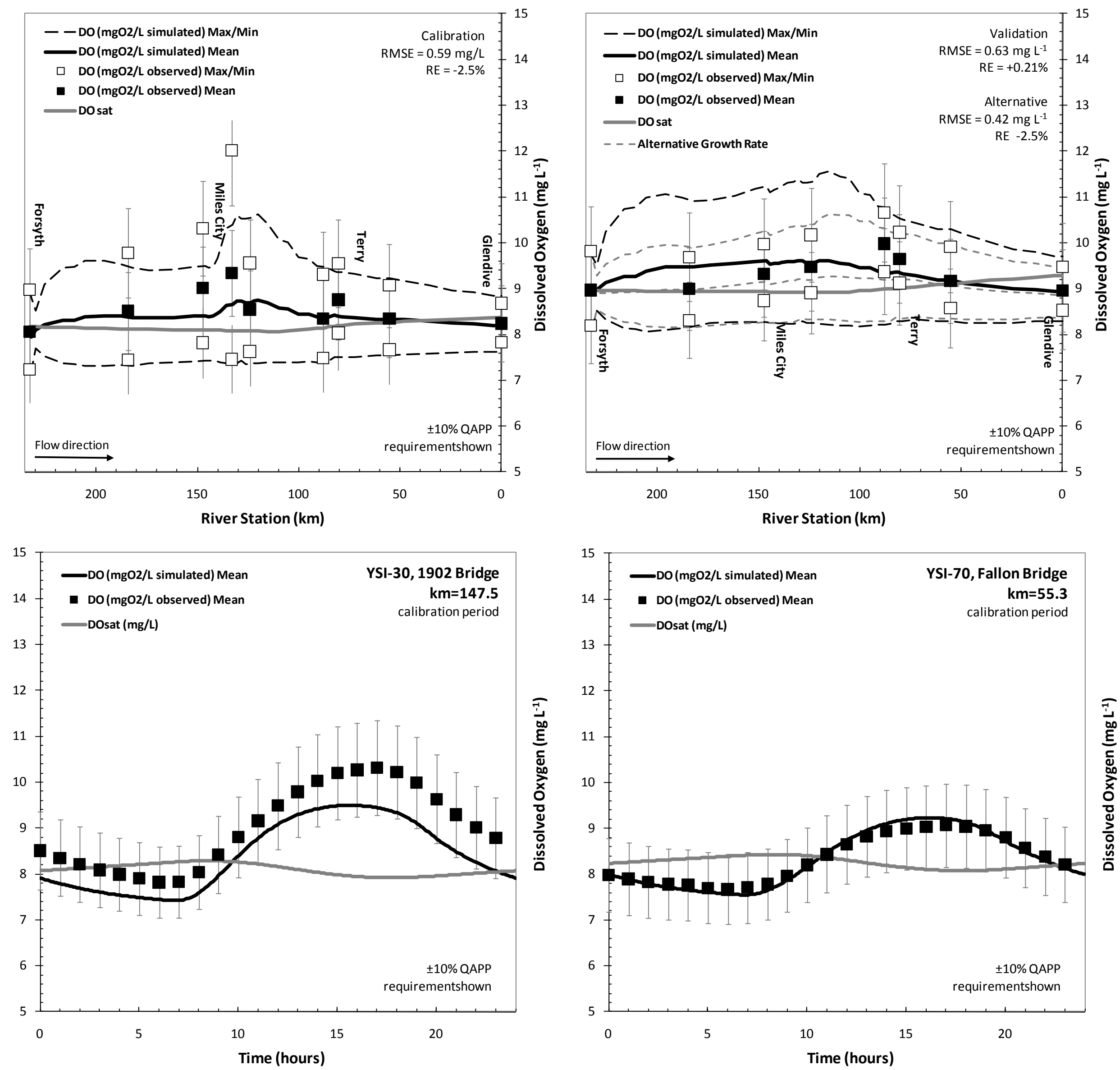

Figure 10-17. DO simulations for the lower Yellowstone River during 2007.
(Top left/right panel) Simulated and observed DO for August and September 2007. The highest areas of productivity are near Miles City and moderate downstream due to light limitation. (Bottom left/right panel) Diurnal simulations for the 1902 Bridge (km 147.5) and Fallon Bridge upstream of O'Fallon Creek (km 55.3) which are roughly at ⅓ and ⅔ of the project reach length.

Diurnal model evaluations also provide insight into short-term field processes such as photosynthesis and respiration. We evaluated two sites located at approximately ⅓ and ⅔ of the overall reach (km 147.5 and km 55.3). Modeled diurnal DO is quite reasonable (**Figure 10-17**, Bottom left/right panel) and minima occur near daybreak (6:00 a.m.), with means near midday and midnight, and maximums near 5:00 or 6:00 p.m. The influence of solar noon on algal productivity, respiration, and reaeration typify the sinusoidal DO pattern over the day (Chapra and Di Toro, 1991; Odum, 1956).

10.4.5 Carbon

10.4.5.1 pH and Alkalinity

Both pH and alkalinity are related. The former varies as a function of total inorganic carbon (C_T) which increases due to carbon oxidation and respiration, and decreases due to photosynthesis (note: reaeration causes either an increase or decrease depending on saturation). Alkalinity is a measure of the river's ability to neutralize acids (buffering capacity) or maintain a pH. Many processes affect alkalinity including nitrification and denitrification, OrgN/P hydrolysis, photosynthesis, respiration, nutrient uptake, and excretion by both benthic algae and phytoplankton.

As shown in **Figure 10-18** (Top left/right panel), pH simulations for the Yellowstone River were fairly good in calibration and marginal for validation (similar to what was described previously for other state-variables). RMSE and RE were 0.16 and 0.18 S.U. and 0.9 and 1.2% respectively. The alternative growth rate yielded very similar results with an RMSE=0.16 and RE=-1.1%. Simulated pH tracks well with known areas of productivity and is greatest in areas of the highest algal growth due to the fact that photosynthesis reduces available carbon dioxide and subsequently carbonic acid and hydrogen ion concentration. Diurnal variability is greatest in these locations as well (e.g., near Miles City. km 150). In calibration, pH was found to depend more on the groundwater influx than any of the surface water exchanges. Shifts were apparent at each of the major tributaries (Tongue River and Powder River) and elsewhere pH was found to be more like surface water than groundwater. The key difference was believed to be subsurface water returning to the river from irrigation return flow. This was validated through the calibration of conductivity as detailed in **Section 10.4.6**.

Alkalinity was also evaluated (**Figure 10-19**, bottom). Little can be said however due to the fact that we only collected a single round of alkalinity measurements to make model comparisons[75]. RMSE and RE were 1.5 and 2.5 mg CaCO3 L^{-1} and 0.0 and -1.4%. Statistical values probably best reflect an estimate, but even so we feel the simulations reasonably represent conditions during 2007. Values were consistently around 170 mg $CaCO_3$ L^{-1}, with only slight shifts near the Tongue and Powder Rivers.

[75] Alkalinities were monitored only in September and for just the mainstem river and influent WWTPs. Estimates were made for rest of the tributaries using the geometric mean of the values over the August and September measurement period at USGS gage sites. If no gage was present, a reasonable approximation was made from nearby field data.

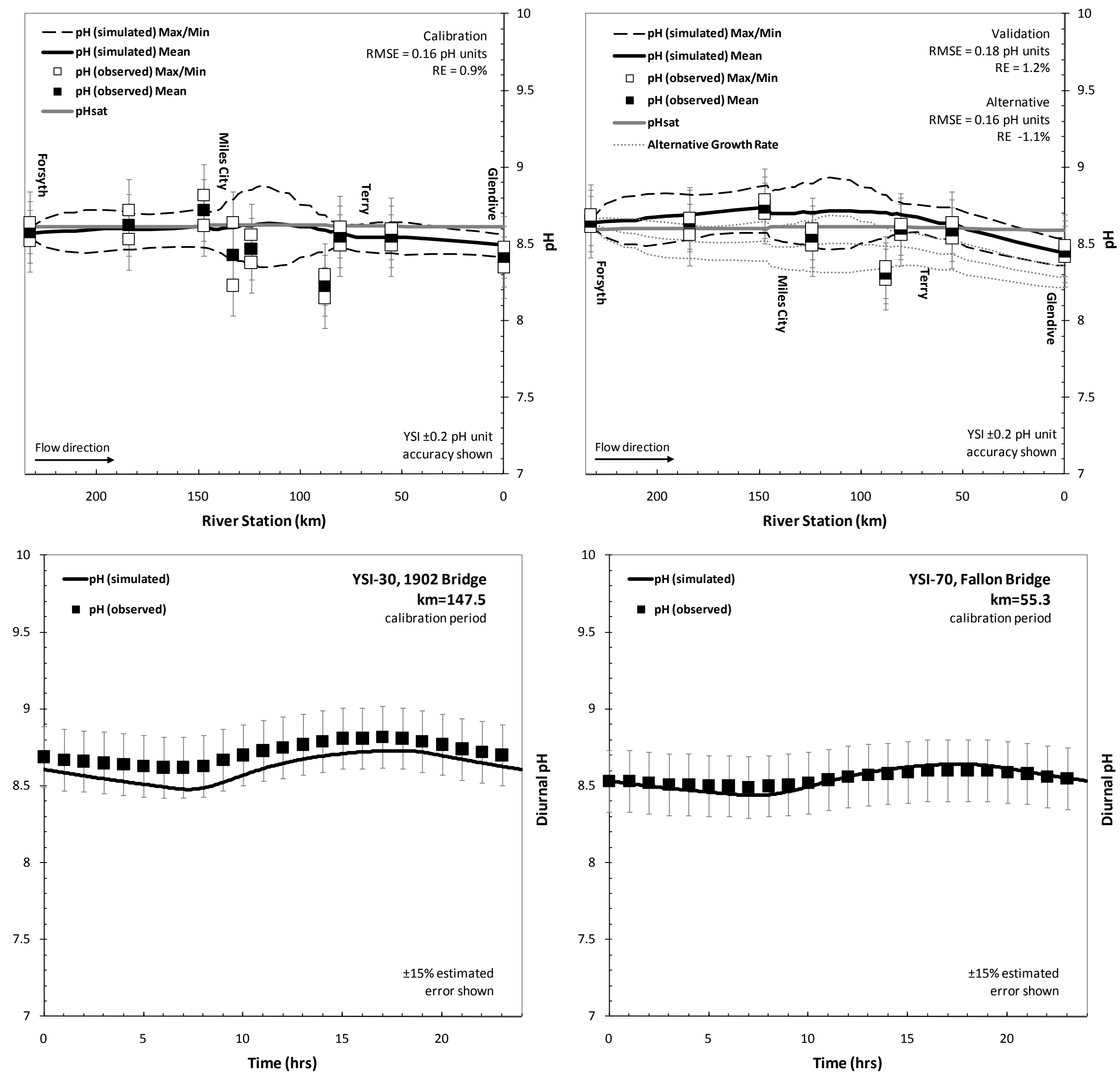

Figure 10-18. pH simulations for the Yellowstone River during 2007.

(Top left/right panel) Simulated and observed pH (S.U.) for the August and September 2007 calibration and validation periods. (Bottom left/right panel) Diurnal simulations for several of the sites previous; the 1902 Bridge (km 147.5) and Fallon Bridge upstream of O'Fallon Creek (km 55.3).

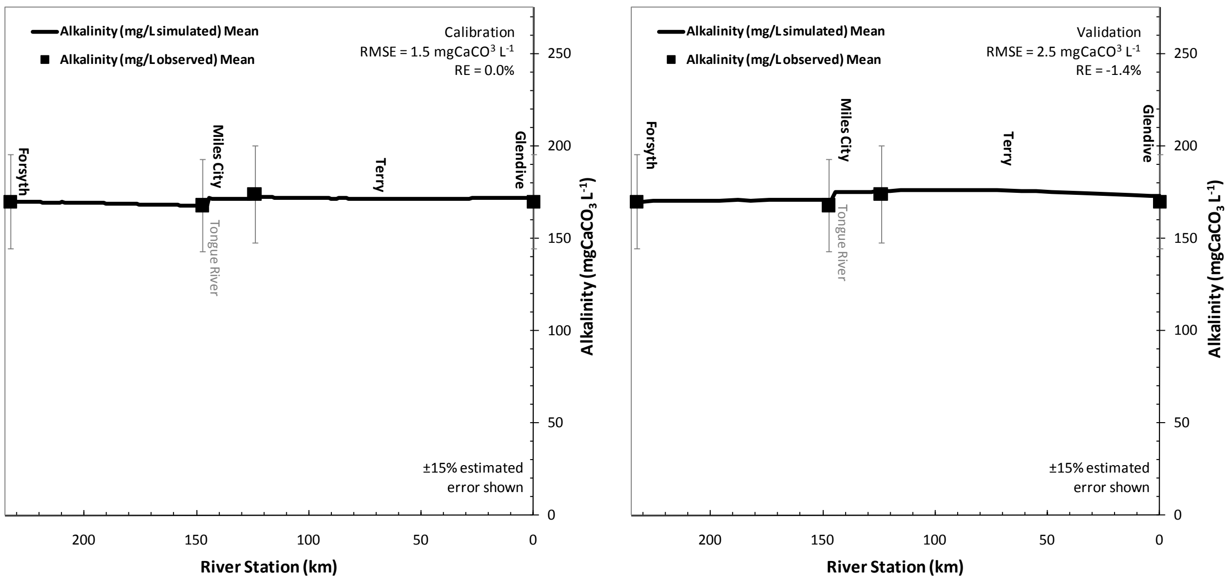

Figure 10-19. Alkalinity simulations for the lower Yellowstone River during 2007.
(Left panel) Simulated and observed alkalinity for the August 2007 calibration period. (Right panel) Same but for the validation period.

10.4.5.2 CBOD

Carbonaceous biochemical oxygen demand (CBOD) reflects the oxygen demand exerted on the river through oxidation of carbon. Only one type of CBOD (fast, $CBOD_f$) was modeled as all forms of CBOD on the Yellowstone River were believed to be similar. $CBOD_f$ is gained from the dissolution of detritus and is lost from oxidation and denitrifcation. In 2007, we measured 5-day $CBOD_f$. Values were quite low, below the analytical reporting limit of 4 mg L^{-1} therefore we only had unreported laboratory values available. These ranged from 0.32-2.3 mg L^{-1} and were determined to be unreliable given their wide inter-site variability and apparent lack of data structure. We chose instead to use historical field dissolved organic carbon (DOC[76]) data normalized to CBOD units[77] to estimate $CBOD_f$.

Model comparisons were made with CBOD-ultimate ($CBOD_u$) which is the sum of $CBOD_f$ (as described in the previous paragraph) and CBOD in particulate form ($CBOD_p$). Since particulate organic carbon was measured in the field, we simply assumed that 2.67 grams of oxygen were required to oxidize one gram of particulate carbon and summed this with our previous estimate. $CBOD_u$ model simulations are shown in **Figure 10-20** (Top left/right).

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[76] The particulate organic carbon (POC) measurements between 2000 and 2007 were very similar. Both were approximately 1 mg C L^{-1} at Forsyth. Hence we assumed DOC to be similar between the years.

[77] It was assumed that all of the organic carbon would ultimately be oxidized. The stoichiometric mass relationship of 2.67 : 1, or $\frac{1\,\mathrm{mole}\,O_2}{1\,\mathrm{mole}\,C}\times\frac{32\,\mathrm{g}\,O_2}{1\,\mathrm{mole}\,O_2}\times\frac{1\,\mathrm{mole}\,C}{12\,\mathrm{g}\,C}$ was used.

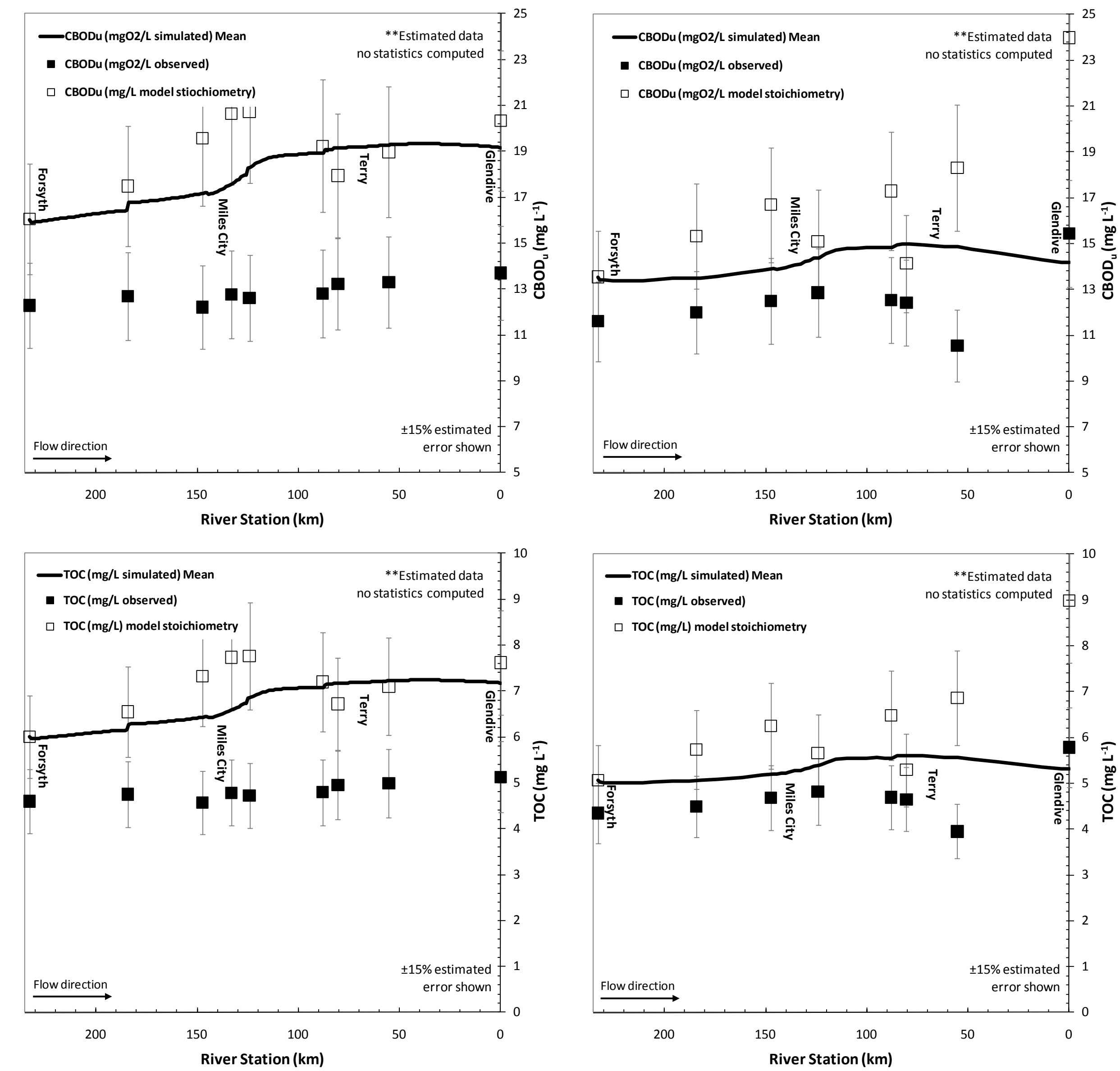

Figure 10-20. CBOD-ultimate and TOC simulations for the Yellowstone River during 2007.
(Top left/right panel) Simulated and estimated observed CBOD for the calibration and validation respectively. (Bottom left/right panel). Same but for TOC. The filled squares reflect the actual CBOD/TOC estimates for the river while the open squares reflect the data if calculated from the model stoichiometry (see discussion in text). The consistent deviation between the two data requires a correction factor of approximately 0.65.

Overall, we see that $CBOD_u$ is over-simulated in all instances, but by a similar percentage. The deviation stems from the fact that carbon to detritus ratio is assumed to be ~2.5:1 (107:43), typical of algae when they die (Chapra et al., 2008). However, the actual detrital makeup of the river is far removed from that, more on the order of 9:1 (perhaps allochthonous or of terrestrial origin). Thus we have artificially inflated the amount of carbon in detritus and overestimated $CBOD_u$. Checks can still be used see if the model is reasonable by correcting the field data with the implied stoichiometry. This has been done and is shown in the figures. With this understanding, we feel that the model simulates CBOD reasonably (despite the problems mentioned previously) and that the overall longitudinal profile of the river is well-represented. A relatively constant increase in CBOD occurs until Miles City (km 150), is followed by a marked increase due to increases in productivity, and then finally decreases in the lower river due to light limitation. In all instances, CBOD simulations are 35% high, and carbon in the model must be scaled down by a factor of 0.65 to reflect actual field conditions. This overestimation slightly affects computed pH values in the model (more C_T causes a decrease in pH).

10.4.5.3 Total Organic Carbon (TOC)

TOC is calculated by Q2K as the sum of DOC (dissolved carbon) and POC (suspended carbon). It is also affected by the carbon inflation discussion previously (with reference to CBOD). TOC was not collected or analyzed, but is shown in **Figure 10-20** (Bottom left/right) where the observed data reflect the sum of the carbon in phytoplankton and detritus. Simulated TOC values range from around 5-6 mg C L^{-1} at the headwater to around 6-7mg C L^{-1} at the end of the reach which must be adjusted down by the multiplier of 0.65 to reflect the true carbon content of the river. No statistics were computed for TOC given this consideration and the lack of available data. Given that TOC in the river is already above the drinking water regulatory threshold of 2 mg C L^{-1} (where treatment for disinfection by-products becomes a requirement), and is in an adjusted range of 3.3-4.6 mg C L^{-1}, it is likely not a direct factor in criteria development for the river.

10.4.6 Conductivity

Conductivity is a conservative substance (i.e., reflective of salts in solution) and a good overall check on the validity of the model. Although unrelated to nutrients, it is presented here as final consideration. Conductivity simulations (**Figure 10-21**) were fairly consistent in 2007 and were primarily a function of headwater conditions and slight changes coincident with groundwater inflow and major tributaries. As identified previously, the conductivity calibration caused us to believe that the primary recharge source was irrigation return flow (subsurface) as relatively clean low influent water was needed to calibrate the model (as opposed to the fairly saline regional groundwater flow systems).

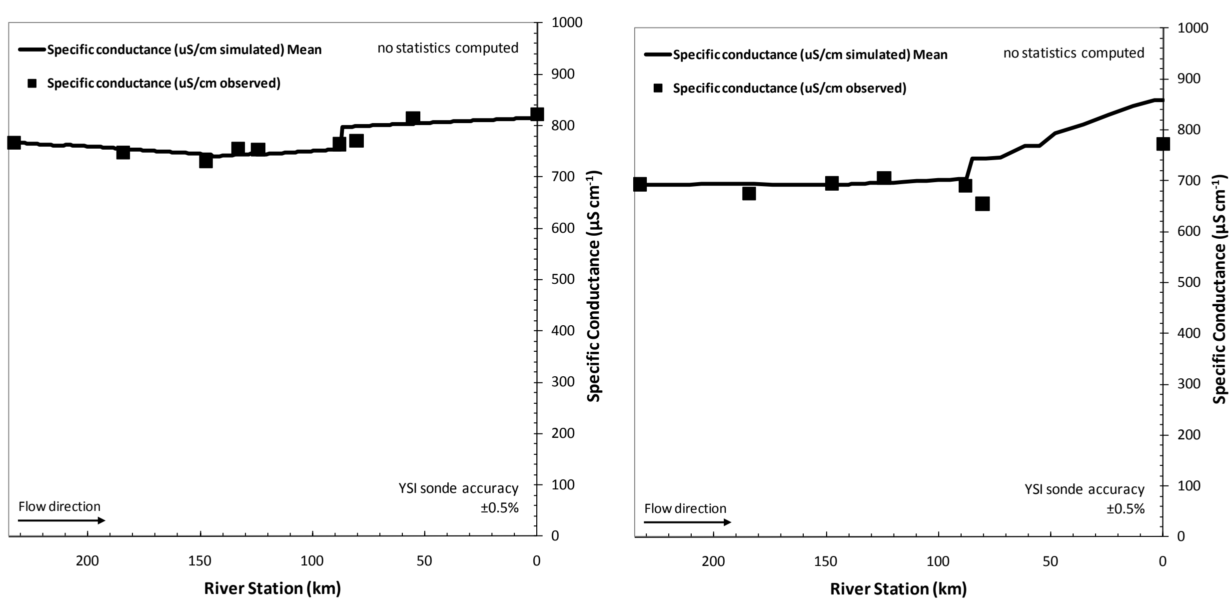

Figure 10-21. Conductivity simulations for the Yellowstone River during 2007.
(Left panel) Simulated and observed conductivity for the August 2007 calibration period. (Right panel) Same but for the validation period.

11.0 Model Evaluation Beyond the Original Validation

It was apparent in the previous section that difficulties were encountered with the model validation. We were able to isolate this to a specific functional group in Q2K (i.e., benthic algae), but additional information was needed to make any robust conclusions about the model's predictive utility. We consulted with experts from the Philadelphia Academy of Natural Sciences (PANS) regarding the differences between the August and September 2007 data and also completed a second validation using a low-flow dataset collected by U.S. Geological Survey (USGS) in August of 2000. Each of these activities is detailed below.

11.1 Phycological Evaluation and Growth Rate Changes as Evidenced by Algal Taxa

Taxonomic evaluations of algae from the Yellowstone River were completed by phycologists (i.e., those who specialize in algae taxonomy and ecology) at PANS using the August and September 2007 data. Samples collected by USGS in 2000 were also considered. While no single overwhelming difference was identified between the periods, the following dissimilarities were noted (Charles and Christie, 2011):

- **Differences in Algal Taxa** – PANS identified that algal counts in August 2007 and September 2007 showed differences in the relative abundance of taxa (20% different), most notably the proportion of the diatoms that were eutrophentic (i.e., high productivity). Eutrophentic species are often associated with higher nutrient conditions and faster growth rates and were more abundant in August than September. Likewise, the percentage of dominant taxa were higher in August than September, another factor that typically occurs with higher growth rates.

- **Changes in 3-D Structural of benthic algal matrix** –The percentage of motile taxa (i.e., those that can move up or down in the algal mat) were also higher in August than September of 2007 which suggests that the algal mat may have been thicker and more productive in August than September. There was also a higher relative biovolume of *Cladophora* in August than September. *Cladophora* provides a three-dimensional structure for algal growth that allows more efficient and greater use of light and also provides a greater surface area on which diatoms can grow (i.e., diatoms tend to have a faster growth rates than filamentous algae).

A number of other, less probable factors were also identified (Charles and Christie, 2011). For example, it was suggested that changes in phytoplankton density from August to September could be responsible for the shift. However, the model already accounts for the phytoplankton shift and the state-variable was well represented in the model during both simulation periods. A second possibility was photosynthetic use efficiency changed (i.e., adjustment of Chl*a* levels to varying light levels). Hill et al., (1995) show that shaded periphyton are two times more efficient in their photosynthetic response at low-irradiances compared to non-shaded periphyton. Similarly, algae are capable of adapting to very low irradiances with respect to growth rate (Falkowski and LaRoche, 1991; Rier et al., 2006). However, we did not necessarily see a shift in the ratio of Chl*a* to ash-free dry weight in our field data (meaning algae did not have more Chl*a* per unit biomass in September). Additionally, light half-saturation constants used in the modeling were near the middle of the range reported in the literature (Hill, 1996) and adjustment of this parameter didn't seem justified (i.e., there was no reason to believe that a major shift in light use efficiency occurred).

Finally, PANS noted differences in water temperature between the periods. A change of about 5°C occurred between August and September (from 21°C to 16°C), which from a theoretical standpoint would reduce the rate of productivity by about 25% (i.e., a doubling in growth rate occurs per 10°C, per Arrhenius). Since such changes are already accounted for in the model, adjustments outside of the range reported by Eppley (1972) seem inappropriate without data suggesting otherwise.

A final consideration is senescence. While the process is not well-understood, there are four commonly recognized causes for growth termination at the batch culture level. These include changes in: (1) pH, (2) CO_2 concentration, (3) light, and (4) nutrients (Daley and Brown, 1973). The most measurable indicator is a decline in photosynthetic productivity (less DO production) accompanied by alteration of the C:Chl*a* ratio. Senescence-like responses can occur from changes in photoperiod (i.e., the length of the day), and have been shown to occur seasonally in algal spore germination for the purpose of overwintering (Suzuki and Johnson, 2001). In such cases, plants and animals sense the duration of the day and/or night and respond appropriately.

A similar example observed in the field by DEQ was found in Montana's prairie streams. In Box Elder Creek a detailed whole stream fertilization study was conducted to establish the dose-response relationship with nutrients. In that instance, senescence was found to occur in very late September/early October (DEQ, 2010) at a time when nutrient additions were still occurring, meaning the effect was not related to nutrient depletion but rather to water temperature/length of day; similar to when leaves fall off terrestrial vegetation in autumn. The most noted response was a decreased diurnal DO flux, a strong decline in DO concentrations relative to saturation indicative of a shift from algal production to decomposition, and accumulation of dead/dying algae in the channel. A similar set of conditions occurred in the Yellowstone River during late September 2007, including observations of sloughed algal accumulations on the shoreline. Perhaps, then, senescence is another consideration.

In summary, a number of plausible explanations reflect our inability to represent the river response during fall 2007. We do not know the exact mechanism, but whatever the case, the river was more productive in August than in September and we believe the cause to be benthic algae. We confirmed this through two lines of evidence, our model simulations as well as the expert review by PANS. Adjustment of the algal growth rate was a simple remedy to fix the problem (which could have been done though a number of other possible mechanisms) and moderated the difference beyond calibration and validation. However, because questions regarding the rigor of such an approach may remain (i.e., is the model really validated?) we further addressed this concern below.

11.2 Cross-Validation with 2000 USGS Data

A second piece of validation work was completed by DEQ using an independent dataset collected by USGS during August of 2000 (Peterson et al., 2001). This "cross-validation" allowed for a second model confirmation. Both pros and cons of such an approach are summarized below.

Pros:

- Data collection from 2000 took place at a similar time in August (near the peak of productivity) and therefore may be better suited to our original calibration.

- Hydrologic conditions during 2000 were very similar to 2007, but were quite different in terms of water quality, thereby representing a set of different loading conditions.

- Algal taxa were different in 2000 compared to 2007. The percent similarity was between 20-30% (Charles and Christie, 2011). This allows us to compare and contrast the effect of taxa differences on river conditions.

- The conditions of the model (i.e., low-flow and warm climate) are very similar to the hydrologic design flows used later for criteria development.

- Data were collected on a much larger section of the lower river extending from Billings to Sidney, MT, allowing evaluations over a much larger spatial extent.

Cons:

- The data were not collected specifically for the purpose of modeling.

- Reporting limits used in the USGS study were not as low as desired. They were: NO_3^- =50 µg L^{-1}, NH_4^+=20 µg L^{-1}, and SRP=10 µg L^{-1}, which present problems in understanding biological responses to low-level soluble nutrients.

- There was less diurnal data (e.g., DO, pH, SC, turbidity) specific to our project reach. Only three sites had data (Forsyth, Miles City, and Terry).

- The more detailed features of 2007 (e.g., irrigation withdrawals or return flows, smaller tributaries, WWTP contributions, etc.) were not monitored by USGS.

- A different method was used to characterize benthic algae biomass. We used cross-sectional averages while USGS characterizes the richest target habitat.

Based on the considerations above, we felt that the pros of using the USGS dataset outweighed the cons. One of the most attractive features being that it was collected during another low-flow year and during peak productivity, precisely the condition the model is intended to simulate.

In application of the model to 2000 conditions, steps identical to those described in **Section 8.0 and 9.0** were used. Because diurnal data were only available for three locations in our modeling extent (4 of the sites had chemistry data), the confirmation model was for the entire lower river from the USGS gage at Billings to the Montana state line (586 km). This encompassed the following sites: the Yellowstone River at Billings (diurnal & water chemistry), the Yellowstone River at Custer (diurnal & water chemistry), the Yellowstone River at Forsyth (diurnal & water chemistry), the Yellowstone River at Miles City (diurnal & water chemistry), Yellowstone River at Terry (diurnal & water chemistry), the Yellowstone River at Glendive (water chemistry), and the Yellowstone River at Sidney (diurnal & water chemistry). The longer reach provides a more robust validation and corroborates whether calibrated rates apply to a much longer area of the river (while still within class B-3 waters).

The longer simulation length does have drawbacks however. First and foremost, loadings outside of our 2007 study reach (e.g., Billings to Forsyth and Glendive to Sidney) were not detailed[78]. We used as much

[78] As a result, the larger model (Billings to State Line) is skeletal outside the detailed study area and accounts only for major tributaries and features (e.g., Billings WWTP, Huntley Diversion Dam, Bighorn River, etc.). Stationing for

discretion as possible to fill data gaps. In many cases we applied tributary flows, irrigation exchanges, etc. measured during 2007, which seemed to be consistent with 2000. Results of the model validation are described in the following sections. Statistical results have been detailed previously in **Section 10.0**. Overall we found that model performance was acceptable based on comparisons of TSS, nutrients, algae, and diurnal DO, pH, and temperature data.

11.3 RESULTS

11.3.1 Streamflow Hydrology

In 2000, streamflow ranged from 50 $m^3 s^{-1}$ near Billings (km 586) to just over 120 $m^3 s^{-1}$ in the lower river near Miles City (km 310) and Glendive (km 280). Simulated and observed streamflows are shown in **Figure 11-1**. Overall, the model reflects the water balance quite well. The two primary drivers were the inflow of the Bighorn River which effectively doubles the flow at km 490, and then numerous declines from irrigation throughout the lower river (Huntley, Waco-Custer, Rancher’s, Yellowstone, etc.). As mentioned previously, values for the irrigation diversions in the non-detailed study reach were estimated from irrigated area in the DNRC water resource surveys (as done in the 2007 model). Additionally, diffuse source accretions were used to bring the mass balance into agreement if surface water exchanges alone did not adequately reflect observed streamflow data in the mainstem river. Note that the gage at Glendive was not in operation in 2000.

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the model was based on the 2001 color-IR centerline (converted to km) as used previously (AGDTM, 2004). Locations of diversion dams were identified through the U.S. Fish and Wildlife Service (U.S. Fish and Wildlife Service, 2008). Many simplifications were subsequently necessary due to a lack of data in the unmonitored reaches, in particular, relative to irrigation practices and river hydraulics. To make up for these deficiencies (and any discrepancies in the hydrology mass balance) an accretion term was added to the model to makes sure simulated streamflows at each gage were correct. Likewise, if any differences in temperature, pH, etc., were identified, they too were accommodated through the diffuse accretion term (again only in the non-detailed study reach) to improve the simulation. In the detailed study reach (Forsyth to Glendive), information was kept exactly as in the 2007 model (river stationing, irrigation, etc.). The only adjustment was gaged tributary inflows and associated water quality boundary conditions measured in 2000.

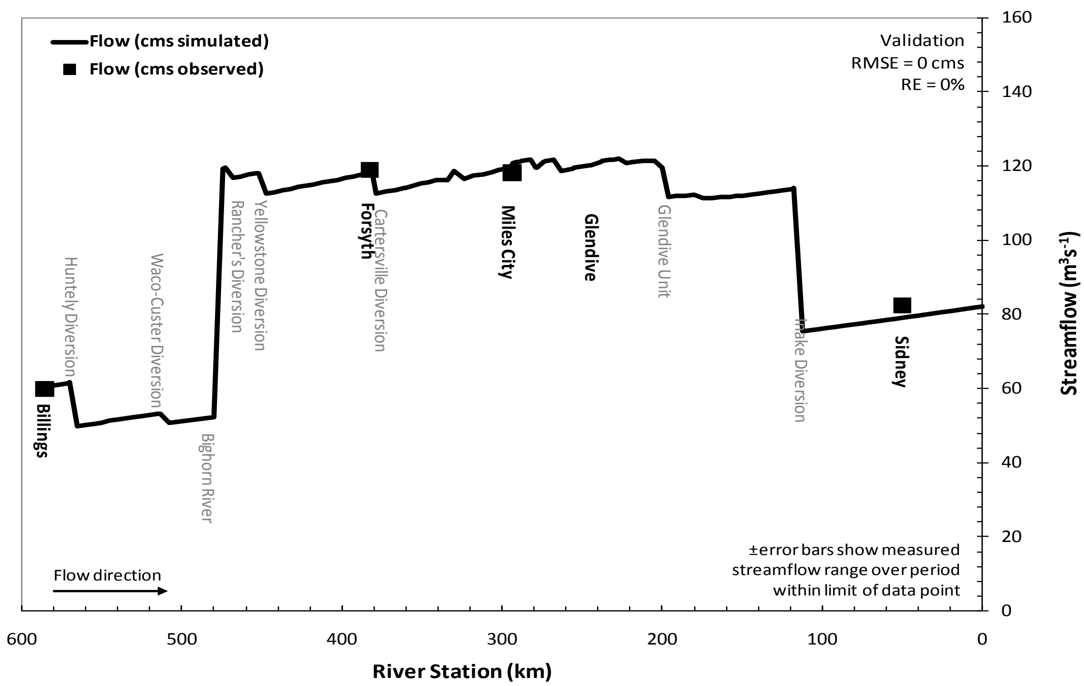

Figure 11-1. Simulated and observed streamflow for the Yellowstone River during 2000.

11.3.2 Mass Transport

The same mass transport functions and indicator variables described in **Section 9.0** were considered in the USGS validation. In this instance there was no width or depth information against which to make comparisons thus we can only speculate as to the model's reliability. From **Figure 11-2** (top) we can see that the model reasonably reflects the major features of the river (dams, inflows, etc.) and seems to be in good agreement with our 2007 detailed study reach. Prominent features include the six major low-head diversion dams on the river (Huntley, Waco-Custer, Rancher's, Yellowstone, Cartersville, and Intake at km 570, 510, 470, 450, 380, and 120, respectively), the large shifts at the Bighorn River brought about by increases in flow, shallowing and widening at Miles City, and then reductions in width and increases in depth in the lower river.

Travel-time and reaeration were also assessed for comparative purposes (**Figure 11-2**, bottom). Again, the adjusted travel-time calculator was used to make estimates for the given flow condition (corrected to the dye-study as done previously for the 2007 model). Accordingly, travel time was estimated to be approximately 11 days from Billings to the state-line which is in good agreement with the model. Reaeration also patterns the 2007 model with the highest reaeration rates in the area of the river where velocities are the greatest (due to gradient, km 586-500 and 300-220 km), and then reductions in the downstream direction as the river deepens.

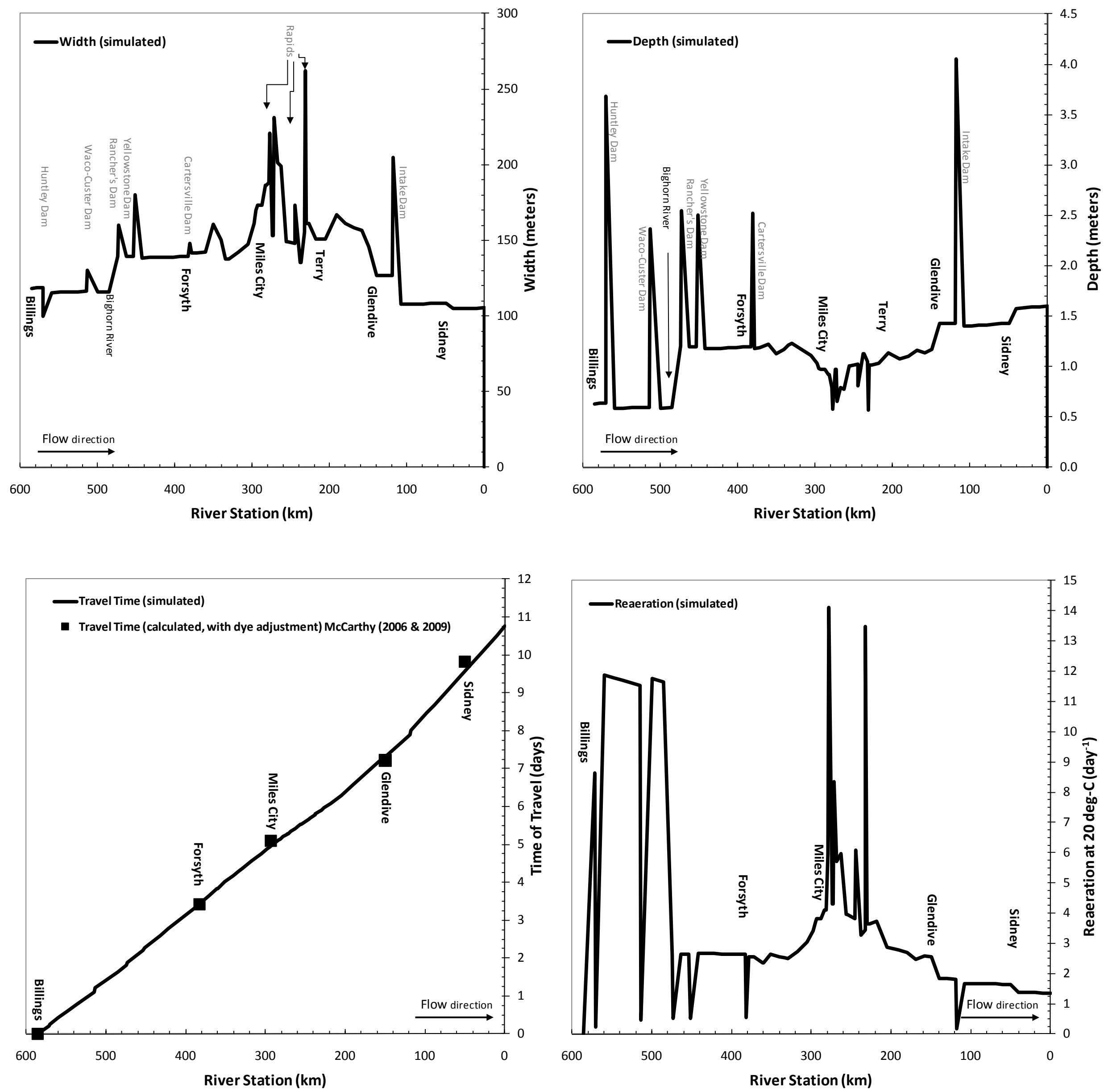

Figure 11-2. Mass transport indicators for the lower Yellowstone River during 2000.
(Top left/right panel). Simulated width and depth. (Bottom left/right panel) Travel-time and reaeration.

11.3.3 Water Temperature

The water temperature simulation was marginal for the 2000 validation (**Figure 11-3**, left). This partially reflects the sequential way the diurnal data was collected by USGS, which consisted of measuring different days at different sites. For example, data was gathered at Billings over the period of August 23-25, Custer on August, 26-28, Forsyth on August 26-28, Miles City on August 29-30, Terry on September 13-14, and Sidney on August 28-30. Thus datasets are very short compared to the multi-week datasets from DEQ and do not share a single common time period. Consequently, there is no reason to expect good correlations in temperature. Despite this limitation, the validation was within the ±5% acceptance criteria. RMSE and RE were 1.4°C and -0.2%. Note that Terry was not included in the statistical analysis as the data was collected two weeks later than the other sites.

A diurnal temperature plot of Miles City (e.g., the middle of the project reach) is shown in **Figure 11-3**, right). It is probably one of the better diurnal plots and shows that the model adequately reflects temperature over the course of the day. Differences are most apparent at midday, and it is possible that this is more a function of the data collection methodology than model error (given the non-typical shape of the observed data). In either case, the QAPP criteria are met and we believe the model is responding appropriately.

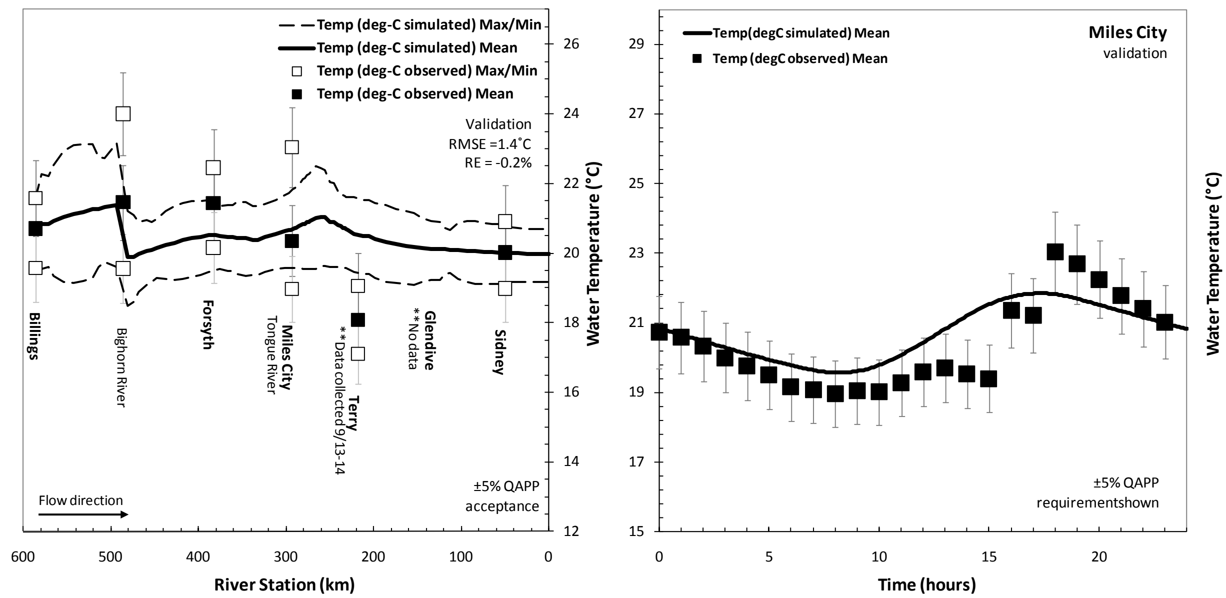

Figure 11-3. Temperature simulations for the Yellowstone River during 2000.
(Left panel) Simulated and observed values for the August 2000 validation period. (Right panel) Diurnal simulations for Miles City.

11.3.4 Suspended Particles

Only suspended sediment concentration (SSC) measurements were made in 2000 which is slightly different than TSS because the entire sample (not an aliquot of the sample) is filtered in the laboratory and dried and weighed for analysis. Consequently differences can arise between the two measures, most notably when heavy particles such as sands are in suspension but are not readily captured in the aliquot. Given the particle size composition of the Yellowstone River under low-flow conditions (primarily clays as demonstrated previously), this difference is not a concern and SSC measurements should be very comparable to TSS measured in 2007[79].

Several assumptions were required to partition SSC into appropriate model compartments. The following relationships were used: ISS = 0.8 * SSC, and detritus = 0.15 * SSC, which are based on the ratios obtained during August and September 2007. Applying these in model, the simulations of TSS are quite good (**Figure 11-4**). RMSE and RE were 2.1 mg D L^{-1} and -7.0% and the plots show similar structure

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[79] It should be noted that TSS is actually a calculated variable in the model. It is computed as the sum of the ash-free dry mass(AFDM, mg L^{-1}) of ISS, detritus, and phytoplankton.

to that of 2007 with a significant increase after the Powder River. Model output for ISS is shown for comparative purposes only.

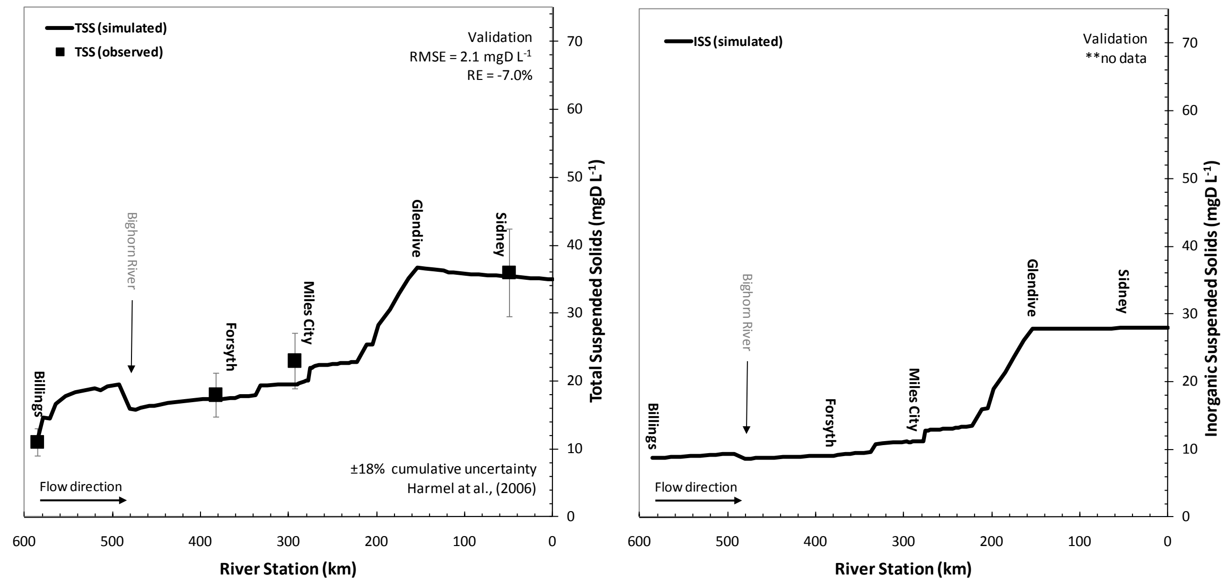


Figure 11-4. TSS simulations for the Yellowstone River during 2000.
(Left panel) Observed and simulated TSS. (Right Panel) Same but for ISS (no field data collected).

11.3.5 Nutrients

Nutrients were substantially different in 2000 than in 2007. The biggest difference was the deviation between soluble nitrogen concentrations longitudinally. At Billings (where the Billings WWTP and other nutrient sources occur) soluble nutrients were quite high whereas they were much lower in the downstream (below detection).

11.3.5.1 Nitrogen

Overall the model did a fair job representing TN in validation (**Figure 11-5**). Concentrations ranged from 300 to 400 µg TN L^{-1} (compared to 500-600 µg L^{-1} in 2007) and RMSE and RE from the simulation were 71 µg L^{-1} and 15.5%. The most significant deviation was near the beginning of the project reach (near Billings) where the model shows a near instantaneous increase in TN (≈100 µg L^{-1}). This occurs from algal uptake of soluble N from the Billings WWTP and subsequent conversion/recycling upon death. Clearly, the rate at which this is occurring in the model is too fast. More N should be bound in the algae instead of recycled through death [i.e., algal death rates need to be much lower (near zero) to reach biomasses approaching those observed in Billings (≈800 mg Chl*a* m^{-2}, *Cladophora* streamers)]. Hence the model does not reflect *Cladophora* growth accurately, or for that matter any case where biomass far exceeds the expected value for diatom-like functional groups such as which the model was originally calibrated (see discussion in **Section 10.4**).

To accommodate this deficiency, site-specific rates were used at Billings (through adjustment of algal death rate), which considerably improved the TN simulation by decreasing the amount of OrgN generated from algal death. Since only a single change was necessary, we can reasonably conclude that other N-related rates in the model are still satisfactory (hydrolysis, nitrification, etc.), and thus a shift in algal rate coefficients may be necessary in locations where *Cladophora* growth are significantly abundant, or where algae are in a better physiological condition due to excess light and nutrients.

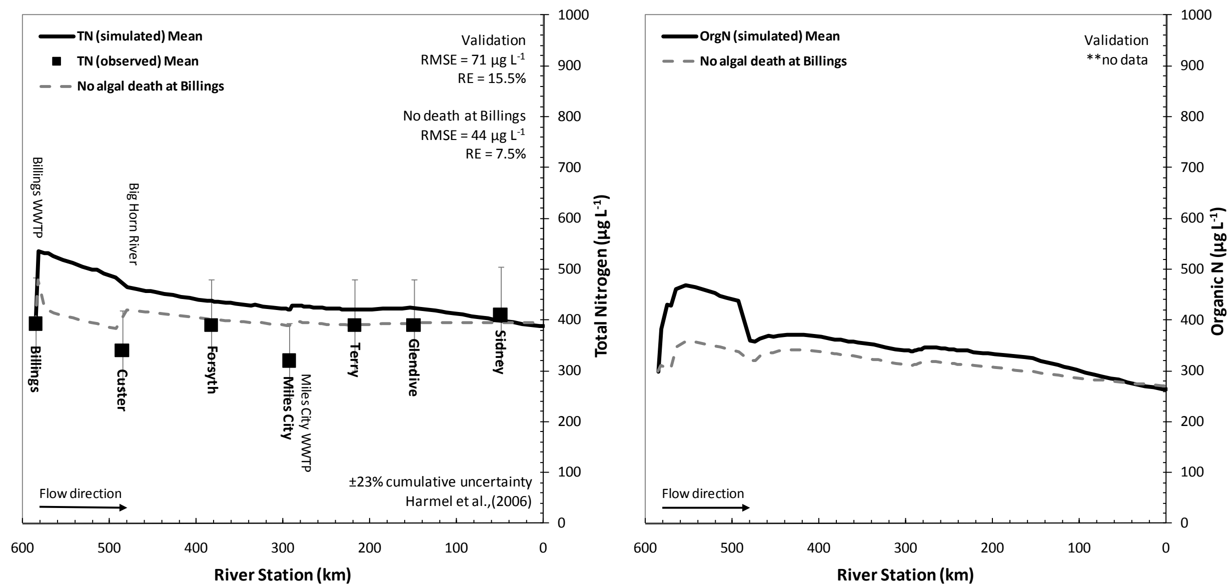


Figure 11-5. Total and organic N simulations for the Yellowstone River during August 2000. (Left panel). Observed and simulated total nitrogen. (Right panel) Same but for organic nitrogen (for reference purposes only, no data collected).

The remaining N data (NO_3^- and NH_4^-) were non-detect and only allow qualitative comparisons. Graphical plots tend to show interesting trends over the study reach (**Figure 11-6**), for example we see that high NO_3^- concentrations occur near Billings (both up- and down-stream of the WWTP), below the Bighorn River, and below the Miles City WWTP. Similar increases are evident for NH_4, though not as exaggerated. The locations generally correlate to areas of highest productivity (as shown later in this section). Since the model generally shows reasonable structure below the detection limit, we can qualitatively conclude that simulation is sufficient. Quantitative data is necessary to make any definitive determination.

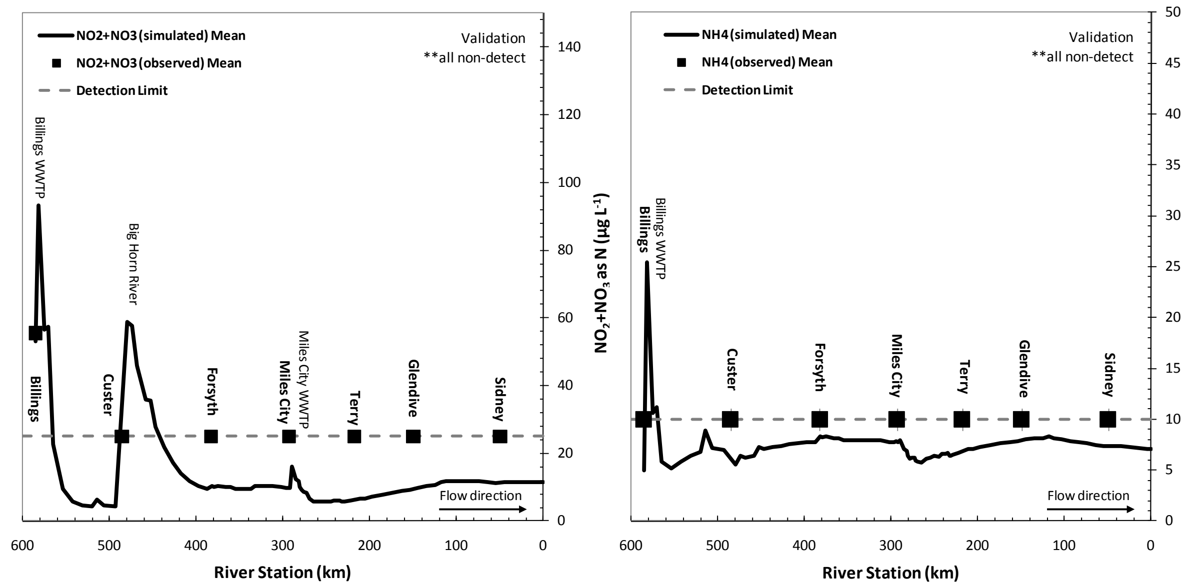


Figure 11-6. Nitrogen simulations for the Yellowstone River during August 2000.
(Left panel). Simulated and observed nitrate. (Right panel) Same but for ammonia. Both of the soluble N data are below the detection limits referenced by the dotted line.

11.3.5.2 Phosphorus

Phosphorus follows a similar pattern to nitrogen. A clear and consistent TP profile occurs characteristic of increased levels near Billings from the Billings WWTP plant, declines at the Bighorn River due to dilution, and then increases downstream of the Powder River (**Figure 11-7**, Left panel) (recall that accretion of ISS below the Powder River also includes OrgP from P sorption). RMSE and RE were 5 µg L^{-1} and -11.7% for TP, and overall, concentrations ranged from 25-50 µg L^{-1}. TP was less affected by the algal conditions described previously for nitrogen due to a lower stoichiometric order.

From **Figure 11-7**, Right panel, SRP could not be characterized due to the fact that it was below detection at all locations. It appears to be most influenced by the Billings WWTP (which caused a quadrupling in concentration), and then from dilution by the Big Horn River and loadings from the Miles City WWTP. The model generally underestimates the decline of SRP downstream of Billings as P depletion had not occurred to appropriate levels before arriving at Custer. This is somewhat masked in the results due to the Big Horn River inflow which occurs directly downstream of Custer. Thus P uptake may be understated in the model. In the lower reaches, SRP levels remain quite low, similar to concentrations observed in 2007, and track quite well. This means that our model may be better trained to simulating lower SRP concentrations (and associated uptake) than those instances approaching an order of magnitude greater in the Billings region.

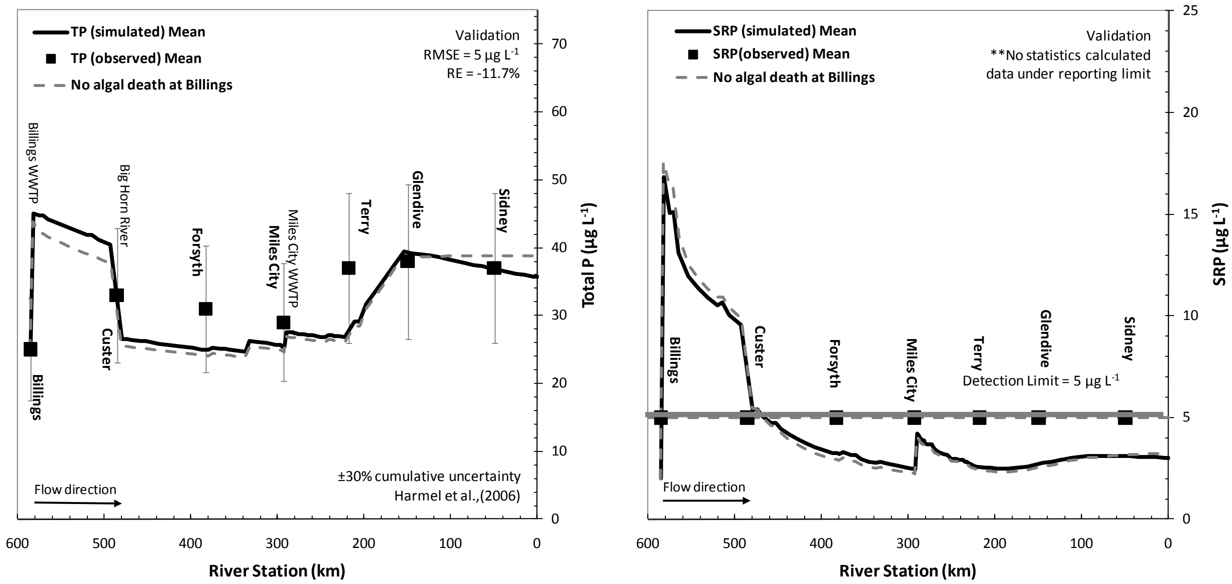
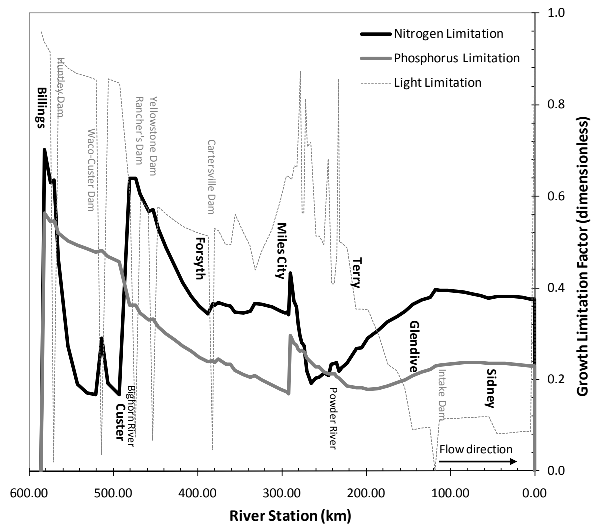


Figure 11-7. Phosphorus simulation for the Yellowstone River during August 2000.
(Left panel). Simulated and observed total phosphorus. (Right panel) Same but for SRP. Note that the USGS observation site at Billings is directly upstream of the WWTP.

11.3.5.3 Nutrient Limitation

As done for the 2007 work, nutrient limitation factors were evaluated for the 2000 condition model. The profile is quite interesting and shows a variety of shifts in the limiting nutrient for benthic algae (**Figure 11-8**, Left panel). Bottom algae switch limitation very quickly based on shifts in ambient concentrations and alternate between P and N limitation successively. Light limitation for benthic algae is also very interesting. Three distinct regions of light occur longitudinally: (1) the region upstream of the Bighorn River (limitation factor of ≈0.9), (2) the Bighorn River to Powder River (limitation factor of ≈0.5), and (3) Powder River to State line (limitation factor of ≈0.1). Given this consideration, our decision to break the river into different distinct nutrient criteria assessment units was a good decision (see **Section 4.4**). The most downstream region (Powder River to state-line) is highly light limited. Hence it is apparent why a shift from benthic algae to phytoplankton dominance has occurred.

For phytoplankton things are less clear and the state of nutrient limitation is strongly dependent on the initial conditions of the model. Since no C:N:P data were collected during 2000, we had to use the data from 2007. In 2007, they were N limited which by default forced us to assume that phytoplankton were N limited in 2000. We have no way to verify this assumption, but based on the similarity of both N and P limitation factors (**Figure 11-8**, Right panel, ≈0.8-0.9), there would be very little difference in the simulation switched to P limitation (only a reduction in growth rate of ≈0.1 would occur).



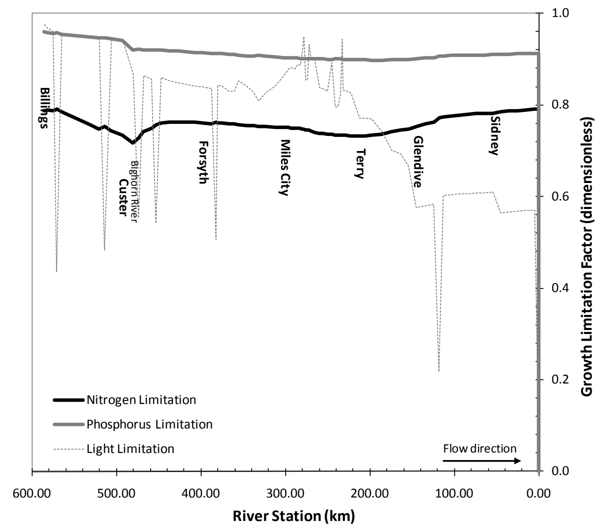


Figure 11-8. Nutrient and light limitation factors for the Yellowstone River during 2000.
(Left panel) Nutrient and light limitation factors for benthic algae. (Right panel) Same but for phytoplankton. The effect of the diversion dams are apparent on both benthic algae and phytoplankton as they increase the river depth and cause strong light limitation.

11.3.6 Algae

Both benthic algae and phytoplankton were considered as part of the cross-validation. As identified in **Section 10.0**, benthic algae are sensitive to element depth in Q2K and unfortunately USGS did not acquire this data during 2000. Nor was their collection methodology similar to ours[80]. As a result, we were not able to make direct statistical comparisons between Q2K benthic output and the USGS data. Qualitative comparisons are shown in **Figure 11-9** (Left panel). Given that USGS data are collected in the richest targeted habitat (see footnote), we expected somewhat higher field biomasses than output by Q2K. This was not the case though as the model over-simulated on five out of seven of the data points. In general, the trend was under-simulation of large biomasses and over-simulation of smaller densities. The overall gradient is reflected in the model however.

Phytoplankton simulations are shown in **Figure 11-9** (Right panel). They were within the expected variation of phytoplankton (Håkanson et al., 2003) and were not significantly different than the calibration values, with RMSE of 1.8 µg L^{-1} and RE of +18.5%. The RE is slightly outside the recommended values for the project QAPP, and we slightly overestimated phytoplankton. Simulations are well within the recommended range reported by others however (**Table 9-1**). Deviation could be due to a number of reasons including slightly different growth rates between 2000 and 2007 (different species or conditions), problems with the temperature simulation, or perhaps differences in available light

[80] The USGS collections were completed using the National Water Quality Assessment (NAWQA) protocol (personal communication, D. Peterson). In this method, emphasis is placed on acquiring data from the richest targeted habitat (RTH), which for the Yellowstone River, was riffles. As a result, data was collected primarily from shallow regions. Additionally, the data were collected only at five different locations at each site. This was done by scraping algae at five separate locations in a wadeable riffle (maximum of 1,000 m longitudinal reach length) and then compositing these samples together for analysis.

(referring back to the suspended particles discussion previously). In any case, the simulations typify literature results and are acceptable to DEQ.

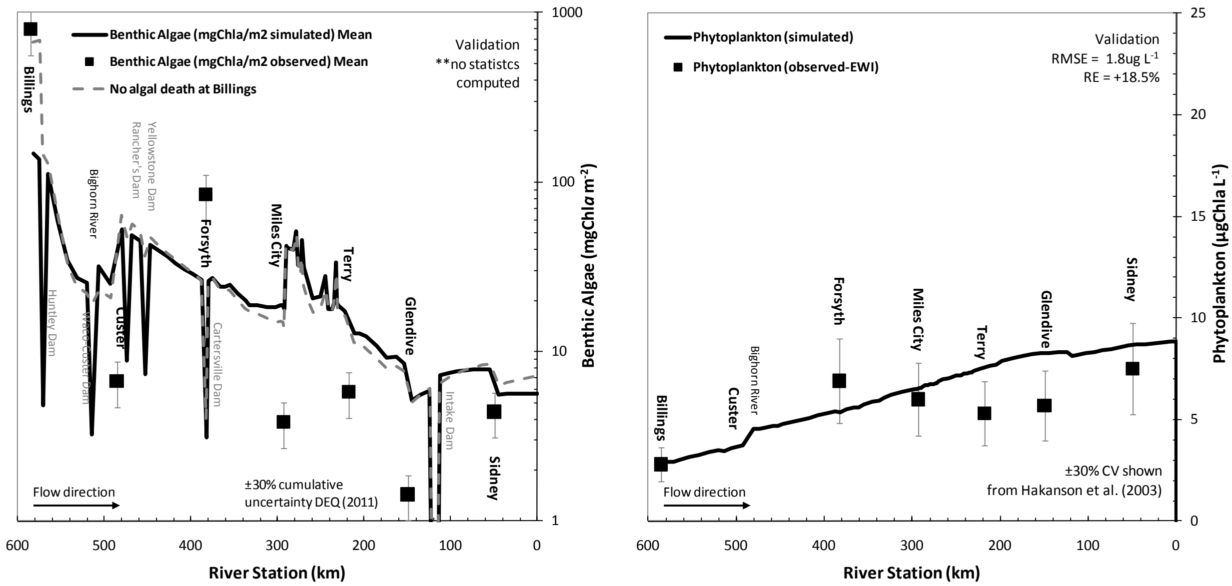


Figure 11-9. Algal simulations for the Yellowstone River during August 2000.
(Left panel) Simulated and observed benthic algae. (Right panel) Same but for phytoplankton.

11.3.7 Oxygen

The dissolved oxygen simulation is shown in **Figure 11-10** (Left panel). Overall, we met the QAPP requirement with RMSE of +0.36 mg L^{-1} and RE of 1.8%. This suggests that the model performs adequately in simulating river productivity in two low-flow situations (2000 and 2007). Very large diurnal DO swings were identified in the Billings region (km 586, 10 mg L^{-1} daily flux) then the river declines in production steadily downstream. The exception is near Miles City (km 580) where wastewater contributions drive productivity back upward for a short period (similar as to seen in 2007). The impact of the low-head dams is also observed pushing the DO minimum and maximum towards saturation.

There was one difficulty in the DO simulation near Forsyth (km 390). Maximum observed DO at this site barely reached saturation levels which is unlikely given the rest of the river profile. Consequently, there was either a problem with the observed data, or a large DO sink (either SOD or CBOD) that we missed[81]. Given the discontinuity in the temperature data (shown previously), and from incidental analysis in the model, we concluded that it was most likely due to the instrumentation placement at Forsyth (i.e., it was not representative of the river). Consequently, that particular data site was omitted.

The diurnal simulations were also quite reasonable. An example for one site, Miles City, is shown in **Figure 11-10** (Right panel). Productivity was at its highest near solar noon and varied consistently with

[81] The Bighorn River enters near this location which could possibly be a source of dead/decaying algae. A CBOD source was already specified for the Bighorn (≈10 mg L^{-1}), which was based on calibration of CBOD (no data were collected in 2000). Historical measurements show very high dissolved organic carbon concentrations can occur from this source.

sunrise and sunset. The model tended to over-simulate temperatures throughout the day at this location. Other sites had better agreement as can be seen in the longitudinal plot.

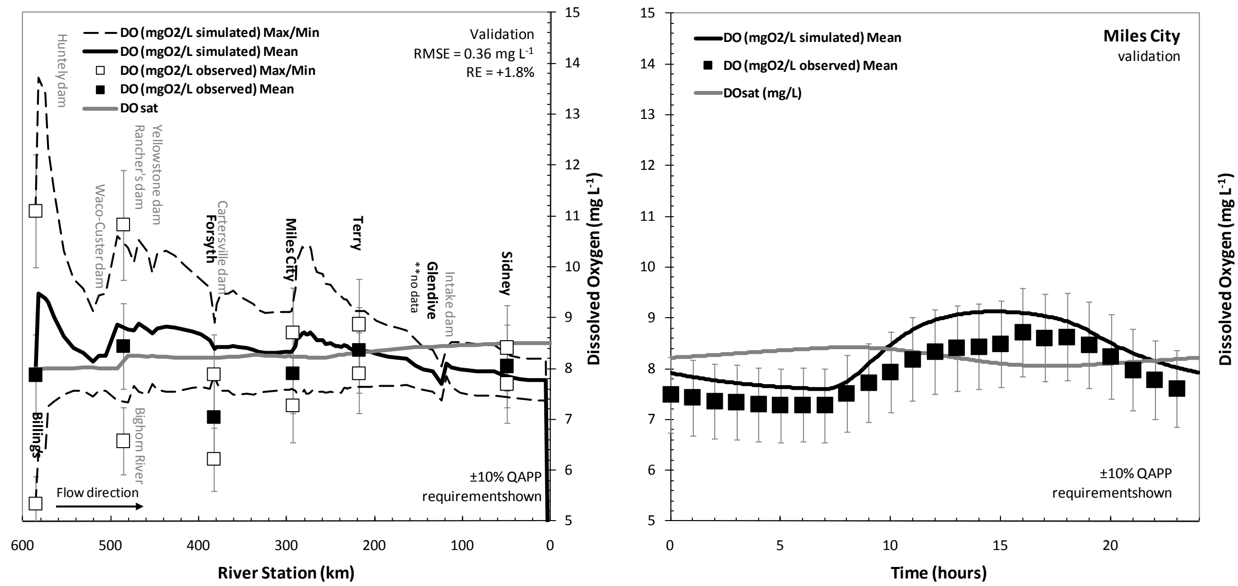


Figure 11-10. Dissolved oxygen simulations for the Yellowstone River during August 2000.
(Left panel) Simulated and observed DO. (Right panel) Diurnal DO simulations for Miles City.

11.3.8 Carbon

Carbon related variables such as pH, alkalinity, and TOC are shown in **Figure 11-11**. Discussions about each are in the following sections.

11.3.8.1 pH and Alkalinity

Longitudinal simulations of pH (**Figure 11-1**, Top left panel) are fairly good with RMSE of 0.07 S.U. and RE of -0.2%. Overall pH correlated well with other productivity-related variables such as DO and benthic algae and showed the widest diurnal variability in the Billings region due to high nutrient levels and algal growth. There was then a consistent decline in pH flux downstream short of a small increase is in the vicinity of the Miles City WWTP (km 250). Diurnal pH was hard to discern due to the multi-day collection method by the USGS but a plot for Miles City is shown in **Figure 11-11** (Top right panel). Alkalinity is shown in **Figure 11-11** (Bottom left panel). Very little data was available to evaluate the latter, but it happens to be reasonable with RMSE and RE of 2.9 mg $CaCO_3$ L^{-1} and -0.9%.

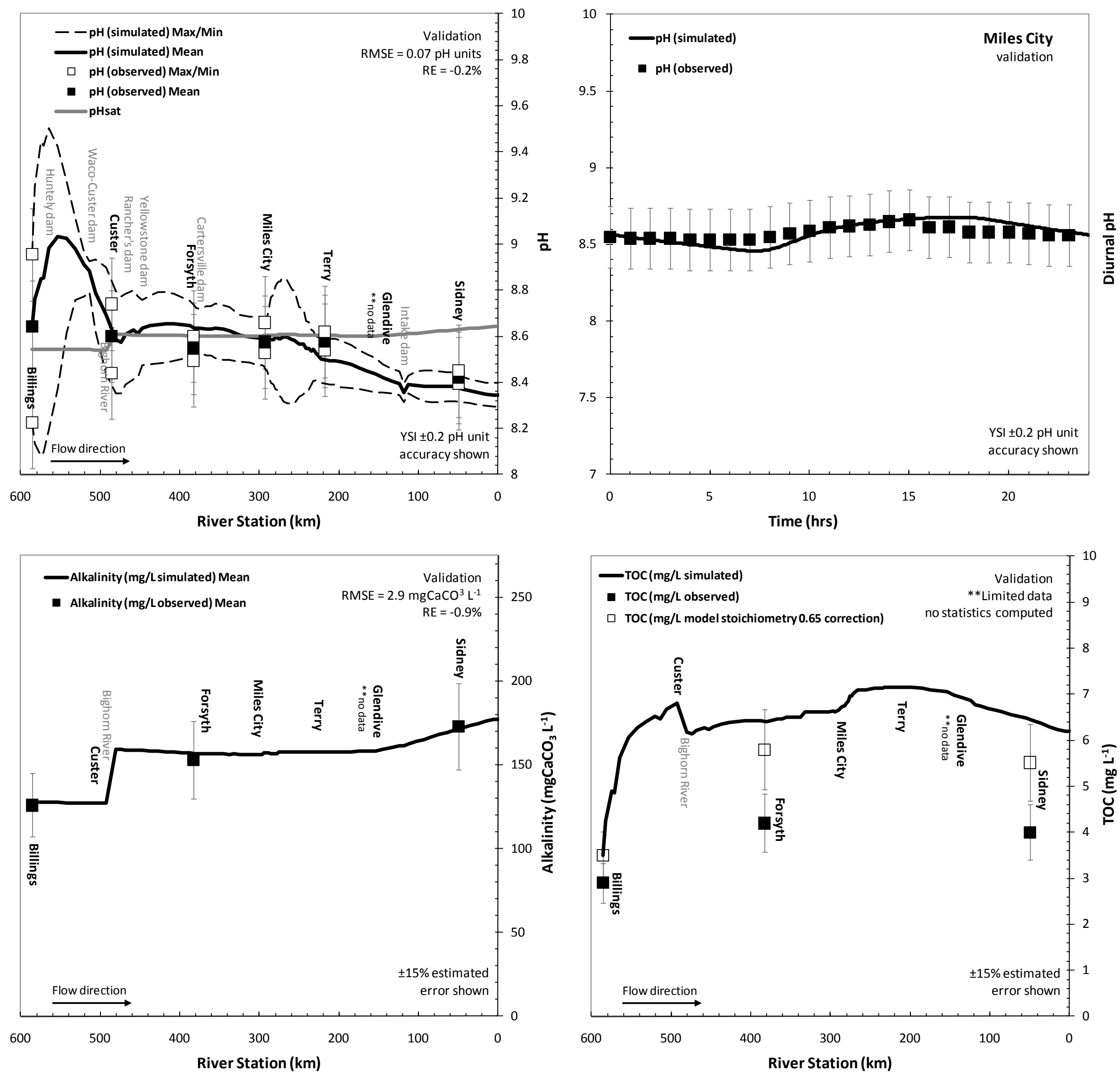


Figure 11-11. pH, alkalinity, and TOC simulations for the Yellowstone River during 2000.
(Top left/right panel) Simulated and observed longitudinal pH and diurnal pH simulation for Miles City. (Bottom left/right panel) Simulated and observed alkalinity and total organic carbon (TOC, as detailed in the next section).

11.3.8.2 CBOD and TOC

Little information exists to make CBOD or TOC comparisons. In 2000, three sites (Billings, Forsyth, and Glendive) had carbon-related variables measured. These included dissolved and particulate organic carbon (USGS pcode 681 and 689) that together sum to form TOC. Comparisons of TOC are presented in **Figure 11-11** (Bottom right panel) with the caveats identified previously in **Section 10.4.5** (regarding the fact that TOC is a calculated variable in and other stoichiometric issues related to the inflation of carbon from detritus).

11.3.9 Conductivity

Similar to the previous section, conductivity was used as a final estimate of model validity. The conductivity simulation for the river is shown in **Figure 11-12** and is very reasonable. The only major change occurred at the Bighorn River.

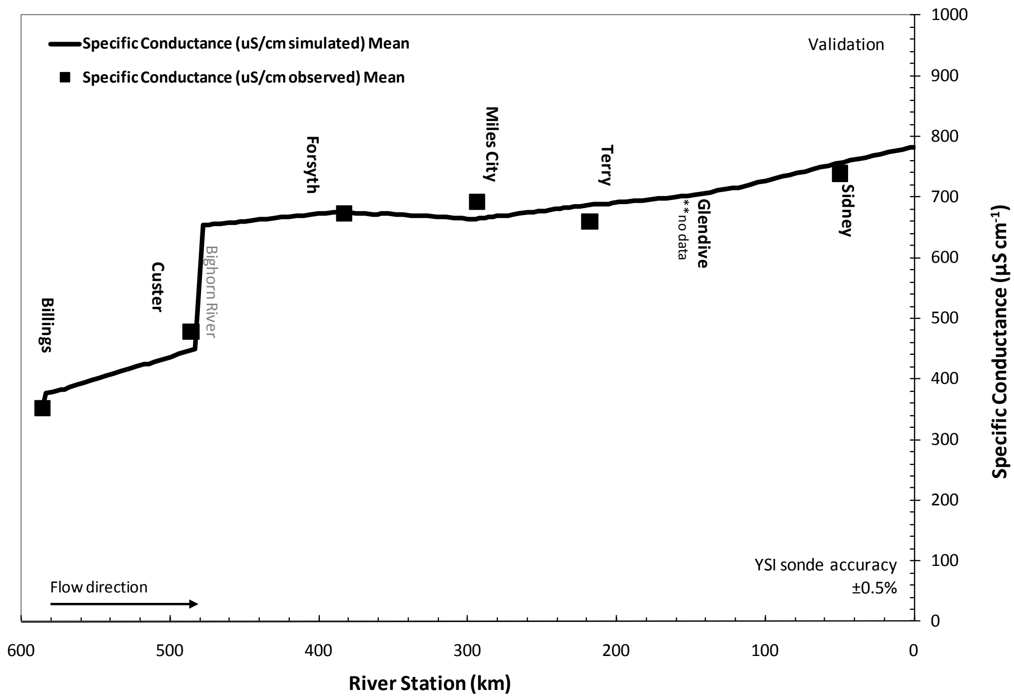


Figure 11-12. Simulated and observed conductivity for the Yellowstone River during 2000.

11.4 Some Final Thoughts on the Model

Based on this second evaluation (i.e., the cross-validation), we conclude that the Q2K model of the Yellowstone River meets the acceptance criteria specified in the project QAPP, or alternative recommendations from the literature. As such, the model is valid for use for nutrient criteria development. However some caveats do apply, specifically in regard to the time-period that the model is appropriate.

Given the conditioning detailed in previous section, we feel that the model is valid only to those circumstances encountered in model development, in particular when river productivity is near its peak during low-flow. Thus it should not be applied to high-flows (we did not apply or test the model against high flow conditions), late-season fall condition where algal growth is beginning to senesce (such as observed in our late September data), or any other condition outside the calibration and cross-validation described previously. It could perhaps be expanded to include months where the river has settled into a state of hydrologic and thermal stability during the growing season (but not beyond). Likewise, a relatively useful range of different soluble nutrient conditions were evaluated over the longitudinal extent of the model (e.g., from 5-105 µg L^{-1} for nitrate and 3-17 µg L^{-1} for SRP) during model development. This greatly enhances the biogeochemical predictability of the model over the critical time-period, albeit the spatial extent of this understanding was much greater for N than P.

12.0 Critical Low-flow Design Conditions for Nutrient Criteria

Critical low-flow conditions and the design climate for criteria development are described in this section. The logic behind this information and supporting details are found in the following sections.

12.1 Design Flows for Water Quality Modeling Studies

DEQ currently uses a seven-day, ten-year design flow (7Q10) to establish Montana Pollutant Discharge Elimination System (MPDES) permits (ARM 17.30.635). Dilution requirements for this critical low-flow require that existing water quality standards, including those linked to nutrients (e.g., benthic algae, dissolved oxygen, pH, etc.) be in accordance with use support requirements. Flow-based designations such as the 7Q10 are a common water quality practice and are used by most states. Recommendations largely stem from a single source, *"Technical Guidance for Performing Wasteload Allocations, Book VI, Design Conditions, Chapter 1 Stream Design Flow for Steady-State Modeling"* (EPA, 1986b).

However, the intent of the 7Q10 was for regulation of toxic substances, where the "7" reflects the flow duration over which the concentration in question is averaged, while the "10" reflects the frequency of allowable excursions from the criterion (i.e., once every 10 years). In theory, allowable excursions should be infrequent enough to allow the aquatic community to recover in the interim years. Although the 7Q10 has a long history of use, it was only an interim recommendation (U.S. Environmental Protection Agency, 1985). Preference for site-specific, biologically-driven approaches were rather given (by EPA) based on criteria continuous concentrations (CCC). The CCC is the highest concentration of a pollutant to which aquatic life can be exposed for an extended period of time (4 days) without deleterious effects (i.e., a chronic impact). The intent is that the 4-day average concentration should not exceed the CCC more than once every 3 years to allow sufficient recovery time.

The use of dynamic models to predict the frequency and duration events exceeding the CCC was originally envisioned by EPA (1985). The data requirements and model complexity make this approach very limited. As such, they offered the 7Q10 as an approximate surrogate for the 4-day/3 year biological (4B3) after doing a comparison of the hydrologically-based 7Q10 and 4B3 flows for a set of 60 rivers in the U.S. (U.S. Environmental Protection Agency, 1991). It was concluded that the relation between the two were acceptable, but generally the hydrologically-based approach allowed somewhat more excursions than the biological approach (U.S. Environmental Protection Agency, 1991).

Both methods continue to be recommended by EPA. However, Montana currently uses the hydrologically-based approach (ARM 17.30.635). Given that the 7Q10 has never really been vetted for nutrients, we explore more suitable design conditions as directed in ARM 17.30.635. Per ARM 17.30.635(2), "The Department shall determine the acceptable streamflow for disposal system design for controlling nitrogen and phosphorus concentrations". This work is described below.

12.2 Identifying an Appropriate Design Flow Duration

Methods to identify design flow durations for large rivers are detailed herein.

12.2.1 Algal Growth as an Indicator of Time to Nuisance Biomass

Algal growth rates govern the time required to reach nuisance biomass which precedes all attendant eutrophication responses. Hence we used net accumulation rates as a way to establish design flow durations for nutrient control on large rivers. The decision was based on a number of factors including their direct relevance to eutrophication, the fact that they are well reported in the literature, and that they are easily measured. Our position is that the design flow should be protective of water quality over the same duration that it takes that waterbody to reach an adverse response from nutrient loadings.

To help conceptualize this understanding, DEQ considered work by Stevenson et al., (1996). Key points of biomass accrual include peak biomass (PB) and time to peak biomass (T_{PB}) (**Figure 12-1)** which are influenced by colonization, exponential growth, and autogenic sloughing and loss phases. Since our interest is nuisance biomass, we defined a new ordinate and abscissa in the accrual phase curve called nuisance biomass (NB) and time to nuisance biomass (T_{NB}), which occurs somewhere between initial colonization and PB. For any effective nutrient control strategy, algal biomass must be less than or equal to NB to restrict nuisance growth and meet water quality standards. Hence by default NB must equal PB. For our purpose we define nuisance levels as those identified in Suplee et al., (2009).

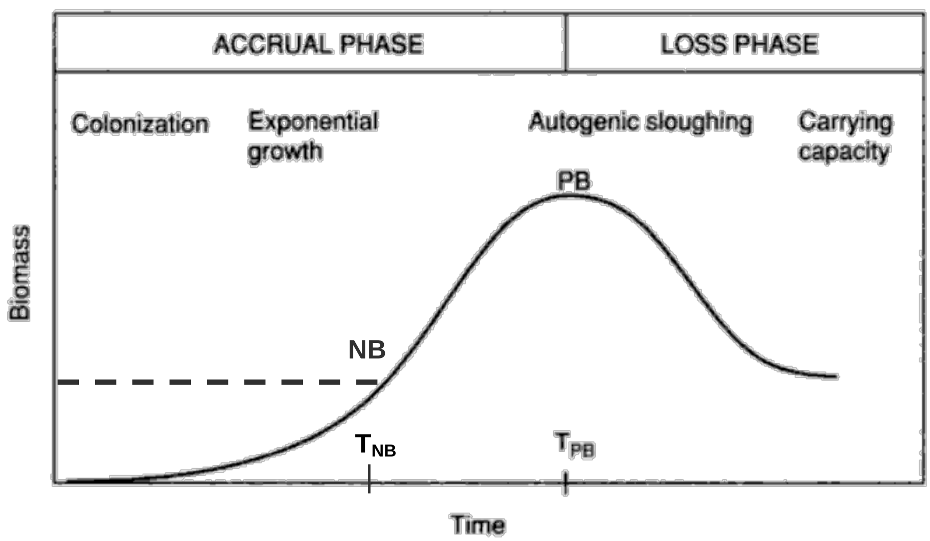


Figure 12-1. Idealized benthic algae growth curve.
Reproduced from Stevenson et al., (1996). Modified to include nuisance conditions. NB = nuisance biomass, TNB = time to nuisance algae , PB=peak biomass, TPB=time to PB from colonization.

For illustrative purposes, the accrual portion of the growth curve described previously can be modeled using an exponential growth equation (**Equation 12-1**) with space limitation (**Equation 12-2**) as shown in Chapra et al., (2008), where *Chla* = biomass at day *t* (mg Chl*a* m^{-2}), a_b = initial biomass (mg Chl*a* m^{-2}), *k* = the growth rate (day $^{-1}$), *t* = time (days), ϕ_{Sb} = a space limitation factor (dimensionless), and $a_{b,\max}$ = maximum carrying capacity of biomass (mg Chl*a* m^{-2}). Given a known relative specific growth rate (i.e., measured in either the field or the laboratory) and maximum carrying capacity [which is also well characterized in the literature, see Horner et al., (1983)], T_{NB}, *PB*, and T_{PB} can all be readily estimated.

(Equation 12-1) $$Chla = a_b \times \phi_{Sb} \times \exp^{kt}$$

(Equation 12-2) $$\phi_{Sb} = 1 - \frac{a_b}{a_{b,\max}}$$

The equations above can subsequently be used to describe algal growth kinetics as detailed in the next section.

12.2.2 Enrichment Studies Detailing Algal Growth Kinetics

To estimate a plausible timeframe to reach nuisance conditions in large rivers, we compiled as many field studies as we could that had time-variable algal biomass measurements in response to nutrient enrichment. Previous work (Horner et al., 1983; Stevenson et al., 1996) shows that peak biomasses can be achieved in as little as two weeks, or as long as two months, depending on relative specific growth rates. Hence the time to nuisance biomass is likely quite variable and system specific. The magnitude of P_B is also believed to vary, ranging from 300-400 mg Chl*a* m^{-2} Chl*a* for diatoms (Bothwell, 1989), to >1,200 mg Chl*a* m^{-2} for filamentous algae like *Cladophora* (Stevenson et al., 1996).

While the methodologies of identified studies vary, those with reliable and reproducible indicators of relative algal growth rates (and multiple algal collections over time) were of primary interest. A final requirement was that published studies must have water temperature data so that we could make corrections to standard reference temperature (20˚C). Work conducted under moderate enrichment conditions (similar to our modeled nutrient-addition scenarios described later), which met the specified criteria mentioned previously are shown in **Table 12-1**.

Table 12-1. Enrichment studies and associated growth rates adjusted to 20 degrees C.
Growth rates are corrected to the reference temperature using the Arrhenius equation.

Algae Type	Net Specific Growth Rate at 20˚C (k, day^{-1})	Reference	Location	Comment
Diatoms	0.50	Klarich (1977)	Yellowstone River, MT	Near Huntley Billings WWTP
Diatoms	0.55	Bothwell and Stockner (1980)	McKenzie River, OR	5% kraft mill effluent
Cladophora	0.71	Auer and Canale (1982)	Lake Huron, MI	Harbor Beach WWTP
Green algae	0.52	Horner et al., (1983)	Lab Flume	Laboratory N & P addition
Diatoms	0.42	Bothwell (1985)	Thompson River, BC	Downstream of WWTP
Diatoms	0.62	Bothwell (1988)	S. Thompson River, BC	Flume with N & P addition
Diatoms	0.58	Biggs (1990)	South Brook, New Zealand	Downstream of WWTP
Diatoms	0.45	Stevenson (1990)	Wilson Creek, KY	Agricul. stream after spate

Adjusted growth coefficients (k, day^{-1}) are very consistent and have a mean of 0.55 ±0.09 day^{-1} (95% confidence level). When applied to **Equation 12-1** and **Equation 12-2**, they suggest that T_{NB} would be on average 14 ± approximately 3 days under enriched conditions[82] (**Figure 12-2**).

[82] An initial biomass of 0.1 mg m^{-2} Chl*a* was assumed in all calculations. Times to nuisance biomass range from approximately 11-17 days based on the studies evaluated.

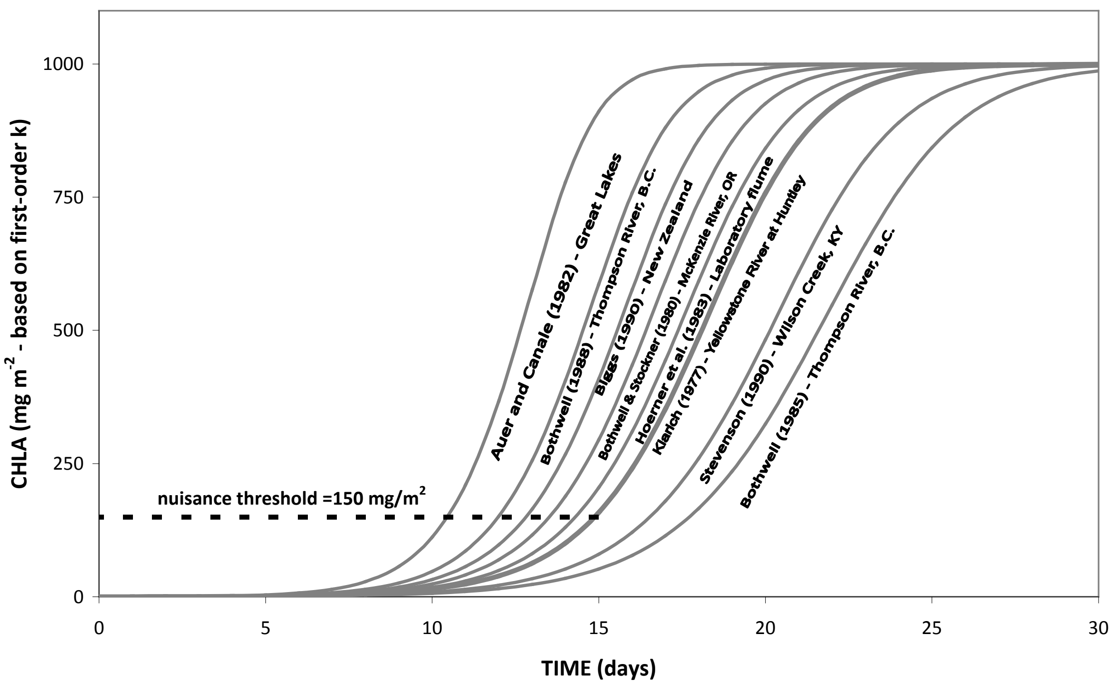


Figure 12-2. Estimated time to nuisance algal biomass under moderately enriched conditions.
Each curve was generated using the *k* value reported in **Table 12-2**. To be consistent with the study completed on the Yellowstone river by Klarich (1977)[83], an estimate of 14 days was believed to be an appropriate (slightly protective) design flow duration estimate. It was also within the margin of error of the original estimate of all studies (±3 days).

12.2.3 Justification of Time to Nuisance Algae Estimate

The time to nuisance biomass estimate described previously is not without problems and warrants a discussion. The most uncertain part of the estimate is whether the algal growth rates identified in the literature are suitable for criteria development for the Yellowstone River. If a proposed criteria induces a lower level of enrichment than detailed in the literature, a reduction in relative growth rate would be ensue. This would extend the time to nuisance biomass and lengthen the associated design-flow duration. The general consensus from the literature and site-specific data from both the Yellowstone River and a nutrient enrichment study in eastern Montana[84] suggest a 14-day duration design flow is appropriate. One could perhaps argue that this estimate is artificially fast given we cannot characterize

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[83] This Klarich (1977) work was completed in the Billings area (Huntley site downstream of Billings) using diatometers, which are glass slides placed in the river over a specified period of time. The most productive of all locations was downstream of the Billings WWTP, hence it was believed to be a good estimator or algal growth rates under enriched conditions.

[84] This was a recent stream fertilization (nutrient addition) study completed by DEQ on a similarly turbid waterbody in eastern Montana (DEQ, 2010). In this work, peak algal biomass at the most dense location in the study reach occurred 14-20 days after N and P dosing began (peaking at 1,092 mg Chl*a* m^{-2}) and was documented by photo series and by measurement of benthic Chl*a* several times. The biomass peak was filamentous algae, not diatoms. Mean stream water temperature over the time period was 21.8°C (16.2°C min., 28.9°C max.), very close to our reference temperature of 20°C.

the extent of the enrichment. Clearly PB approaches equality with NB as growth rates reduce. However this argument could be countered with the assumption that our initial starting biomasses used in constructing the growth curve (0.1 mg Chl*a* m^{-2}) was too low (i.e., probable standing crops of algae in late summer would be much higher, more like 5-40 mg Chl*a* m^{-2}). As a result, 14 days to NB is a reasonable (neither overly conservative nor overly liberal) duration for nutrient control.

Finally, it should be mentioned that the idealized growth curve described previously doesn't really exist. Rather, some approximate form of it occurs, in which the growth rate is continually adjusting to the varying continuum of light, temperature, and nutrients over time. Consequently, algal biomasses once established may have more to do with prior river conditions (e.g., a result of luxury uptake of nutrients), than conditions observed at the exact time of monitoring. We have selected a time of stable conditions for criteria development to hopefully minimize this disconnect, reflecting a period of optimal growth (warm temperature, stable flows, good light conditions, etc.)
.

12.3 Frequency of Low-flow Occurrence on the Yellowstone River

We have modified the design flow to a 14Q5 (1 in 5 year low-flow condition) to better align with EPA recommendations on allowable frequency of exceedance of standards which were originally based on a biologically-based 4-day average flow once every 3 years (i.e., 4B3). Having independently derived the 14 day duration as appropriate for constraining nuisance algae growth (**Section 12.2.3** above), we needed to determine the allowable frequency of exceedance. Once every three years is the basis for U.S. EPA chronic aquatic life criteria (U.S. Environmental Protection Agency, 1985), and since nutrient impacts are roughly analogous to chronic impacts (as opposed to acute ones), once every five years was ultimately selected for nutrient criteria (ergo, the 14Q5 flow).

In consideration of the proposed design flow, it is slightly protective, thus it addresses the concern that the 14 day duration may be too liberal (given that benthic algae in moderately enriched rivers would rarely begin at a base biomass as low 0.1 mg Chl*a* m^{-2}). Likewise it is consistently calculated and reported by USGS (e.g., McCarthy, 2004). The latter makes the duration-frequency selected practical for NPDES permitting where the seasonal period of July 1 – September 30 coincides with the growing season defined in Suplee et al., (2007). The final period of application of these criteria may differ somewhat from this period once adopted into law. This is to ensure adequate water quality protection during years when warm, stable conditions extend into October (as was observed in October 2012), albeit the way the statistic is calculated (July-September) will not change.

To characterize typical low-flow conditions on the Yellowstone River, 41 different seasons of low-flow data were evaluated over the period of 1968-2008 (**Figure 12-3**). We found that recent years (2000-2008) contained the 1st, 2nd, 3rd, 4th, 7th, 8th, 9th, and 10th lowest 14-day flows over the period of record (**Figure 12-4**, left) which suggests non-stationarity in streamflow statistics. Fortunately USGS is currently compiling new values. The 14-day low-flow period occurred most frequently (≈60% of the time) between the third week in August and first week of September (**Figure 12-4**, right) and thus this is a primary period of interest in evaluating river response.

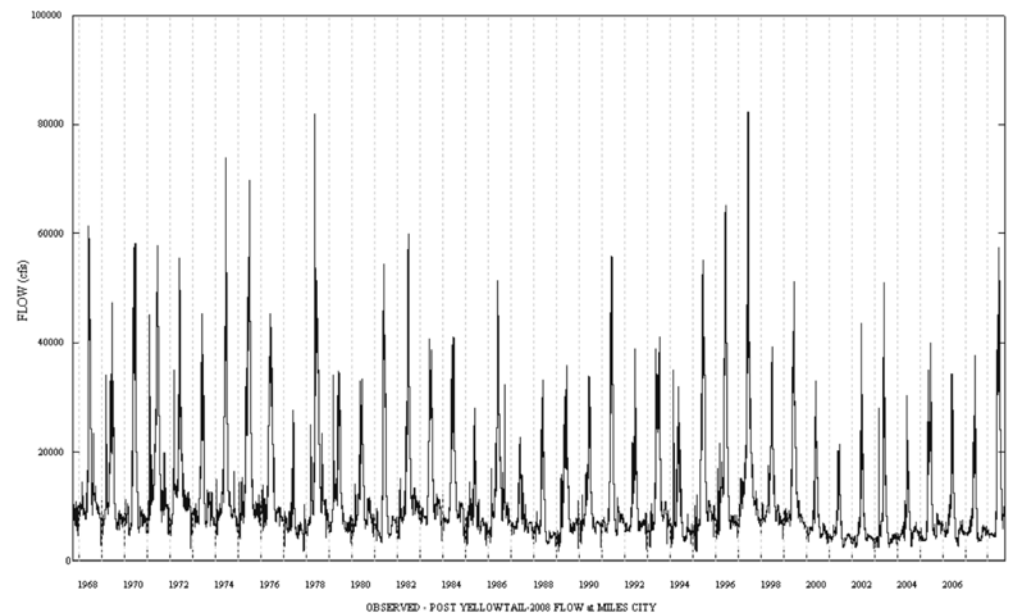

Figure 12-3. Period of record used in low-flow frequency at Miles City.

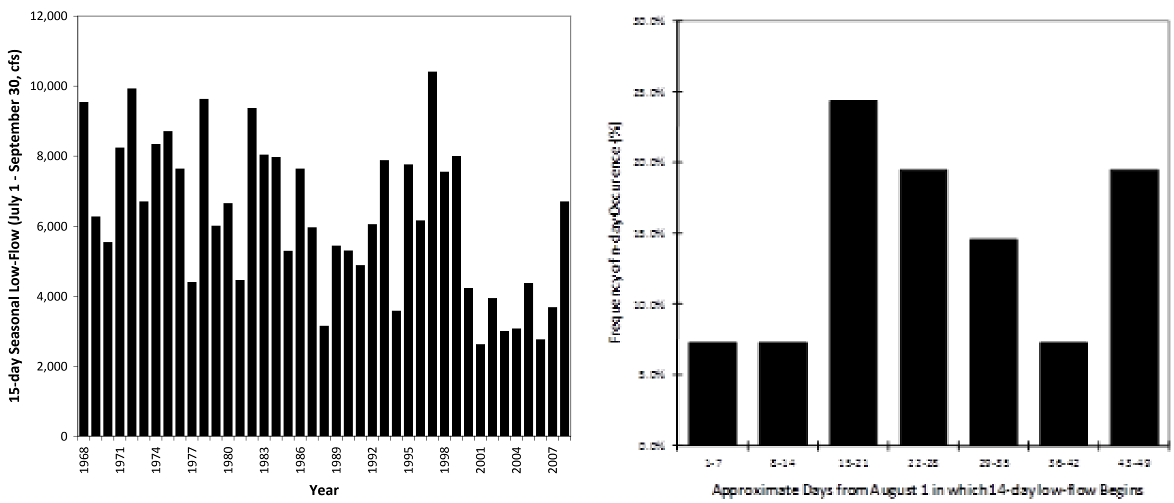

Figure 12-4. Low-flow analysis for Yellowstone River at Miles City (1968-2008).
(Left) Annual 14-day seasonal low-flow data over the period of record at Miles City. (Right) Number of days following August 1 in which the 14-day low flow began at Miles City (originally calculated using 15 days).

12.4 14Q5 Design Flow for the Yellowstone River

Low-flow duration and frequency statistics on the Yellowstone river have significantly changed in recent years (**Table 12-2**); the principal difference being the inclusion of six years of additional low-flow data. A preliminary update for low-flow frequency has been completed USGS (provisional data) and the updated 14Q5 for the new period of record (1966-2009) is 118.652 m^3s^{-1} (4,190 ft^3s^{-1}) (P. McCarthy, personal communication) which is significantly different than the 14Q5 reported earlier by McCarthy (2004) for the period of 1968-2002 at Miles City [134.5 $m^3 s^{-1}$ (4,750 $ft^3 s^{-1}$)[85]]. Also shown, are estimated 14Q5s for other gaged sites in the project reach (i.e., Forsyth and Glendive) which were estimated using a scaling factor based on the ratio of the mean discharges over a common period during August and September (2003-2007)[86]. The scaling factor was very close to 1.0 for all sites, and was estimated for Terry.

Table 12-2. Comparative summary of 14Q5 low-flow analysis for the Lower Yellowstone River.

Location	USGS 14Q5 (1968-2002) $m^3 s^{-1}$ ($ft^3 s^{-1}$)	Aug-Sep MAD (common period) $m^3 s^{-1}$ ($ft^3 s^{-1}$)	Scale factor	USGS Provisional 14Q5 $m^3 s^{-1}$ ($ft^3 s^{-1}$)
Forsyth	n/a	119.9 (4,234)	1.009	119.720 (4,230)
Miles City	134.5 (4,750)	118.8 (4,195)	1.000	118.652 (4,190)
Terry	n/a	n/a	1.005[EST]	119.245 (4,210)
Glendive	n/a	120.1 (4,240)	1.011	119.957 (4,240)

DEQ is recommending the use of the 14Q5 for all nutrient criteria design flows. It is commonly reported by USGS, is very close to the suggested duration-frequency identified in our analysis, and is a period over which we believe the regulated community will ultimately be able to control their waste-treatment process. Therefore in the criteria development for the Yellowstone River, we used the provisional 14Q5 of 118.652 $m^3 s^{-1}$ (4,190 $ft^3 s^{-1}$) at Miles City which has recently been determined by USGS (personal communication, P. McCarthy). This translates to a headwater flow of 119.720 $m^3 s^{-1}$ (4,230 $ft^3 s^{-1}$)[87].

12.5 Design Climate

The design climate for the criteria analysis is described in this section.

12.5.1 Climatic Conditions Associated with the 14Q5

Climatic conditions coincident with the 14Q5 are required for criteria development. It would be inappropriate to apply meteorological information outside of that context. To some degree, summer weather conditions (or climate in the context of long term weather averages) are independent of streamflow, especially in a river like the Yellowstone whose flow depends to a large extent on the prior winter's snowpack. As a result, low-flows do not necessarily depend on summer climatic conditions and therefore an underlying climatic series is needed to go along with the assigned design flow. To ensure

[85] It should be noted that when applying the Miles City design flow in combination with the scaled headwater boundary conditions, we could not exactly achieve the specified design flow at Miles City and Glendive. Rather there was some variation around the true value at each site (±5%) due to differences between the statistic and the actual water balance. We will incorporate this ±5% variance into the uncertainty analysis.

[86] Use of different periods of record would result in inconsistent low-flow frequencies between the sites. As a result, a scaling factor was proposed by DEQ whereby 14Q5 discharges at Forsyth and Glendive were identified using the ratio of the August-September mean annual discharge at Miles City from its 14Q5. The scaling factors were computed over a common period of record of low flows (2003-2007).

[87] It should be noted that when applying the Miles City design flow in combination with the scaled headwater boundary conditions, we could not exactly achieve the specified design flow at Miles City and Glendive. Rather there was some variation around the true value at each site (±5%) due to differences between the statistic and the actual water balance. We will incorporate this ±5% variance into the uncertainty analysis.

that we maintain the 20% recurrence interval (as implied in the selected 5-yr streamflow condition), a 1-yr climate is required[88].

We have already shown that the 14Q5 low-flow condition can occur most any time during the seasonal low-flow calculation period (e.g., July 1 – September 30). Most frequently though, it occurs during the 3rd and 4th week of August as shown in **Figure 12-4** (Right panel) which means we should apply the climatic conditions from that period to our analysis (i.e., August 14-28th). The only challenge is finding an unbiased daily estimator of this period. Because any selection by DEQ may be considered preferential, and period-based averages are also in-appropriate (i.e., they tend to mute diurnal variation), we used an independent data source to develop the design climate as described in the next section.

12.5.2 Typical Meteorological Year

A typical meteorological year (TMY) is a pre-determined dataset containing hourly meteorological values that typify a location over a longer period of time (in most cases 30 years). The National Renewable Energy Laboratory (NREL) currently publishes one such dataset which includes stations specific to our project area (i.e., Miles City and Glendive, MT[89]). We used the information from the 1976-2005 TMY to develop the design climate consistent with the most probable low-flow period. The data consists of 12 typical meteorological months (January through December) that are concatenated together without modification to form a single year of serially complete data (NREL, 2007; Wilcox and Marion, 2008). Missing data are filled or interpolated when necessary, giving natural diurnal and seasonal variation.

The TMY selection method involves identifying representative individual months from different years judged to be most typical per the TMY algorithm. Nine daily weighted indices are used which include: (1) dry bulb and dew point temperature (minimum, maximum, and mean for each); (2) maximum and mean wind velocity; and (3) total global horizontal solar radiation. Weightings are: 10/20 on radiation, 4/20 on air temperature, 4/20 on dew point, and 2/10 on wind velocity. Given the interdependence of many of these variables, the TMY is a good approximation of expected climatic conditions. Because adjacent months in the TMY may be selected from different years, discontinuities can potentially occur. Six hours on each side of the month are smoothed to accommodate this difference (NREL, 2007; Wilcox and Marion, 2008). An example TMY series of temperature for Miles City is shown in **Figure 12-5**.

[88] A climatic condition with prob**a**bility of 1.0 is required (i.e., 100% chance that this climate condition would happen every year) to ensure that the 20% chance non-exceedance probability of the low-flow condition is maintained (i.e., to not alter the overall frequency of occurrence). In other words, the probabilities are multiplicative, and a 0.20 streamflow probability multiplied by 1.0 climate probability is still a 0.20 chance occurrence (or 5-year) event.

[89] The two TMY datasets available for our project site are: 742300 Miles City Municipal Airport and 726676 Glendive AWOS. The Miles City site had 22 years of candidate data (1976-2005), which excluded six years influenced by volcanic eruptions at El Chichón in Mexico in 1982 and Mount Pinatubo in the Philippines in 1991, as well as two years of missing data (i.e., 22/30 years were considered). The Glendive station only had 12 candidate years of record, therefore was not suitable for our analysis.

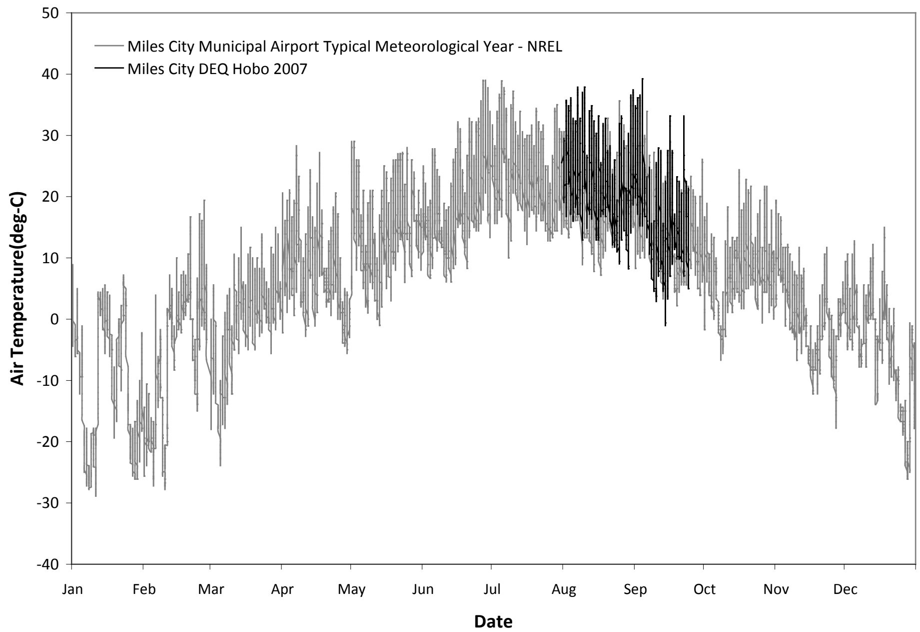

Figure 12-5. Example TMY air temperature plot for 742300 Miles City Municipal Airport. Field observations from 2007 are shown for reference purposes.

12.5.3 Adjustments to the TMY Based on Field Observations

As indicated previously (**Section 7.4**), the Miles City Municipal Airport (APT) is not sufficiently representative of the river corridor. To better approximate river conditions we used the corrections shown in **Table 12-3** (next page). These were determined through paired station analysis and indicate that the river had consistently lower wind speed and higher dew point than the APT. The disparity was due to the fact that the airport is located on a bluff adjacent to the river and experiences different climatic conditions than the river itself. Plots of adjusted TMY data are shown in **Figure 12-6** and compare very similarly to conditions during 2007.

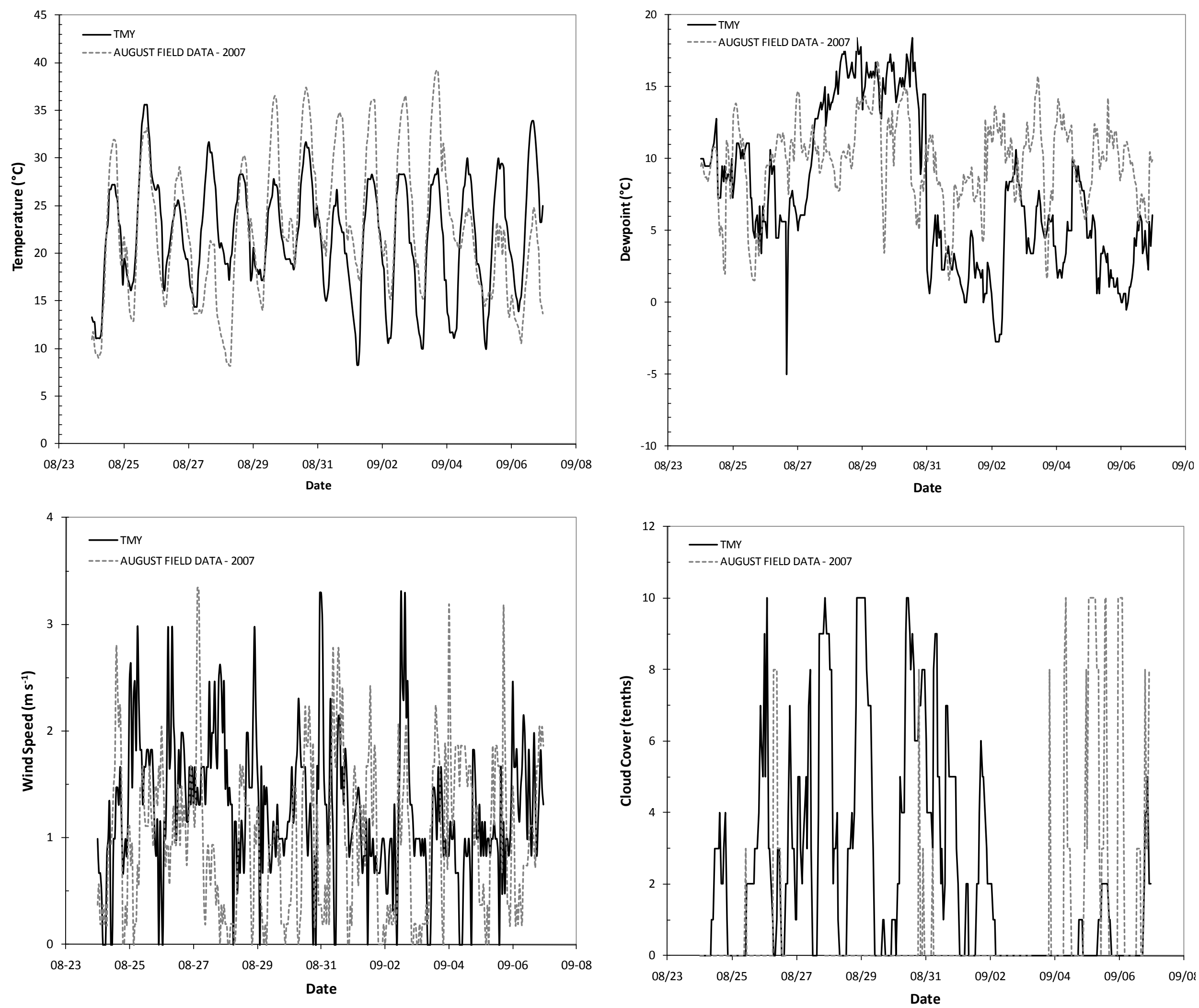

Figure 12-6. Plots of adjusted TMY in comparison with August 2007 field data.
(Top left/right) Air temperature and dew point temperature (˚C). (Bottom left/right) Wind speed (m s^{-1}) and cloud cover (tenths). Adjustments are based on **Table 12-3**.

Table 12-3. TMY adjustments based on paired station analysis from August 1- September 21, 2007.

Climatic Variable	Miles City Municipal APT	DEQ Hobo Site (on island in river)	Adjustment Factor
Temperature	21.0	21.0	0.0 degrees C
Dew Point	6.5	8.2	+1.7 degrees C
Wind Speed @ 10 m	4.2	1.1	x 0.32 m/s
Cloud Cover	0.15	N/A	none

12.5.4 Extrapolation of TMY Data to the Other Climatic Regions in the Project

The adjusted TMY data from Miles City were than extrapolated to other regions in the river corridor based on the long-term relationships from Hydmet (2009). Associated averages and adjustment factors for the four climatic zones used in our model (i.e., Sweeney Creek, Miles City APT, Terry AgriMet, and Glendive AgriMet) are shown in **Table 12-4**.

Table 12-4. TMY adjustment factors for climatic regions used in the Q2K model.
Data from 1999-2008. Miles City APT site already adjusted according to the factors in **Table 12-3**.

Location	Air Temp. (˚C)	TMY Adjust (˚C)	Dew point Temp. (˚C)	TMY Adjust (˚C)	Wind Speed @ 7 meters (m/s)	TMY Adjust (m/s)	Cloud Cover (tenths)	TMY Adjust (tenths)
Sweeney Creek	19.5	+0.2	8.1[1]	+0.2	2.8	No adj.	Same	No adj.
Miles City APT	19.3	---	7.9	---	1.3	---	---	---
Terry	18.4	-0.9	7.3	-0.6	2.4	No adj.	Same	No adj.
Glendive	17.8	-1.5	7.5	-0.4	2.4	No adj.	Same	No adj.

[1] Sweeney Creek dewpoint not consistent with other locations. Used ratio between Sweeney Creek and W7PG-10.

In summary, the overall trend in the dataset seems to be:

- A slight longitudinal cooling effect with air temperature from Forsyth to Glendive. This was confirmed by a secondary data source (PRISM Climate Group, 2006).
- Fairly consistent dew point at all locations, except at Sweeney Creek, where it was higher.
- Much higher wind speeds at Sweeney Creek, Terry, and Glendive than in Miles City.
- Inconclusive information on cloud cover.

Of all climatic variables, wind speed was most interesting due to the large difference between the adjusted Miles City site and that of the other sites. Evaluation of wind rose data provides some insight about the differences (**Figure 12-7**).The primary consideration appeared to be wind direction, and its relationship with river aspect. Sites most perpendicular to the river appear to have more wind sheltering than those in adjacent areas. For example, the DEQ Hobo and Miles City APT indicate a disproportionate number of percent calms (21% vs. < 6%), nearly three times greater in the river than at the airport. A shift in direction also occurred indicating eddy and turbulence effects.

There were also differences longitudinally. Percent calms were much lower at Terry and Glendive (≈1%) than at Miles City APT. This is at least qualitatively indicates that the lower river should be both windier and cooler than the upper river. Such assertions were verified through calibration of water temperature in the model and therefore the wind gradient was not altered except in the case of Miles City.

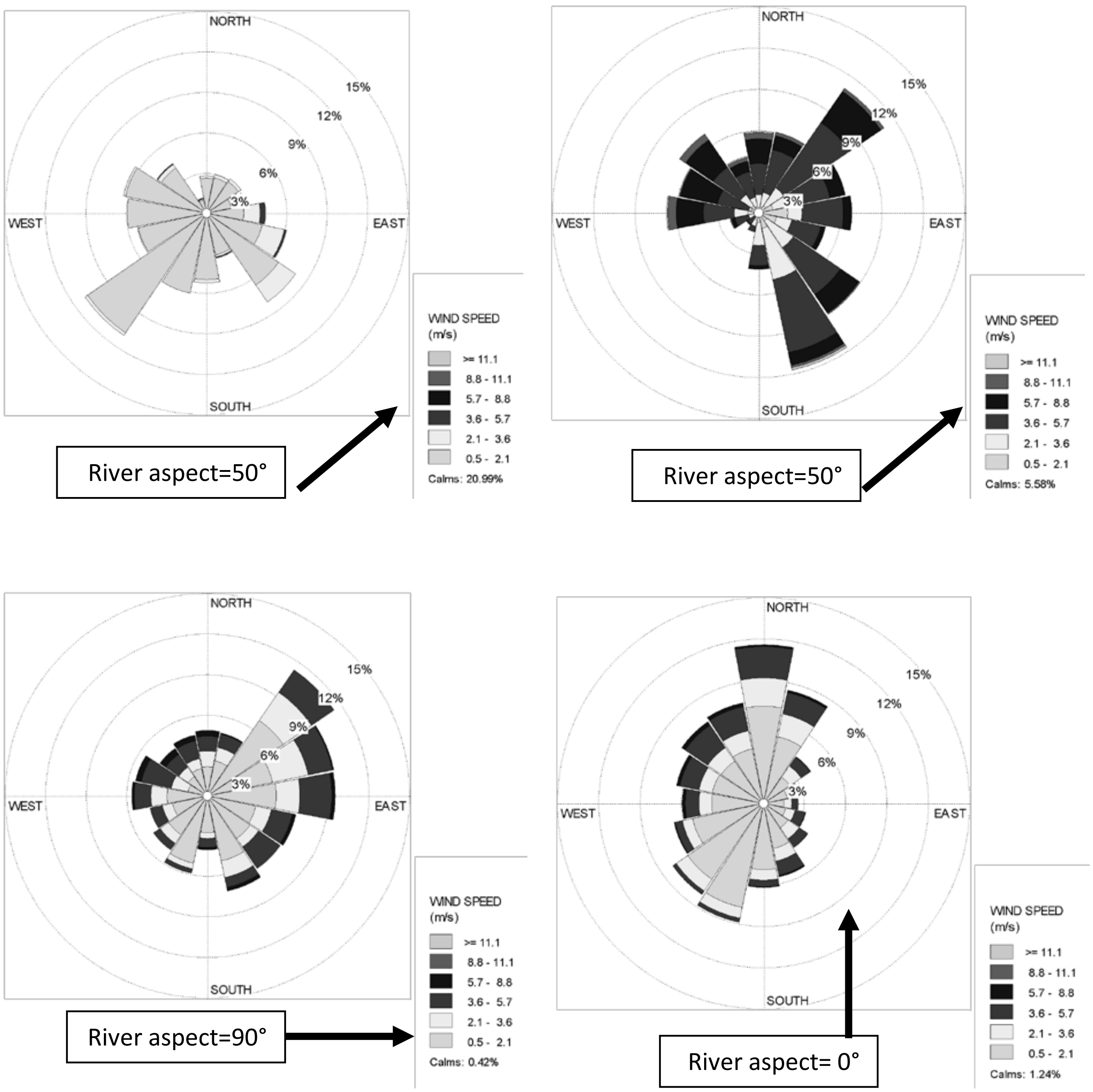

Figure 12-7. Wind Rose data on the lower Yellowstone River.

(Top left/right panel) Wind observations at the DEQ Roche Juan Island and Miles City APT. (Bottom left/right panel) Same but for Terry and Glendive AgriMet. Percent calms increased substantially between the river station and Municipal airport site. This illustrates the effect of sheltering by topography and vegetation. In cases where the wind vector was parallel to the river aspect, these effects were diminished.

12.6 Water Quality Boundary Conditions

Appropriate water quality boundary conditions must also be specified. Included are the headwater boundary condition (i.e., Forsyth), incoming tributary information, irrigation exchanges, etc. We compiled data from the ten lowest-flow years on record to attribute these features for the Yellowstone River. Some data was available most years and is shown in **Table 12-5**. Diurnal data was only available for two of the years (2000 & 2007). A direct average of the observations was applied in the model.

Table 12-5. Low-flow water quality summary for the Yellowstone River.
Data from USGS at Forsyth (headwater boundary condition of our study reach).

Low-flow ranking, time (am/pm), and date of observation (out of 41 years)	**Temperature (°C)**	**pH**	**SC (µS/cm)**	**DO (mg/L)**	**Alkalinity (mg/L)**	**TSS (mg/L)**	**TN (mg/L)**	**NO_2+NO_3 (mg/L)**	**NH4 (mg/L)**	**TP (mg/L)**	**SRP (mg/L)**	**Phyto (µg/L)**
Rank=1, 1200 pm August 21, 2001	23	8.4	805	9.6	161	18	0.47	0.05	<0.04	0.032	<0.02	n/a
Rank=2, 1216 pm August 9, 2006	26	n/a	596	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Rank=3, 1000 am August 21, 2003	22	8.3	701	8.2	129	22	0.48[E]	0.06[E]	<0.04	0.042	<0.02	n/a
Rank=4 1200 pm Sept. 8, 2004	n/a	8.4	n/a	n/a	145	37	0.52	0.15	<0.04	0.056	<0.006	n/a
Rank=5 0940 am August 30, 1988	18	n/a	945	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Rank=6 0310 pm August 30, 1994	18	n/a	673	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Rank=7_a 0430 pm August 18, 2007	23.5	8.7	762	9.0	170	26	0.519	0.102	0.015	0.041	0.003	8.0
Rank=7_b 0430 pm August 27, 2007	21.6	8.7	760	9.5	170	35	0.507	0.107	0.008	0.044	0.003	9.6
Rank=8 0900 am August 2, 2002[1]	20	8.3	540	8.2	130	62	0.74	0.36	<0.04	0.107	0.02	n/a
Rank=9_a 0300 pm August 16, 2000	22	8.9	636	9.5	134	18	n/a	<0.05	<0.02	n/a	<0.01	n/a
Rank=9_b 1200 pm August 26, 2000	21.2	8.5	676	7.5	n/a	58	0.39	<0.05	<0.02	0.031	<0.01	6.9
Rank=10 1231 pm August 9, 2005	22.5	n/a	590	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Averages[2]	**21.8**	**8.6**	**714**	**8.9**	**152**	**31**	**0.48**	**0.07**	**0.009**	**0.041**	**0.003**	**8.2**

[1]This data excluded from analysis (low-flow period was significantly after sampling date).
[2]For values below reporting limit (e.g., <0.02, etc.) use ½ detection limit which is ¼ the reporting limit.
[E] Estimated values

For variables less monitored (CBOD, ISS, detritus, etc.), relationships established during August and September 2007 were used. This included the scaling factors of ISS=0.8*TSS and detritus = 0.15*TSS (as described previously). For the diurnal data, much less information exists. Data from August 2000 and 2007 were used to establish appropriate ranges in field water quality variables (i.e., temperature, pH, and DO). These data are shown in tabular form in **Table 12-6**.

Table 12-6. Diurnal variation in low-flow water quality.

Period	Temp Range/2 (°C)	DO Range/2 (mg L^{-1})	pH Range/2 (pH units)	Conductivity Range/2 (µS cm^{-1})
August 26-28, 2000	1.15	0.84	0.05	n/a
August 17-26, 2007	0.95	0.87	0.06	n/a
Average	1.05	0.85	0.06	n/a

A sine function was used to distribute these values over the course of a day (**Equation 12-3**), where S_t=state variable at time t,

Table 12-7. Low-flow conditions on gaged tributaries to the Yellowstone River.

Yellowstone River low-flow ranking (out of 41 years)	**Year**	**06296003 Rosebud Creek at mouth near Rosebud Record=1975-2006 ($m^3\ s^{-1}$)**	**06308500 Tongue River at Miles City Record=1938-2010 ($m^3\ s^{-1}$)**	**06326500 Powder River near Locate Record=1938-2010 ($m^3\ s^{-1}$)**
1	2001	0.082 (2.91 $ft^3\ s^{-1}$)	0.600 (21.2 $ft^3\ s^{-1}$)	2.093 (73.9 $ft^3\ s^{-1}$)
2	2006	0.000 (0.00 $ft^3\ s^{-1}$)	0.365 (12.9 $ft^3\ s^{-1}$)	0.062 (2.19 $ft^3\ s^{-1}$)
3	2003	0.000 (0.00 $ft^3\ s^{-1}$)	2.685 (94.8 $ft^3\ s^{-1}$)	0.153 (5.42 $ft^3\ s^{-1}$)
4	2004	0.000 (0.02 $ft^3\ s^{-1}$)	0.838 (29.6 $ft^3\ s^{-1}$)	1.082 (38.2 $ft^3\ s^{-1}$)
5	1988	0.000 (0.00 $ft^3\ s^{-1}$)	1.147 (40.5 $ft^3\ s^{-1}$)	0.037 (1.3 $ft^3\ s^{-1}$)
6	1994	0.014 (0.50 $ft^3\ s^{-1}$)	0.617 (21.8 $ft^3\ s^{-1}$)	1.096 (38.7 $ft^3\ s^{-1}$)
7	2007	n/a	4.112 (145 $ft^3\ s^{-1}$)	7.439 (262 $ft^3\ s^{-1}$)
8	2002	0.078 (2.75 $ft^3\ s^{-1}$)	0.530 (18.7 $ft^3\ s^{-1}$)	0.736 (26 $ft^3\ s^{-1}$)
9	2000	0.000 (0.01 $ft^3\ s^{-1}$)	2.144 (75.7 $ft^3\ s^{-1}$)	0.272 (9.6 $ft^3\ s^{-1}$)
10	2005	0.007 (0.26 $ft^3\ s^{-1}$)	4.225 (149 $ft^3\ s^{-1}$)	1.767 (62.4 $ft^3\ s^{-1}$)
Low-flow Mean		**0.020 (0.72 $ft^3\ s^{-1}$)**	**1.726 (61.0 $ft^3\ s^{-1}$)**	**1.474 (52.0 $ft^3\ s^{-1}$)**
Long Term Mean		**0.204 (7.20 $ft^3\ s^{-1}$)**	**4.984 (176 $ft^3\ s^{-1}$)**	**5.805 (205 $ft^3\ s^{-1}$)**
% Long Term		**10%**	**35%**	**25%**

13.0 Water Quality Model Nutrient Additions to Identify Numeric Criteria

Nutrient additions were completed using conditions described previously in both Q2K and AT2K so that DEQ could determine appropriate nutrient thresholds for the Yellowstone River. A number of plausible water quality endpoints were evaluated including DO, pH, benthic algae, TOC, etc., (see **Figure 1-1**). The most limiting endpoint would become the driver for the numeric nutrient criteria (i.e., the one that would push the river into a state of non-compliance with a water quality standard first). The August 2007 parameterization was used for the analysis, which we felt was best suited toward low-flow conditions (and high productivity) when criteria apply. This was used in combination with information in **Section 12.0** to determine critical nutrient limits. Methodologies and findings are detailed in this section.

13.1 Methodology Used to Identify Critical Nutrient Concentrations

To resolve the water quality response of the river to nutrients, we adjusted soluble nutrient concentrations in the model until a water-quality limiting eutrophication response ensued (similar to what is done in a field dosing study but through the mechanistic relationships in the model). When considering the lower Yellowstone River, nutrient additions were required because concentrations are below those that impair uses (see **Section 12.0**). However, it should be noted that nutrient reductions could theoretically be necessary if the river were already in excess of state water quality standards (which link to eutrophication). If this would have been the case, DEQ would have run nutrient reduction scenarios instead.

The following reaches in the river were considered for criteria development to be consistent with previously established criteria assessment units (**Section 4.4)**:

- Forsyth to Powder River (reflective of Criteria Assessment Unit 3 –Big Horn to Powder river)
- Powder River to Glendive (reflective of Criteria Assessment Unit 4 –Powder River to state-line)

Two scenarios were evaluated for each reach: (1) a case where nitrogen (N) was limiting and (2) if phosphorus (P) were limiting. Effectively this covers all plausible outcomes and allows us to set control limits for both N and P over the growing season.

13.1.1 Form of Nutrient Additions and How They Were Introduced Into Q2K

Nutrient additions in Q2K were done through the adjustment of soluble nutrients in the model. Perturbation was completed so that both the headwater boundary condition and diffuse source accretion terms[90] maintained consistent soluble N and P concentrations across the modeling extent. Dosing increments used in each scenario are shown in **Table 13-1**, with one nutrient being set at non-

[90] A different diffuse term was used every 10 km.

limiting levels so that the other could be evaluated[91]. In all P evaluation scenarios, soluble N was set to 1,000 µg L^{-1} (a non-limiting level) whereas for all N evaluations, soluble P was set to 100 µg L^{-1} (again non-limiting). Wastewater inflows were also removed to create a more uniform nutrient profile.

Table 13-1. Soluble nutrient concentrations used to evaluate limiting water quality responses.

Trial	Nitrogen Limiting		Phosphorus Limiting	
	NO_3- (µg L^{-1})	SRP (µg L^{-1})	NO_3- (µg L^{-1})	SRP (µg L^{-1})
1	6	100	1,000	2
2	8	100	1,000	3
3	10	100	1,000	4
4	15	100	1,000	6
5	20	100	1,000	8
6	25	100	1,000	10
7	30	100	1,000	15
8	50	100	1,000	20
9	70	100	1,000	30
10	100	100	1,000	50

Output tables were then constructed for each scenario to identify thresholds where nutrient levels would most impact beneficial uses (e.g., pH vs. soluble N, DO standards vs. soluble P, etc.) thereby forming the foundation of our nutrient criterion for the river. Endpoints that apply to the Yellowstone River (all related to water use class B-3) are reiterated below and preface our analysis:

- **Dissolved oxygen**, which according to ARM 17.30.625 must not be reduced below applicable Circular DEQ-7 levels. For B-3 waters, instantaneous minima should be greater than 5 mgO_2 L^{-1} to protect early stages of aquatic life (DEQ-7).
- **pH**, where induced hydrogen ion concentration variation must be less than 0.5 units within the range of 6.5 to 9.0, and maintained without change if natural is beyond those limits to protect aquatic life. Natural pH above 7.0 must also be maintained above 7.0. (ARM 17.30.625). Further discussions regarding pH are contained within this section.
- **Algae**, whose benthic biomasses should be less than 150 mg Chl*a* m^{-2} to protect recreational use (Suplee et al., 2009). DEQ requires that the mean biomass of the wadeable region[92] not exceed this threshold in large rivers.
- **Total dissolved gas**, which should not exceed 110% of saturation (DEQ-7).
- **TOC**, whose removal is required at the levels shown in **Table 13-2** (EPA rule EPA 816-F-01-014)[93].

[91] The model operates on Liebig's Law of the minimum, where only a single nutrient can limit growth at any given time, thus both macronutrients (N and P) required consideration.
[92] Wadeable defined as ≤1 meter, (Flynn and Suplee, 2010), again see **Section 1.4**.
[93] Primarily, we are concerned with whether or not any scenario would push the river over a required treatment threshold (such as >8 mg L^{-1} if the waterbody was already in the 4-8 mg L^{-1} range).

Table 13-2. Required TOC removal based on EPA Stage 1 disinfectants and disinfection byproducts.
Based on EPA 816-F-01-014, June 2001.

Source Water TOC ($mg\ L^{-1}$)	Source Water Alkalinity ($mg\ L^{-1}$ as $CaCO_3$)		
	0-60	**>60-120**	**>120**
>2.0-4.0	35%	25%	15%
>4.0-8.0	45%	35%	25%
>8.0	50%	40%	30%

13.1.2 Upstream Boundary Condition Considerations

As mentioned previously, future conditions at our headwater boundary condition (Forsyth) will presumably change over time as the river is allowed to shift closer towards the identified criteria (recall that our model begins in the middle of criteria assessment unit 3 (**Section 4.4**) and any incremental increase in nutrients will alter water quality conditions at the beginning of our project reach). An approach is therefore necessary to forecast these changes. After much consideration, two methods were used.

First, for variables that have a direct relationship with total nutrients (such as phytoplankton Chl*a*), the literature was relied upon to estimate phytoplankton biomass changes that would occur with increasing nutrient levels. For other variables that have an unknown or indirect relationships with total nutrients (such as OrgN and OrgP, detritus, or other variables), the model was used to evaluate longitudinal buildup from ambient conditions and to prescribe a likely headwater condition that would minimize the gradient with respect to longitudinal distance (under the assumption that an equilibrium concentration could be achieved). These methods are better described below.

Phytoplankton concentrations increase longitudinally given sufficient nutrients and light. For example, recent studies show that water column Chl*a* can routinely reach concentrations of 70 µg Chl*a* L^{-1} in eutrophied rivers (Royer et al., 2008). Phytoplankton concentrations also correlate well with total nutrients. A relationship has been observed between TP and phytoplankton concentration by many authors (Basu and Pick, 1995; Basu and Pick, 1996; Basu and Pick, 1997; Heiskary et al., 2010; Van Nieuwenhuyse and Jones, 1996). One also exists with TN (Dodds, 2006). We can therefore estimate probable future phytoplankton values at our upstream study limit using one of the published equations (**Table 13-3**).

Among the studies evaluated, we selected the Dodds (2006) equation for TN and the Basu and Pick (1996) relation for TP. Dodds (2006) was used for the lack of better information (it was the only one we could identify for N)[94] and justification for use of the TP equation is as follows: (1) it was developed during summer conditions similar to what we evaluated on the Yellowstone River, (2) it applies to large northern temperate rivers and produced results very similar to those observed in the Yellowstone (i.e., in regard to observed TP and Chl*a* concentrations), and (3) its results fall in the midrange of the studies identified (**Table 13-3**). Hence it was a good fit to our project. For each intended nutrient-addition scenario, the total N or P concentration in question was applied to the appropriate equation and the

[94] The Dodds (2006) equation underestimated phytoplankton Chl*a* concentration relative to the actual measured TN/phytoplankton concentrations measured in the Yellowstone River. Therefore, we used a constant Chl*a* correction factor (Chl*a* result from Dodds (2006) x 2. 5 µg Chl*a* L^{-1}) to make the estimates.

resultant phytoplankton concentration was iteratively input into the headwater boundary condition until convergence was acheived prior to running the scenario.

Table 13-3. Published equations relating phytoplankton Chl*a* to TP or TN concentration.

Authors	Sampling Timeframe	River(s) Description	Equation[1]
Van Nieuwenhuyse and Jones (1996)	May-September	292 temperature streams, mainly tributaries to the Missouri and Mississippi	Log (Chl*a*) = -1.65 +1.99 (Log TP) - 0.28 $(\text{Log TP})^2$
Basu and Pick (1996)	July	31 large rivers (Strahler ≥ 5th order) in southern Ontario and western Quebec (flow range 0.9-250 $m^3\ s^{-1}$)	Log (Chl*a*) = -0.26 + 0.73 Log (TP)
Basu and Pick (1997)	May-October	Rideneau River, southern Ontario (mean annual flow 38.9 $m^3\ s^{-1}$)	Log (Chl*a*) = -0.62 + 1.02 Log (TP)
(Dodds, 2006)	May-September	Similar to Van Nieuwnehuyse and Jones (1996), but original data re-analyzed focusing on TN	Log (Chl*a*) = -1.25 + 0.68 Log (TN)
Heiskary et al. (2010)	Summer-early fall	>40 streams and rivers (Strahler order 4th-7th) in Minnesota, with summer flows from 1.8-233 $^3\ s^{-1}$)	Log (Chl*a*) = -1.82+ 1.47 Log (TP)

[1] All units (nutrients and phytoplankton) are in $\mu g\ L^{-1}$.

The second consideration was the remaining headwater conditions that would be affected by upstream changes in productivity. All state-variables will likely experience some alteration in the future. For example, a net increase in mean constituent concentration will occur such as in the case of detritus, OrgN or P, while others will show greater diurnal variability (such as pH and DO). For the purpose of our analysis, we were most concerned with shifts in the nutrient-related species, specifically, the headwater organic nutrient concentration and detritus (by-products of increased algal productivity.

Given the lack of suitable alternatives to characterize these variables, the model itself was used to evaluate buildup rates in the downstream direction and prescribe the most likely headwater condition that would minimize the change in those variables with respect to distance. We first achieved the target soluble nutrient concentrations in the model as shown in **Figure 13-1** (Left panel) and then iterated headwater conditions until an approximate threshold or flattening was observed with respect to distance[95] (**Figure 13-1**, Right panel). Once determined, these were used in the nutrient addition simulation as the anticipated change in the headwater boundary condition from upstream degradation over time[96]. In this illustration, only Criteria Assessment Unit 3 was evaluated. A similar exercise would be required for Unit 4, although in this instance the downstream boundary condition for Unit 3 is the upstream boundary for Unit 4.

[95] It should be noted that this also required additional iteration of diffuse source terms. Any change in headwater conditions alters soluble nutrients (more mass going through hydrolysis reaction).
[96] Again, the idea is that factors change upstream of our modeled reach as the river moves closer to the nutrient standard. Once a different criteria unit was encountered, a new condition could be expected.

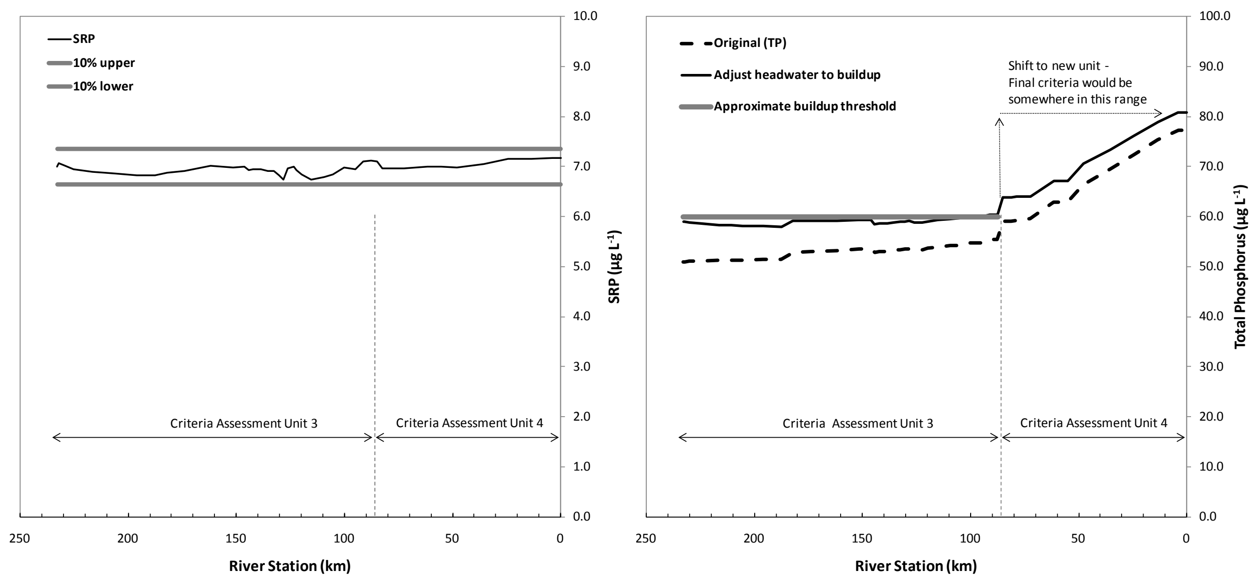

Figure 13-1. Example of the iterative procedure required to assign headwater boundary conditions. (Left panel) Soluble phosphorus simulation shown for an example TP optimization (in this instance a target SRP concentration was 7 µg L^{-1}). (Right panel) Modeled TP with (solid line) and without (dashed line) adjustment of headwater TP concentrations to reflect future eutrophied conditions (adjustment done through addition of organic P and phytoplankton P to reflect the approximate downstream buildup).

In some instances, the headwater change was very apparent. Detritus is one such example shown in **Figure 13-2** (Left panel). It is evident from the plot that adjusted conditions more closely reflect the continuum within the channel. Adjustments are not exact, however, but do not cause concern as initial condition error diminishes in the downstream direction[97]. The phytoplankton change (from literature review described previously) is also shown (**Figure 13-2**, Right panel). Less of a longitudinal equilibrium exists with this variable.

[97] Other factors become increasingly important including the effects of model rate coefficients, and boundary conditions at the air-water interface (see sensitivity analysis for verification).

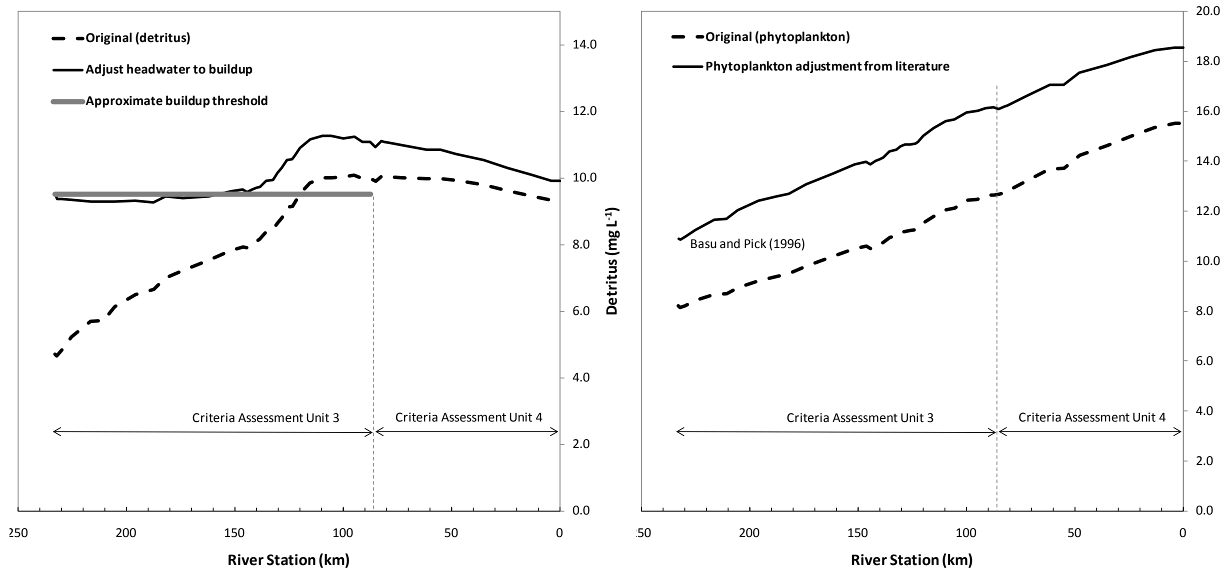

Figure 13-2. Consideration of headwater detritus and phytoplankton concentrations.
(Left panel) Longitudinal plots of detritus with (solid line) and without (dashed line) adjustment for future eutrophied conditions where approximate downstream buildup was used as the guiding factor. (Right panel) Same but for phytoplankton using the literature compilation. Note that detritus (which is productivity based) peaks around km 125 while phytoplankton concentrations continually increase in the downstream direction. Both runs reflect a hypothetical run concentration of 7 µg L^{-1} SRP.

13.2 Results of Nutrient Addition Simulations

To recap, nutrients in the Yellowstone River are currently *below* levels that will cause violations to existing state water quality standards and nutrient *additions* were completed to identify levels that would be limiting. Ten model runs with incremental changes in ambient N and P concentrations were completed to assess the eutrophication response of the river. Results are shown in the **Table 13-4** (for N) and **Table 13-5** (for P).

Output variables evaluated in each model run included total N and P concentration (µg L^{-1}); DO minima (mg L^{-1}); DO delta (mg L^{-1}), i.e., maximum DO minus minimum DO at a station; pH maximum and minimum; pH delta (maximum pH deviation between baseline and scenario condition); mean benthic algae biomass (excluding the four rapids, mg Chl*a* m^{-2}); mean benthic algae biomass in the wadeable region[98] (mg Chl*a* m^{-2}); TOC flux over the criteria unit (mg L^{-1}); and total dissolved gas (TDG) as calculated on an elevation basis, assuming 100% saturation of atmospheric nitrogen and argon gas. The most limiting threshold was used to set the recommendation for the nutrient criteria.

[98] As evaluated through AT2K. The most limiting transect entrance geometry was used in this assessment (i.e., the one that grew the highest mean biomass in the wadeable region). 19 transects were considered in total.

Table 13-4. Model simulations to evaluate the relationship between TN and waterbody response.
Runs carried out under non-limiting P conditions (i.e., 100 µg L^{-1}).

Criteria Unit 3 – Bighorn River to Powder River (as represented by Forsyth to Powder River in model)											
NO_3- (µg L^{-1})	**TN (µg L^{-1})**	**Minimum DO (mg L^{-1})**	**DO delta (mg L^{-1})**	**pH (max)**	**pH (max) delta**	**pH (min)**	**pH (min) delta**	**Benthic algae (mg Chl*a* m^{-2})**	**Benthic algae wadeable zone per AT2K (mg Chl*a* m^{-2})**	**TOC flux (mg L^{-1})**	**TDG (% sat)**
6	370	7.26	1.7	8.66	0.00	8.26	0.00	13.5	16.0	0	105
8	419	7.22	2.3	8.73	0.16	8.29	-0.04	24.9	16.0	1.05	105
10	490	7.23	2.9	8.84	0.27	8.31	-0.10	33.0	65.8	1.65	106
15	591	7.22	3.8	8.94	0.38	8.32	-0.17	43.5	88.8	2.42	108
20	659	7.18	4.3	*9.00	0.43	8.33	-0.20	49.2	103.5	2.82	109
25	745	7.16	4.7	9.05	0.48	8.36	-0.25	53.5	114.7	3.10	110
30	799	7.15	4.8	9.06	0.50	8.37	-0.27	55.0	120.7	3.27	110
50	921	7.10	5.3	9.11	0.54	8.37	-0.30	60.5	139.1	3.74	111
70	1090	7.07	5.6	9.13	0.57	8.38	-0.33	61.9	147.8	4.11	112
100	1241	7.06	5.7	9.15	0.58	8.39	-0.35	62.7	154.2	4.37	113
Criteria Unit 4 – Powder River to state-line (as represented by Powder River to Glendive in model)											
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20	696	7.22	3.0	8.80	0.00	8.60	0.00	23.3	106.7	0.00	106
25	713	7.22	3.2	8.80	0.02	8.53	-0.02	24.7	115.6	0.06	106
30	728	7.21	3.3	8.81	0.03	8.53	-0.03	26.2	121.8	0.09	106
50	780	7.20	3.6	8.84	0.07	8.55	-0.07	30.1	140.8	0.20	107
70	820	7.19	3.8	8.86	0.09	8.57	-0.09	32.4	*150.6	0.25	108
100	871	7.19	3.9	8.88	0.11	8.59	-0.11	34.3	159.9	0.31	108

*Limiting factor

Table 13-5. Model simulations to evaluate the relationship between TP and waterbody response.
Runs carried out under non-limiting N conditions (i.e., 1,000 µg L^{-1}).

SRP (µg L^{-1})	TP (µg L^{-1})	Minimum DO (mg L^{-1})	DO delta (mg L^{-1})	pH (max)	pH (max) delta	pH (min)	pH (min) delta	Benthic algae (mg Chl*a* m^{-2})	Benthic algae wadeable zone per AT2K (mg Chl*a* m^{-2})	TOC flux (mg L^{-1})	TDG (% sat)
Criteria Unit 3 – Bighorn River to Powder River (as represented by Forsyth to Powder River in model)											
2	39	7.30	1.80	8.68	0.00	8.33	0.00	16.7	31.0	0.00	105
3	41	7.30	2.34	8.76	0.16	8.33	-0.07	24.1	43.8	0.69	105
4	45	7.30	3.01	8.86	0.26	8.36	-0.13	32.9	59.9	1.23	106
6	54	7.26	3.87	*8.97	0.37	8.38	-0.21	43.3	82.4	2.01	108
8	62	7.22	4.34	9.02	0.42	8.40	-0.25	49.0	96.8	2.45	109
10	74	7.20	4.64	9.05	0.46	8.41	-0.28	51.6	106.3	2.83	110
15	87	7.16	5.13	9.10	0.50	8.42	-0.31	57.1	123.0	3.29	111
20	110	7.13	5.43	9.13	0.53	8.43	-0.34	58.4	132.2	3.73	112
30	136	7.10	5.74	9.16	0.56	8.43	-0.37	60.4	144.0	4.16	113
50	168	7.08	5.97	9.18	0.58	8.43	-0.38	61.2	154.4	4.60	113
Criteria Unit 4 – Powder River to state-line (as represented by Powder River to Glendive in model)											
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---	---	---	---	---	---	---	---	---	---	---	---
6	70	7.29	2.82	8.80	0.00	8.52	0.00	20.8	85.8	0.00	106
8	73	7.28	3.05	8.81	0.00	8.52	-0.03	24.8	101.3	0.09	106
10	77	7.27	3.25	8.84	0.00	8.52	-0.05	27.7	113.3	0.17	107
15	84	7.26	3.57	8.87	0.01	8.52	-0.08	32.2	131.8	0.27	107
20	90	7.24	3.78	8.89	0.01	8.52	-0.10	35.3	*144.2	0.34	108
30	102	7.22	4.02	8.91	0.02	8.52	-0.12	38.8	158.2	0.41	108
50	124	7.20	4.27	8.93	0.02	8.52	-0.14	42.3	172.5	0.48	109

*Limiting factor

The baseline for each simulation (i.e., against which the additions were subsequently compared) reflect conditions in the first line of **Table 13-4** and **13-5** for Criteria Unit 3 whereas the computed criteria for the upper segment were used as the baseline for Unit 4. We used the following to determine the soluble nutrient levels for the baseline at the upstream boundary condition at Forsyth: (1) historical low-flow water quality data (**Table 12-5**), (2) the literature (Meybeck, 1982), and (3) data on streams/rivers elsewhere in the state. We chose soluble N and P targets of 6 µg NO_3^- L^{-1} and 2 µg SRP L^{-1} as naturally occurring, which align very closely to the lower measured concentrations in the Yellowstone during 2007 (3-6 µg NO_3^- L^{-1} and 2-3 µg SRP L^{-1}). They agree reasonably well with Meybeck (1982) who indicates that unpolluted rivers[99] have concentrations ranging from 20-200 µg L^{-1} NO_3^- and 2- 20 µg L^{-1} SRP (P-PO_4), and are consistent with Biggs (2000a) who reports similar values for rivers/streams in New Zealand (slightly lower for soluble nitrogen). Finally they match well with medians (≈5 µg L^{-1} for each nutrient) for reference streams of the Middle Rockies/Northwestern Great Plains (Strahler order 4 or greater) as determined by DEQ.

Our mechanism for determining the criteria based on the numerical experiments described previously are shown in **Figure 13-3**. In general, we wished to first identify the criteria in the upper river (Unit 3) such that this would inform the boundary condition for the lower river. In each case the criteria would be set slightly below the most limiting ecological response.

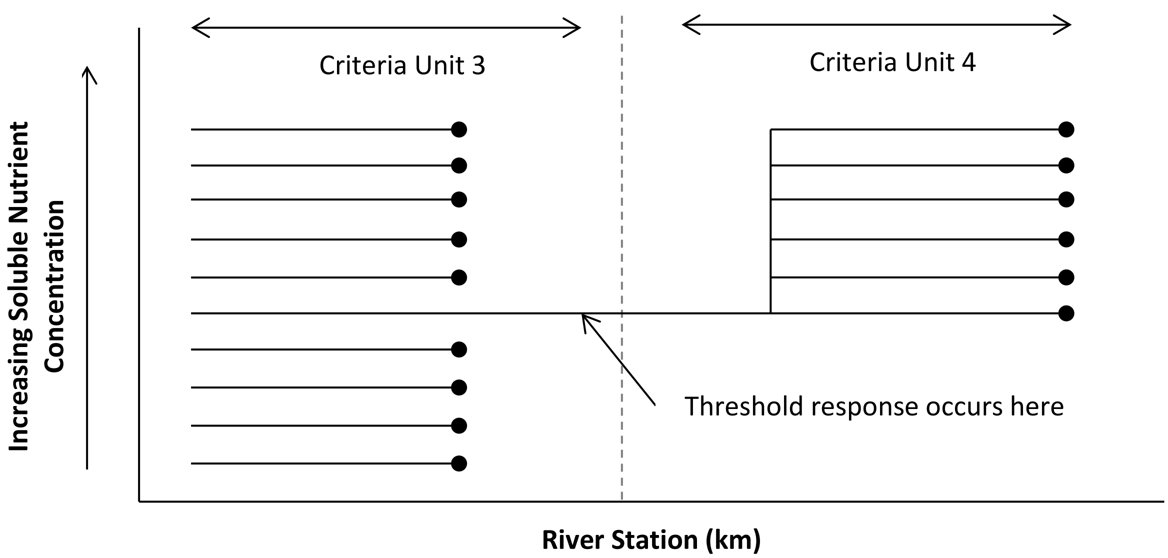

Figure 13-3. Approach toward criteria development.
Soluble nutrients are accreted in the upper river (Criteria Unit 3) until a limiting response is achieved. The threshold then predicates the boundary conditions for Unit 4. Subsequent incremental runs can then be carried out to identify the most limiting response in the lower river.

Upon initial examination of the model output, differences were noted in the behavioral response between the two criteria units. For example in Unit 3, pH was most restrictive to increased nutrient levels, whereas benthic algae caused limitation in Unit 4. The responses tell us pH excursions will likely result at ≈655 µg TN L^{-1} and ≈55 µg TP L^{-1} in the upper river, indicated by a shift to a daily maximum pH

[99] Based on an analysis of the arctic, subartic, and temperate regions of the world.

of 9.0 (see **Section 13.3.1** for further explanation). In the lower river, nutrients should be limited to concentrations of ≈815 µg TP L^{-1} and ≈95 µg TN L^{-1} to prevent nuisance algae (150 mg Chl*a* m^{-2}) in the wadeable zones. And as mentioned earlier, the results in the lower river are a function of the nutrient criteria identified for the upper river which are input at the upper boundary of the lower river. An illustration using actual model runs of thresholds is shown in **Figure 13-4** (Left/right panel). In general, the river was less responsive to nutrients going downstream hence benthic algal biomass was used as the indicator in the lower river.

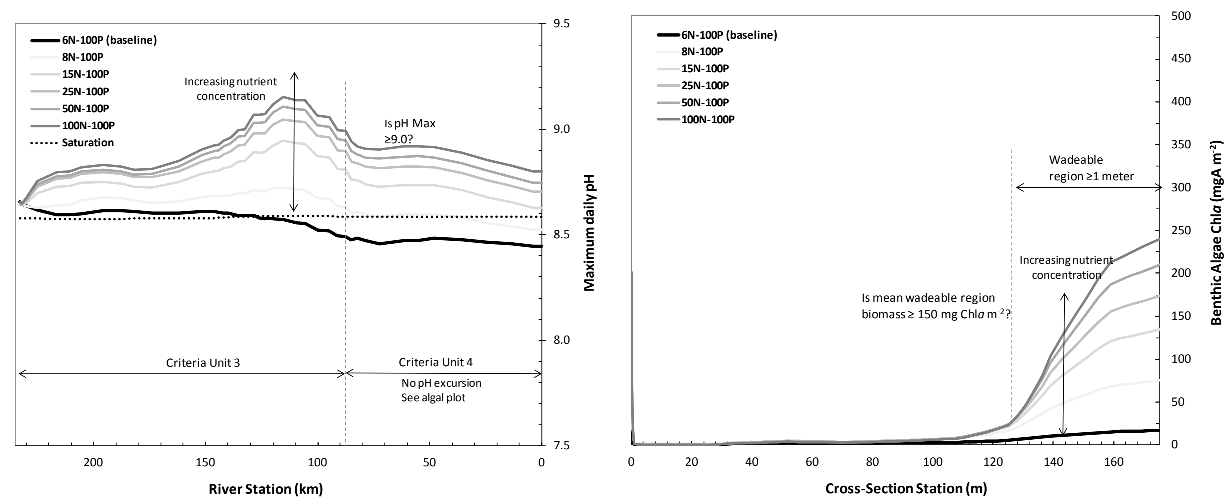

Figure 13-4. Example of a water quality endpoint determination.
(Left panel) Derivation of pH impairment using the differential between baseline and subsequent nutrient addition Q2K runs. (Right panel) Same but for benthic algae, considering the wadeable region in AT2K.

The model outcomes were all non-linear and the response for the most limiting variable in each criteria assessment area is shown in **Figure 13-5**. Each datapoint reflects one of the ten model runs in **Table 13-4** and **Table 13-5** and, overall, there is a good agreement between the simulated nutrient stressor and associated waterbody response. The behavioral curve has three distinct regions; a linear lower leg where changes between nutrients and the response is very sensitive, an inflection point where the sensitivity to nutrients shifts, and then a less sensitive linear portion where large increases in nutrients have only a minor effect. Hence, tipping-point thresholds exist in the river in relation to nutrient levels and associated waterbody response.

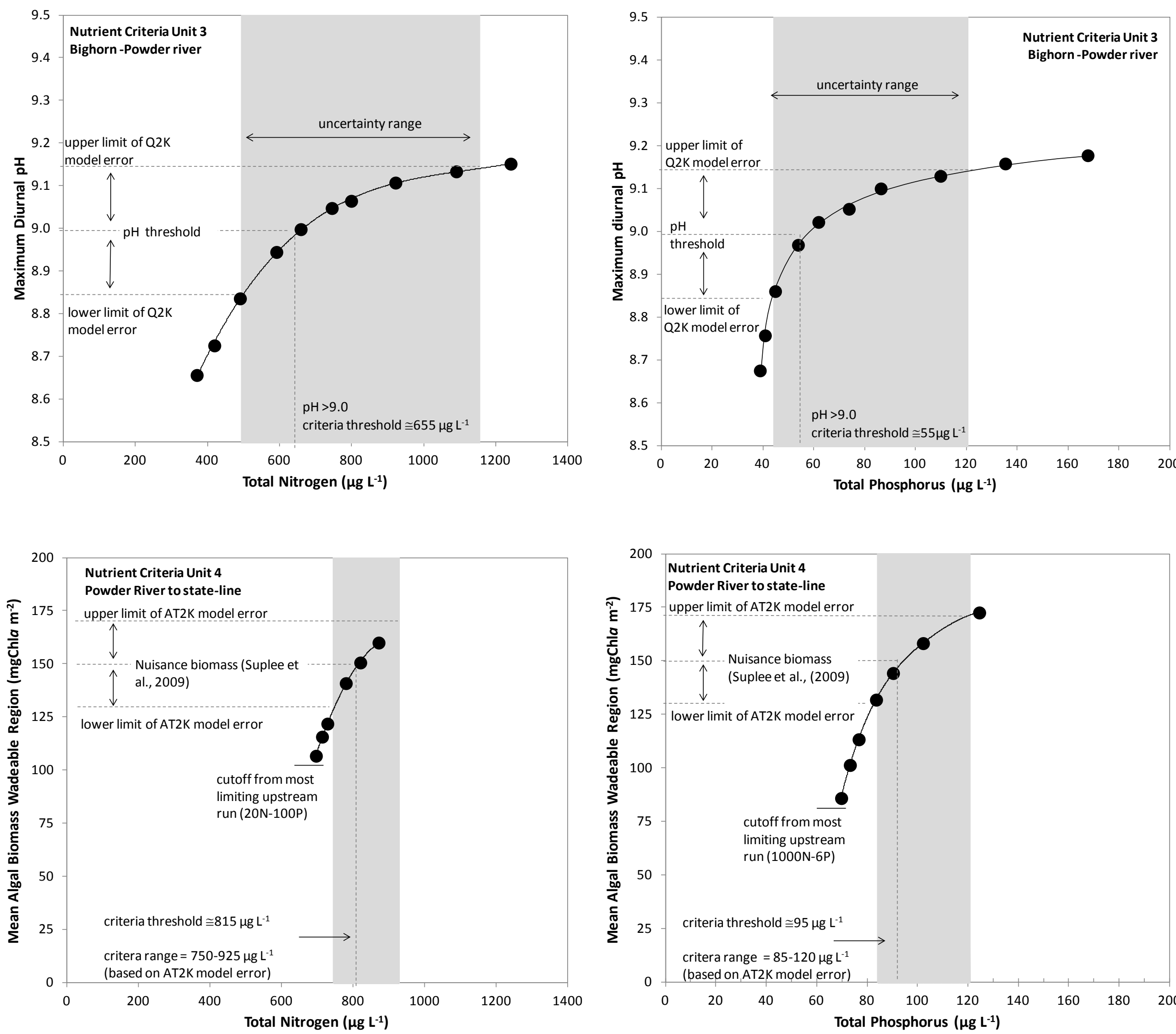

Figure 13-5. Non-linear response relationship between nutrients, pH, and benthic algae.
(Top left/right panel) Most limiting pH response associated with TN and TP concentrations in Criteria Assessment Unit 3 (maximum diurnal pH), which was the basis for establishing the criterion. Criteria are recommended at ≈655 µg TN L^{-1} and ≈55 µg TP L^{-1} in this part of the river. (Bottom left/right panel) Most limiting benthic algae wadeable response in relationship to TN and TP concentration in Criteria Assessment Unit 4. Recreational use biomass thresholds are based on Suplee, et al., (2009) and reflect the mean density in the wadeable zone (see further discussion in **Section 13.3.2**). TN and TP criteria in Unit 4 are therefore recommended at ≈815 µg TN L^{-1} and ≈95 µg TP L^{-1} in this region. Note: The grey shaded area reflects the uncertainty in the model prediction based on model errors determined in **Section 10.0**.

In review of the previous plots, the non-linear response used to interpolate between simulated data provides a reasonable estimate of the criteria. However it is important to note that the fitted line has no apparent physical meaning short of the least-squares fit. Thus it implies better precision than really exists. To acknowledge this fact, thresholds were attained through rounding to the nearest 5 µg L^{-1}. Those determined for the lower Yellowstone River (and noted previously) are shown in **Table 13-6**.

Table 13-6. Recommended numeric nutrient criteria for the Yellowstone River.

Location	Total Nitrogen (µg L^{-1})	Total Phosphorus (µg L^{-1})
Criteria Assessment Unit 3 Big Horn River to Powder River	655	55
Criteria Assessment Unit 4 Powder River to state-line	815	95

13.3 Discussion

Discussions about each of the state-variables considered in the criteria evaluation, including those not used to promulgate criteria, are presented below (i.e., for pH, benthic algae biomass, DO, TOC, total dissolved gas, etc.). They summarize and provide supporting information regarding our conclusions in the previous section.

13.3.1 Nutrient Criteria Based on pH

The segment of the Yellowstone River from Forsyth to the Powder River (Criteria Assessment Unit 3) was found to be most sensitive to induced pH change relative to nutrient additions. Thus some discussion about the Montana pH standard is of merit. The state apparently crafted the pH rule after the national pH standards presented in EPA's blue, red, and gold books (EPA, 1972; EPA, 1976; EPA, 1986a), with emphasis on the blue book (EPA, 1972). A review of EPA (1972) and state law (ARM 17.30.625) shows that the pH standard has two distinct parts. Both induced variation (delta) and the shift to a new pH are important. Water quality standards would be exceeded when the induced change is ≥0.5 units or if the pH is moved outside of the range of 6.5 to 9.0 pH units.

The TN and TP criteria were established at levels just under a pH of 9.0 and, concurrently, at the point where induced change (pH delta) was about 0.4 units (**Tables 13.4, 13.5**). The standard's upper limit of 9.0 reflects current scientific understanding of pH impacts. Recent reviews of the scientific literature show that pH levels moved beyond 9.0 harm fish (Robertson-Bryan, Inc., 2004), corroborating what had previously been established for warm-water fish populations (European Inland Fisheries Advisory Council, 1969). Harm to fish is caused by the greater prevalence of OH^- ions which cause increased basicity and hypertrophy of mucus cells in gill filaments and skin epithelium, and additional detrimental effects on the eye lens and cornea (Alabaster and Lloyd, 1980; Boyd, 1990).

A final consideration is whether human-caused factors may have artificially elevated the boundary condition of the modeling reach (and are reflected in what we are calling naturally occurring). Our understanding is that a pH of 8.5 at Forsyth is natural or close to a natural level. For example, multi-year monitoring studies show a longitudinal change in pH along the Yellowstone River, from just outside of Yellowstone National Park (median: 7.95) to Livingston (median: 8.0) to Billings (median: 8.2) to Forsyth (median: 8.4) (Miller et al., 2004).

Freshwater pH is largely controlled by the carbonate-bicarbonate buffer system (Morel and Hering, 1993) and surface waters in Montana are very often alkaline. Downstream of Billings cretaceous sedimentary rocks underlay the river and contribute to increasing calcium carbonate concentrations that elevate pH (USGS, 2004). In fact, if we use the 25th percentile bicarbonate concentration at Forsyth (90 mg L^{-1}; (Miller et al., 2004) and open carbonate equilibrium theory (i.e., H_2CO_3*=10^{-5} molar and pK_{a1}=6.35), pH should naturally be approximately 8.5 assuming all bicarbonate is geochemically derived (which seems reasonable using the 25th percentile). Finally, the Big Horn River (upstream of the modeled reach) contributes a large proportion of flow to the Yellowstone River and has a median alkalinity of 188 mg L^{-1} as $CaCO_3$ (much higher than the Yellowstone River at Livingston, where median alkalinity is 54 mg L^{-1} as $CaCO_3$). The Bighorn basin is dominated by rangeland land uses which for the most part are natural. Thus while we cannot say with 100% absolute certainty that baseline pH in our modeled reach is natural, the suggested baseline values are fairly typical for larger rivers and streams in the Yellowstone River basin (median range: 8.1 to 8.5) (Lambing and Cleasby, 2006) and reasonable approximations of natural.

13.3.2 Nutrient Criteria Based on Benthic Algae Biomass

Benthic algae increased in all nutrient addition scenarios however only in the lower river did they reach unacceptable algal biomass changes prior to exceedances of other water quality standards. This occurred primarily due to light limitation (i.e., there was not a large enough photosynthetic zone to induce pH changes up to a detrimental level). Hence benthic algae in the wadeable zone of the lower river were identified as the primary driver for establishing nutrient limits in that area.

Levels of benthic algae in excess of 150 mg Chl*a* m^{-2} have been demonstrated to be an unacceptable impediment to river recreation in Montana (Suplee et al., 2009) which proves useful in this work. For example, survey respondents from the Miles City area showed preference for river algae ≤150 mg Chl*a* m^{-2} (Table 6; Suplee et al., 2009), but the sample size was small (n = 13) and not all preference levels were significantly different from 50%. A similar and significant response was received from the Billings area, which means that maintaining river algae below nuisance levels of 150 mg Chl*a* m^{-2} is important to people living and recreating along the lower Yellowstone River. It is also consistent with a number of past literature studies where biomass of 100-200 mg Chl*a* m^{-2} was determined to be nuisance (Horner et al., 1983; Welch et al., 1988) and thus is reasonable for management.

In our case however, direct application of 150 mg Chl*a* m^{-2} to the entire river is not appropriate as light gradients predispose the river to luxuriant algal growth only in shallow regions of the river whereas the remaining sections of river are strongly light limited and are not productive (see **Figure 2-2** in **Section 2.2.1**). It would be very difficult then (if not impossible) to observe a reach-average biomass in excess of 150 mg Chl*a* m^{-2} in the Yellowstone River. Therefore the most relevant nutrient control policy in this case is to limit biomass in the wadeable region of the river where recreation occurs (i.e., wading, fishing, tubing, canoeing, swimming, etc.). An observer must pass through, or directly use the wadeable zone, to gain benefit from the river and thus we require the mean biomass in this region to be ≤150 mg Chl*a* m^{-2}.

The near shore margins of large rivers are also the nursery areas for fish larvae and young-of-year juveniles, which are collectively referred to as the 0+ age class (Copp, 1992; Scheidegger and Bain, 1995). These young river fish have narrow, specific habitat requirements whereas older fish of the same species tolerate wider ecological conditions (Jurajda, 1999). Fish of the 0+ age class are attracted to the river margins by slower velocities, shelter from predators, often warmer temperatures, and the increased availability of food from primary productivity (Pease et al., 2006; Scheidegger and Bain, 1995). Depending on species, 0+ warm-water fish have variable preferences for dense algal growth; some clearly avoid it (Copp, 1997; Copp, 1992). Although it has apparently not been studied in rivers, it is quite conceivable that allowing excessive benthic algae mats to develop in these shallow near-shore margins could impact 0+ fish. Strong detrimental impacts from dense algae mats on commercially-important juvenile fish along shallow (≈1 m depth) marine shorelines have been documented (Pihl et al., 1994; Pihl et al., 1995; Pihl et al., 2005). Presumably a similar impact could occur in large river margins, where too much algae would lead to suboptimal conditions and changes in food resources for many juvenile fish in their critical nursery habitat.

Previous work has defined wadeable as being ≤1 m[100] (Flynn and Suplee, 2010), therefore our only objective was to integrate biomass within the model such that average of this zone was ≤150 mg Chl*a* m^{-2}. Overall, 20 transect locations were evaluated (**Table 13-7**) with the most limiting entrance geometry (i.e., the one that had the largest algal response and that achieved the most nuisance conditions in the wadeable region[101]) being used in formulation of the criteria. We only considered cross-sections where approximately 25% or greater of the overall transect width was wadeable (i.e., a good proportion of the river bed would be affected) and found similarities between limiting geometries at a number of sites.

With this in mind, the manner in which management endpoints are computed strongly affect the criteria. For example, we used the average benthic algal biomass in the wadeable zone (defined as depths of ≤1 m) as our regulatory endpoint. If we managed the river so that no stone were to exceed 150 mg Chl*a* m^{-2}, the criteria would be different and would be lower. However, regulation of nutrients towards a single stone (i.e., the single highest algae replicate observed) would not be consistent with the way the algal biomass threshold was derived. For example, the basis of Suplee et al., (2009) was that participants were shown photos of entire river reaches and were asked their impressions (acceptable/non-acceptable) of the entire scene. Since the impressions would be based on the overall appearance of the algae levels (not a single point) and, correspondingly, the algae biomass values provided were the reach averages (of n=10 to 20 replicates), we must regulate biomass for the average condition of the wadeable region, not the single highest Chl*a* value recorded (i.e., not the single most-green stone). Similarly, it is unlikely that a few stones with very high algae levels could harm 0+ age fish in the near shore nursery area, whereas if the majority of the stones in the near-shore margin were covered with thick mats of algae, an effect on the juvenile fish would be much more likely.

A further consideration with respect to benthic algae biomass is the uncertainty owed to collection and analytical measurement. DEQ has evaluated this in our wadeable streams program and has concluded that a stream whose mean benthic algae level is measured and found to be ≥ 129 mg Chl*a* m^{-2} could plausibly have a true benthic algae level at or above nuisance (Suplee and Sada de Suplee, 2011). Likewise, replicate measurements in the wadeable zone of large rivers indicate that we can be 80% confident that the measured mean algal Chl*a* is within ± 30% of the true mean (DEQ, 2011b). However, data uncertainty has no correlation with model uncertainty (assuming the model is calibrated to uncertainty erring equally in both directions), therefore we chose to use the direct model output for our analysis. We recognize that error in our estimate could go either direction (as shown in **Figure 13-5**), and that this should be considered in regulatory management as the river nears the criteria.

[100] This is the depth that roughly corresponds to the force where a person could still wade and not be over-toppled by the oncoming water force.
[101] The sections surveyed are believed to encompass typical variability in the river. However, only a subset of very long river reaches were evaluated (n=approximately 10 in each reach). Thus the most-limiting cross-section entrance geometry was used towards to make an appropriate determination of nutrient thresholds to restrict algal biomass accumulations at that site to <150 mg Chl*a* m^{-2}. It was determined that mean wadeable depth explained most of the variance in the wadeable and cross-sectional biomass average (r^2=0.93) and only the transect with the shallowest mean wadeable depth required evaluation.

Table 13-7. Locations of benthic biomass evaluation and most limiting transect geometry.

Location	% Wadeable	Mean Wadeable Depth
Criteria Assessment Unit 3		
Meyers Bridge	29.1	0.78
Forsyth Bridge	21.9	0.64
Far West FAS	58.6	0.49
Paragon Bridge	67.3	0.57
Keough Bridge	16.8	0.56
Hwy 59 Bridge	38.8	0.51
Pirogue Island	65.1	0.52
Kinsey Bridge	63.6	0.52
US Powder River	21.0	0.36[1]
Criteria Assessment Unit 4		
US Calypso Bridge	43.4	0.32
Calypso Bridge	12.1	0.43
Terry Bridge	26.8	0.56
US O'Fallon Creek	24.4	0.30[1]
Fallon Bridge	30.0	0.45
Glendive RR Bridge	01.4	0.11
Glendive Bell St. Bridge	12.3	0.34
Glendive I-94 Bridge	09.5	0.53
Sidney Bridge	15.6	0.48
Fairview Bridge	09.3	0.42

[1]Most limiting cross-section to biomass response based on mean wadeable depth

Finally, it should also be noted that a lower benthic algae standard for the Clark Fork River (100 mg Chl*a* m^{-2} as a summer average) was recommended along with a 150 mg Chl*a* m^{-2} maximum in the 1990s as part of the Voluntary Nutrient Reduction Program (VNRP). However, estimates at that time were based on limited academic literature, which did not include evaluation of the public's opinion on the matter. Subsequently, Suplee et al., (2009) show that the public majority in the Clark Fork basin (i.e., Missoula) are accepting of average algae levels up to 150 mg Chl*a* m^{-2} (but no higher). Thus, we believe that the 150 mg Chl*a* m^{-2} benchmark is, on average, appropriate. In regard to aquatic life uses, nutrient criteria are determined according to the most sensitive use. So if aquatic life standards were exceeded according to the model (e.g., pH or DO) they were used in establishing the criteria. We have no evidence that 150 mg Chl*a* m^{-2} impairs aquatic life uses in large rivers (although the possibility may exist for young fish), whereas it does in wadeable streams due to accrual of senesced, decomposing algae in runs and pools (resulting in seasonal DO minima <5 mg L^{-1}).

13.3.3 Dissolved Oxygen Effects

From analysis of DO in the model, it is highly unlikely that DO minima in the river will ever reach the 5 mg L^{-1} threshold. A number of physical factors support this conclusion including: (1) the presence of the Cartersville Diversion Dam which is a reaeration source, (2) high reaeration rates of the river itself, (3) low river SODs, and (4) the fact that much of the bed of the river lies deep below the surface and is unsuitable for aquatic plant growth. Historical data tends to support this assertion as even during times of heavy historical pollution, there has never been a documented DO minima downstream of Forsyth[102] (Knudson and Swanson, 1976; Montana Board of Health, 1956; Peterson et al., 2001). Consequently, we can only foresee DO being a potential problem in two circumstances: (1) if a very large BOD source were to be permitted in the future (which, based on state and federal laws, will not occur) or (2) if excess benthic algal accumulation during the summer months influences the river's DO in the fall when algae die *en masse* during senescence. This was observed by DEQ during a whole-stream fertilization study in eastern Montana recently[103].

The concern about DO demand from algal decomposition is valid[104] as CBOD consumes oxygen when oxidized. To evaluate this consideration, we made some very conservative assumptions regarding the immediate oxidation of algal organic material using a BOD source in the model equal to 150 mg Chl*a* m^{-2} of nuisance algae over the entire cross-sectional biomass (or 16.1 gC m^{-2}). The expected DO demand would be 43.0 gO_2 m^{-2} day^{-1} for river depths of 1 m, which was input to Q2K as a prescribed SOD[105]. Simulations suggest that even a source of this magnitude has negligible impact on DO. In fact, DO minima barely fell below 7 mgO_2 L^{-1}. Thus concerns regarding DO are not valid for the Yellowstone River and no further consideration of DO was completed.

13.3.4 Total Organic Carbon and Disinfection Byproducts

TOC levels were also evaluated to ensure that treatment level thresholds for carcinogenic disinfection byproducts (DBPs) would not be exceeded. Through our model simulations, however, it became apparent TOC levels in the river were perhaps too vague of an endpoint to develop nutrient criteria against. First, we had difficulty identifying treatment and/or filtration costs associated with their removal. Likewise we did not find a suitable way to project those estimates into the future for some hypothetical design capacity of source water treatment plants. Despite these problems, it appears as if TOC is not an important model endpoint anyway. In all model trials, TOC flux was never much over 5.0 mg L^{-1} for the entire river and only marginally induced a change outside of the categorical treatment level of <4-8 mg L^{-1} TOC suggested by EPA as a breakpoint for increased percentage of TOC removal.

13.3.5 Total Dissolved Gas (TDG)

State law requires that induced TDG remain below 110% of saturation (**Table 4-3**) to protect fish from gas bubble disease. However, the standard is intended more to control supersaturation of atmospheric gas below dam spillways. In the Yellowstone River, gas supersaturation is driven predominantly by diel DO changes. A thorough literature review on gas supersaturation effects on fish (Weitkamp and Katz,

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[102] Billings (upstream) is the exception as several exceedances have occurred there (Montana Board of Health, 1956; Peterson et al., 2001).

[103] The whole-stream nutrient-addition study was completed on a wadeable 5th order stream in Eastern Montana (DEQ, 2010). Based on our observations, dissolved oxygen impacts from excess nutrients were out of phase with the period of peak algal production. In fact they occurred entirely after the growing season as algae senesced *en masse*. The decaying material settled in the low-velocity regions of the stream resulting in localized areas of high CBOD which effectively acted like an intense sediment oxygen demand (Appendix B; Suplee and Sada de Suplee, 2011).

[104] In several instances we saw large mats of dead and dying benthic algae washed onto the banks of the river, however, there was no apparent influence on DO based on our sonde data.

[105] The SOD 43.0 gO_2 m^{-2} day^{-1} was about two times higher than the highest river SOD we were able to locate in the literature [e.g., 21.4 gO_2 m^{-2} day^{-1} from Ling et al., (2009)].

1980) shows that fish may be tolerant of higher total gas levels than what is reflected in the state's standard (e.g., 110% saturation) provided that the gas pressure is being driven by biogenic oxygen. For example, fish are shown to develop gas bubble disease only when DO saturation levels reach 300%. When the supersaturation effect is intermittent, as it is in the Yellowstone River, the negative impact on fish is greatly reduced. DO supersaturation levels observed in our highest nutrient-addition model runs peaked at 163% of saturation, which equates to 113% saturation for TDG[106]. At the nutrient concentrations we are recommending to keep pH below 9.0 in the upper river, DO saturation would be no higher than 144% of saturation, which equates to 109% TDG (same assumptions as before) and is below the state's water quality standard. In the lower river TDG levels would be even lower (**Tables 13-4, 13-5**). Given that the supersaturation effect is caused by DO which will remain far below 300% saturation, is intermittent, and is below the state's TDG criterion, we contend that the nutrient criteria recommendations will be protective.

13.4 Summary

A number of plausible nutrient criteria endpoints were evaluated through modeling analysis. It was identified that pH was the most sensitive nutrient-influenced water quality endpoint in Criteria Unit 3 (upstream; model boundary to the Powder River) whereas it was benthic algae in Criteria Unit 4 (downstream; Powder River to state line). The difference between the two regions was primarily light availability (i.e., suspended fines mute primary productivity in the lower river) which necessitates different criteria. Recommended nutrient criteria then are ≈655 µg TN L^{-1} and ≈55 µg TP L^{-1} in Unit 3 and ≈815 µg TN L^{-1} and ≈95 µg TP L^{-1} in Unit 4. Model results also showed a non-linear response to increases in nutrients. There is generally an initial phase where water quality parameters change quickly with nutrient addition, followed by an inflection point, and then a less-responsive phase where elevated concentrations affect the water quality parameters only slightly. We have used the rate of uptake/recycle and associated transport in the model to determine how total nutrients at one point relate to conditions at another (note: these points are different longitudinally because of advection). The nutrient addition simulation runs presented in this section represent the endpoint of our modeling work on the lower Yellowstone River. Uncertainty regarding these simulations is detailed in the next section and then we conclude with final recommendations for the sections of the river evaluated.

[106] Assuming an elevation-based barometric pressure near Miles City of 690 mm Hg and assuming 100% saturation of atmospheric nitrogen + argon gas.

14.0 Uncertainty Analysis

Following the identification of approximate nutrient criteria thresholds for the Yellowstone, an uncertainty analysis was completed to better understand the implications of these findings and prescribe a defensible margin of safety for the final criteria recommendation. The details of the uncertainty analysis follow.

14.1 Error-propagation Methods

Information on uncertainty is necessary to understand the relative confidence in model results. Uncertainty is inherent in natural systems and includes variability brought about by spatial and temporal water quality and underlying processes. Analysis of uncertainty is therefore an important part of both water quality management (Beck, 1987; Brown and Barnwell, 1987; Reckhow, 2003; Vandenberghe et al., 2007; Whitehead and Young, 1979) and ecological modeling (Chapra, 2003; Reckhow, 1994; Reckhow, 2003).

Uncertainty in water quality models can generally can be lumped into three categories: (1) uncertainty about the relationships in the model or model structure, (2) uncertainty about the value of model parameters or rate coefficients, or (3) uncertainty associated with prediction of the future behavior of the system (Beck, 1987). In this application, we are primarily interested in the latter two components, and how they interrelate to inform the overall error in model simulations.

To characterize uncertainty, error-propagation techniques were used to quantitatively express reliability (Whitehead and Young, 1979). Steps towards completing such an analysis include (Vandenberghe et al., 2007):

1. Identifying sources that contribute to the overall uncertainty of the modeling. We have already done this through our sensitivity analysis in **Section 8.1**.
2. Estimating the uncertainty related to those contributors and underlying assumptions (i.e., distribution, etc. as outlined in this section).
3. Propagating the uncertainty through the model (described later).
4. Analyzing results.

The simplest technique for making such an analysis is Monte Carlo simulation. The application of such principles is described in the next section.

14.2 Monte Carlo Simulation

Monte Carlo simulation (MCS) is a computational algorithm that relies on repeated random sampling to compute a statistical result about a model output. It is a way to numerically address uncertainty so that that combined effects of parameter sensitivity and uncertainty are considered (Melching and Yoon, 1996). Input variables are sampled at random from their pre-determined probability distributions (or cumulative density functions; CDF) and the distribution of the output variable is reviewed to yield a new probability distribution of model outcomes. While a large number of model runs is required (i.e., repeated simulations) to make such determinations, the changes in uncertain model parameters using random selections from their assumed or estimated probability distributions reflects the cumulative uncertainty of the model (Whitehead and Young, 1979). A number of possible probability density functions (PDF) for model input can be specified (e.g. normal, log-normal, triangular, uniform, etc.), and numerous model runs are executed to determine the PDF of the output variable(s). In our case, the distribution function for the eutrophication responses variables of interest (pH and benthic algae) and proposed nutrient criteria (TN and TP) will be considered. The sampling process is shown graphically in **Figure 14-1**, with several hypothetical model parameter distributions (normal, uniform, and triangular) for illustrative purposes.

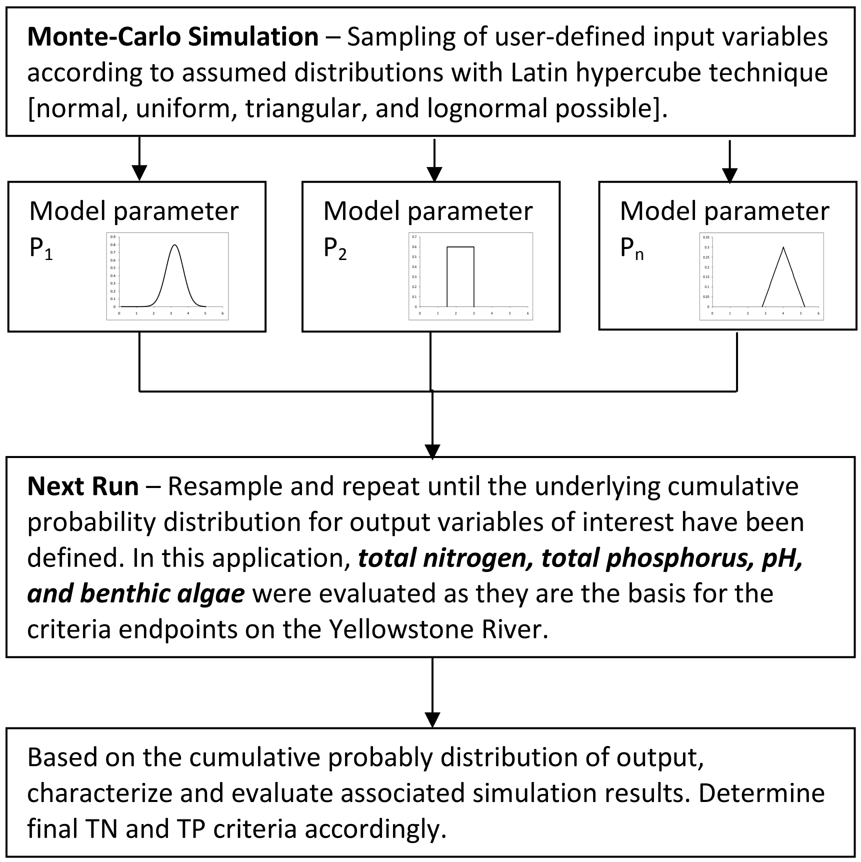

Figure 14-1. Example of Monte Carlo simulation procedure for Yellowstone River.

14.3 ESTIMATES OF UNCERTAINTY IN THE YELLOWSTONE RIVER Q2K MODEL

Four types of uncertainty were considered as part of our MCS:

- Uncertainty in the headwater boundary condition
- Uncertainty in point source pollutants (tributaries/WWTPs)
- Uncertainty in diffuse source pollutants
- Uncertainty in model parameters/rate coefficients

The analysis was carried out using a version of Q2K called QUAL2K-UNCAS. This was developed as part of a cooperative effort between Tufts University and DEQ (Tao, 2008). It is an update of the original QUAL2E-UNCAS (Brown and Barnwell, 1987). Improvements made to the model included the incorporation of Latin Hypercube sampling strategies instead of that of random sampling (McKay et al.,

1979) and added CDF selections for the perturbated rates or input variables (including uniform and triangular).

As identified by others (Beck, 1987; Brown and Barnwell, 1987; Reckhow, 1994; Vemula et al., 2004), the most important consideration in MCS is characterizing the model input uncertainty. This includes characterization of each key input PDF shapes (i.e., normal, lognormal, uniform, triangular, etc.) and associated coefficient of variation or relative standard deviation (COV). Unfortunately, such information is not widely available (Brown and Barnwell, 1987). As a result, we did our best to estimate distributions and COVs from historical field and water quality data. We then used the literature and engineering/scientific judgment in the absence of such data.

Uncertainty estimates for the Yellowstone River are shown in **Tables 14-1, 14-2, 14-3, 14-4**. They generally fall within the range identified by others (Brown and Barnwell, 1987; Manache et al., 2000; Melching and Yoon, 1996; Vandenberghe et al., 2007; Vemula et al., 2004).

Table 14-1. PDF assignments for headwater boundary conditions of the Yellowstone River.
Distributions determined primarily from low-flow data compilation described in **Section 12.0**.

Parameter	Units	Min	Avg	Max	Distribution	Coefficient of Variation (COV) (%)	Literature range[2]
Flow	$m^3 s^{-1}$	93.65	98.58	103.5	n/a	5[1]	5
Temperature	°C	18.0	21.8	26.0	normal	10	1-8
Conductivity	$\mu S\ cm^{-1}$	590	714	945	normal	15	1-15
Inorganic solids[3]	$mg\ D\ L^{-1}$	15	25	47	normal	50	n/a
Dissolved oxygen	$mg\ O_2\ L^{-1}$	7.5	8.9	9.6	normal	10	2-15
CBODfast	$mg\ O_2\ L^{-1}$	8.8	9.4	9.9	normal	15	5-40
Organic-N	$\mu g\ L^{-1}$	not evaluated since already perturbated as part of nutrient addition scenarios detailed in **Section 13.0**.					
Ammonia-N	$\mu g\ L^{-1}$						
Nitrate-N	$\mu g\ L^{-1}$						
Organic-P	$\mu g\ L^{-1}$						
Dissolved-P	$\mu g\ L^{-1}$						
Phytoplankton	$\mu g\ L^{-1}$						
Internal-N	$mgN\ mgA^{-1}$						
Internal-P	$\mu g\ L^{-1}$						
Detritus[3]	$mg\ D\ L^{-1}$	2.8	4.7	8.9	normal	50	n/a
Alkalinity	$mgCaCO_3\ L^{-1}$	129	152	170	normal	10	n/a
pH	pH units	8.3	8.6	8.9	normal	5	n/a

[1] To be consistent with 14Q5, included approximate variation between 14Q5 between gages.
[2] From the following: (Brown and Barnwell, 1987; Manache et al., 2000; Melching and Yoon, 1996; Vandenberghe et al., 2007; Vemula et al., 2004).
[3] ISS based on TSS * 0.8; detritus based on TSS * 0.2 (same distribution and COV assumed)
n/a = not available

As shown previously, conditions at the headwater boundary are normally distributed during low-flow conditions[107]. COVs were low (≤15%, with the exception of ISS and detritus), and most values were within the range reported in the literature. This reaffirms that water quality is not greatly variable during

[107] Most likely the underlying distribution is normal. The Central Limit Theorem would suggest so although it is not entirely valid with a sample size of *n*=10.

that time of the year. As indicated in the tables, N or P loading variance was not included in the uncertainty because each model run was contingent on a specific nutrient concentration in the river. Any change in this load would alter the subsequent concentration and associated outcome.

The point load variance which includes tributary inflow, wastewater effluent, and irrigation main canal return flows (**Table 14-2**) was slightly higher but still fall within the ranges identified in the literature. This was expected given that tributaries are generally flashy and have higher natural variance.

Table 14-2. PDF assignments for point loads on the Yellowstone River.
Distribution determined primarily from database described in **Section 6.0** (August data only).

Parameter	Units	Min[1]	Avg[2]	Max[1]	Distribution	Coefficient of Variation (COV) (%)	Literature range[3]
Flow	$m^3\ s^{-1}$	0	---	7.439	n/a	0[4]	n/a
Temperature	°C	10	---	29	normal	10	1-8
Conductivity	$\mu S\ cm^{-1}$	428	---	3,920	lognormal	5	1-15
Inorganic solids[5]	$mgD\ L^{-1}$	1.0	---	20,610	lognormal	35	n/a
Dissolved oxygen	$mgO_2\ L^{-1}$	3.9	---	17.1	normal	25	2-15
CBODfast	$mgO_2\ L^{-1}$	2.8	---	21.6	lognormal	60	5-40
Organic-N	$\mu g\ L^{-1}$	not evaluated since already perturbated and considered as part of nutrient addition scenarios detailed in **Section 13.0**.					
Ammonia-N	$\mu g\ L^{-1}$						
Nitrate-N	$\mu g\ L^{-1}$						
Organic-P	$\mu g\ L^{-1}$						
Dissolved-P	$\mu g\ L^{-1}$						
Phytoplankton	$\mu g\ L^{-1}$						
Internal-N	$mgN\ mgA^{-1}$						
Internal-P	$\mu g\ L^{-1}$						
Detritus[6]	$mgD\ L^{-1}$	0.1	---	2,160	lognormal	35	n/a
Alkalinity	$mgCaCO_3\ L^{-1}$	142		461	normal	35	n/a
pH	pH units	6.8	---	8.7	normal	5	n/a

[1] Minimum and maximum taken from lumped pool of point load data.
[2] Mean not shown as is dependent on individual point load.
[3] From the following: (Brown and Barnwell, 1987; Manache et al., 2000; Melching and Yoon, 1996; Vandenberghe et al., 2007; Vemula et al., 2004).
[4] Flow was not altered to maintain a 14Q10 streamflow condition in the river.
[5] ISS based on TSS * 0.9 (same distribution and COV assumed)
[6] Detritus based on TSS * 0.1 (same distribution and COV assumed)

Of the point loads, tributary flow had the largest COV (i.e., 135%). We were unable to include this in the analysis however given our requirement to maintain 14Q5 conditions within the river. An example PDF of point load uncertainties are shown in **Figure 14-2 (Left/Right panel)** for water temperature and total suspended solids (as a surrogate for ISS and detritus). In each instance, the normal and natural logarithm (lognormal) distribution best fit the observed data[108].

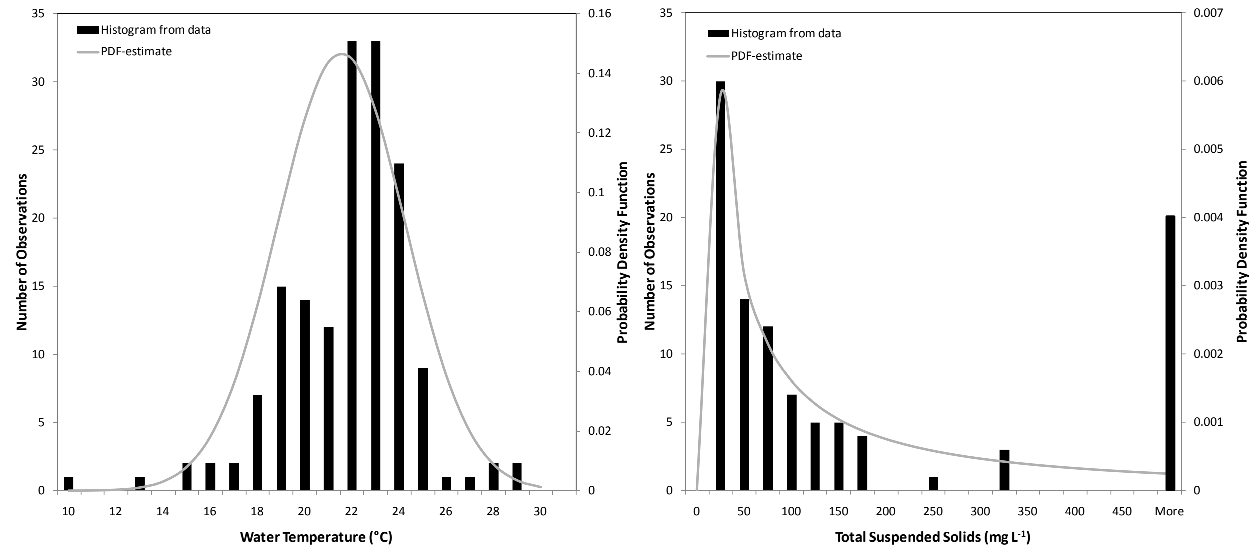

Figure 14-2. Example point load PDF used in Yellowstone River Monte Carlo Analysis.
(Left panel) Normal PDF for water temperature point load to the Yellowstone River. (Right panel) Same but lognormal distribution (natural logarithm) for total suspended solids.

Diffuse boundary conditions (which include groundwater contributions and diffuse irrigation return flows) were difficult to estimate. For example, groundwater inputs are relatively constant whereas irrigation return flows and the diffuse tributary inflows are highly variable. We used the database constructed in **Section 6.0** to estimate the lumped relative uncertainty of these data[109]. Overall diffuse source COVs for the Yellowstone River were slightly higher than point load estimates (**Table 14-3**) which reflect their greater uncertainty. There was no apparent change in the distribution type, only an increase in the variance.

[108] We made only visual comparisons of the histogram to determine the proposed underlying input distribution and then fit those data with the appropriate pdf. The normal distribution was defined as $\frac{1}{\sqrt{2\pi\sigma^2}} e^{-\frac{(x-\upsilon)^2}{2\sigma^2}}$ and the lognormal distribution was $\frac{1}{x\sqrt{2\pi\sigma^2}} e^{-\frac{(\ln x-\upsilon)^2}{2\sigma^2}}$, where μ=mean and σ^2=variance of the variable x's normal or natural logarithm respectively.

[109] QUAL2K-UNCAS software allows only a single average variance for all of the diffuse inflows into the model meaning all diffuse sources are treated as having the same uncertainty (even though the sources may be distinctly different). Thus a lumped analysis is necessary.

Table 14-3. PDF assignments for diffuse loads on the Yellowstone River.
Distribution determined primarily from database described in **Section 6.0** (August data only).

Parameter	Units	Min[1]	Avg[2]	Max[1]	Distribution	Coefficient of Variation (COV) (%)	Literature range[3]
Flow	$m^3\ s^{-1}$		---		n/a	0[4]	n/a
Temperature	°C	9.1	---	34	normal	20	1-8
Conductivity	µS cm^{-1}	493	---	6,970	lognormal	5	1-15
Inorganic solids[5]	mgD L^{-1}	0.3	---	35,382	lognormal	75	n/a
Dissolved oxygen	$mgO_2\ L^{-1}$	3.5	---	12.7	normal	20	2-15
CBODfast[6]	$mgO_2\ L^{-1}$	n/a	---	n/a	lognormal	30	5-40
Organic-N	µg L^{-1}	not evaluated since already perturbated and considered as part of nutrient addition scenarios detailed in **Section 13.0**.					
Ammonia-N	µg L^{-1}						
Nitrate-N	µg L^{-1}						
Organic-P	µg L^{-1}						
Dissolved-P	µg L^{-1}						
Phytoplankton	µg L^{-1}						
Internal-N	mgN mgA^{-1}						
Internal-P	µg L^{-1}						
Detritus[7]	mgD L^{-1}	0	---	3,931	lognormal	75	n/a
Alkalinity	$mgCaCO_3\ L^{-1}$	122	---	1,818	normal	40	n/a
pH	pH units	4.4	---	9.1	normal	10	n/a

[1] Minimum and maximum taken from lumped pool of point load data evaluated.
[2] Mean not shown as is dependent on individual point load.
[3] From the following: (Brown and Barnwell, 1987; Manache et al., 2000; Melching and Yoon, 1996; Vandenberghe et al., 2007; Vemula et al., 2004).
[4] Flow was not altered to maintain a 14Q5 streamflow condition in the river.
[5] ISS based on TSS * 0.9 (same distribution and COV assumed)
[6] Limited BOD data (*n*=5), assume same distribution as point loads, use calculated
[7] Detritus based on TSS * 0.1 (same distribution and COV assumed)

Finally, model rate coefficients were considered. This type of uncertainty is well-detailed in the literature (Brown and Barnwell, 1987; Dilks et al., 1992; Manache et al., 2000; Melching and Yoon, 1996; Reckhow, 2003; Stow et al., 2007; Vandenberghe et al., 2007; Vemula et al., 2004) albeit most estimates originate from a single source (Brown and Barnwell, 1987)[110]. We considered normal distributions (de Azevedo et al., 2000; Melching and Yoon, 1996; Vemula et al., 2004), triangular shapes (Chapra, 1997), and lognormal distributions (Manache et al., 2000) and refined them in the spirit of Hornberger and Spear (1980) as reviewed by Dilks et al., (1992), Reckhow and Chapra (1999), and Stow, et al., (2007) using an informal Bayesian inference approach. The literature was first used to provide an estimate of plausible ranges for a given parameter of interest and then *posteriori* model evaluations were used to narrow that range of acceptable or unacceptable ranges based on site-specific observations. Depending on whether the model gave a behavior generating response [i.e., one that follows the observed data structure, see (1992) and Stow et al., (2007)] or a non-behavior generating response (where the response was beyond acceptable limits), the allowable range was narrowed. Acceptable/non-acceptable responses were based on user best-professional judgment.

[110] Both laboratory and field calibration studies are available to make generalized estimates of uncertainty, however, these have a wide range of outcomes and inconsistent results.

Analysis was completed manually by moving inward from the initial literature array until the parameter value shifted from the non-behavior generating region to that of a behavior generating response[111]. An example of is shown in **Figure 14-3** and was conducted through visual evaluation of all of the model's state-variables. Our recommended distributions, ranges, and the associated COVs for the Yellowstone River MCS are shown in **Table 14-4**.

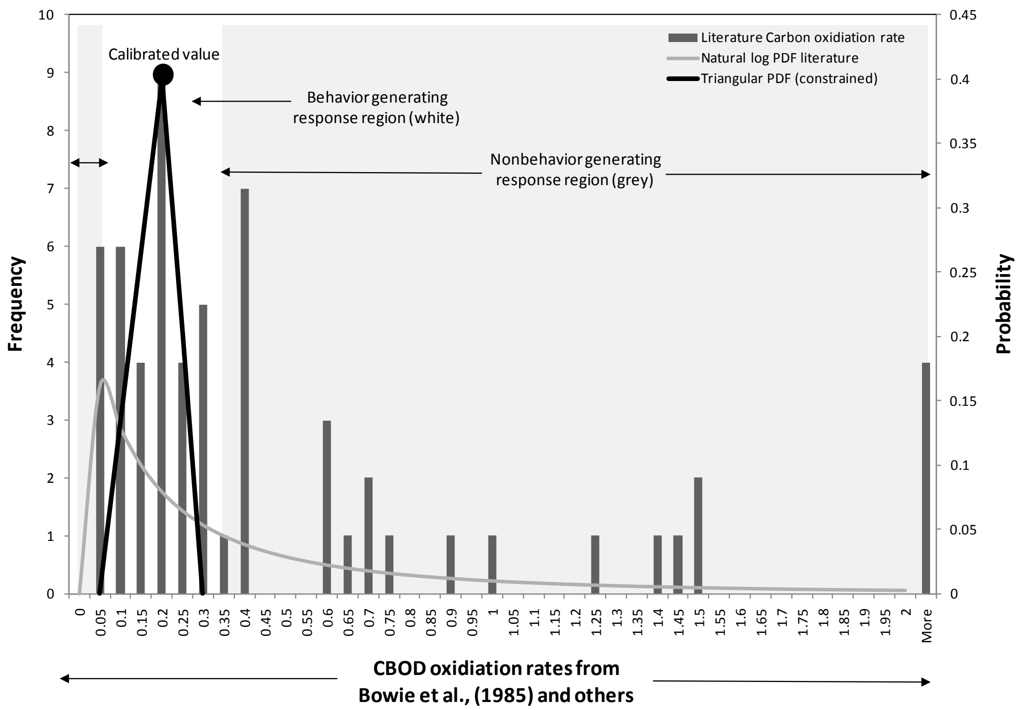

Figure 14-3. Example PDF assignment for model rate coefficients on the Yellowstone River.
The $CBOD_{fast}$ oxidation rate illustrates how the parameter space (and distribution) was first defined using the literature and was then refined using Bayesian principles/*posteriori* analysis of model runs to identify behavior generating response and non-behavior generating response regions for the MCS. A triangular PDF was then fit to these ranges. In the case of CBOD oxidation rate, the PDF approximated a normal distribution. Others are more lognormally distributed.

[111] We did not consider parameter covariances in this determination which means independence between model parameters was assumed (which we know is not totally true, but necessary).

Table 14-4. PDF assignments for rate coefficients for the Yellowstone River.
Distributions determined as noted in table footnote.

Parameter	Units	Min/Max from Bayesian Inference[1]			Distribution	Coefficient of Variation (COV) (%)
		Min	Avg/Mode	Max		
Stoichiometry:[2]						
Carbon	STOCARB	35	43	53	normal	10
Nitrogen	STO NTR	3.7	4.7	4.9	normal	10
Phosphorus	STOPHOS	1.0	1.0	1.0	n/a	n/a
Dry weight	STODRYW	87	107	134	normal	60
Chlorophyll	STOCHLOR	0.4	0.4	1.0	normal	35
$CBOD_{fast}$ oxidation rate	FBODDECA	0.05	0.2	0.3	triangular	n/a
Nitrogen Rates: [3]						
OrgN hydrolysis rate	NH2 DECA	0.05	0.1	0.3	triangular	
Org N settling velocity	NH2 SETT	0.01	0.05	0.1	uniform	n/a
Nitrification rate	NH3 DECA	0.1	2.5	10	triangular	n/a
Denitrification rate	NO3 DENI	0	0.1	2.0	triangular	n/a
Phosphorus Rates: [3]						
OrgP hydrolysis rate	PORG HYD	0.05	0.1	0.3	triangular	n/a
OrgP settling velocity	PORG SET	0.01	0.05	0.1	uniform	n/a
SRP settling velocity	DISP SET	0	0.012	0.1	uniform	n/a
Phytoplankton Rates:[3,4]						
Max growth rate	PHYT GRO	1.7	2.3	2.5	normal	15
Respiration rate	PHYT RES	0.01	0.2	0.5	normal	50
Excretion rate	PHYT EXA	n/a	n/a	n/a	not used	n/a
Death rate	PHYT DET	0.01	0.15	0.25	triangular	n/a
External N half sat constant	PHYNFACT	5	40	200	triangular	n/a
External P half sat constant	PHYPFACT	5	12	100	triangular	n/a
Light constant	PHYLFACT	30	60	90	triangular	n/a
Ammonia preference	PHYPFNH3	5	20	30	uniform	n/a
Subsistence quota for N[5,6]	PHYTQTAN	1.7	2.5	5.9	normal	15
Subsistence quota for P[5,6]	PHYTQTAP	0.06	0.1	0.19	normal	15
Maximum uptake rate for N	PHYT MAXN	10	40	75	triangular	n/a
Maximum uptake rate for P	PHYT MAXP	15	27	50	triangular	n/a
Internal N half sat constant[5,6]	PHYFACTN	1.7	2.5	5.9	normal	15
Internal P half sat constant[5,6]	PHYFACTP	0.03	0.05	0.10	normal	15
Settling velocity	PHYT SETT	0	0.05	1	triangular	n/a
Bottom Algae Rates:[23]						
Max growth rate	BALG GRO	300	400	500	normal	20
Respiration rate	BALG RES	0.01	0.2	0.5	normal	20
Excretion rate	BALG EXA	n/a	n/a	n/a	not used	n/a
Death rate	BALG DET	0.2	0.3	0.4	triangular	n/a
External N half sat constant	BALNFACT	10	250	750	triangular	n/a
External P half sat constant	BALPFACT	30	125	200	triangular	n/a
Light constant	BALLFACT	30	60	90	triangular	n/a
Ammonia preference	BALPFNH3	5	20	30	uniform	n/a
Subsistence quota for N[5,6]	BALGQTAN	1.7	3.2	5.9	normal	15
Subsistence quota for P[5,6]	BALGQTAP	0.06	0.13	0.19	normal	15
Maximum uptake rate for N	BALG MAXN	10	35	150	triangular	n/a

Table 14-4. PDF assignments for rate coefficients for the Yellowstone River.
Distributions determined as noted in table footnote.

Parameter	Units	Min/Max from Bayesian Inference[1]			Distribution	Coefficient of Variation (COV) (%)
		Min	Avg/Mode	Max		
Maximum uptake rate for P	BALG MAXP	3	4	15	triangular	n/a
Internal N half sat constant[5,6]	BALFACTN	1.7	3.2	5.9	normal	15
Internal P half sat constant[5,6]	BALFACTP	0.02	0.04	0.06	normal	15
Detritus:[3]						
Dissolution rate	POM DISL	0.05	0.25	0.5	triangular	n/a
Settling velocity	POM SETT	0	0.05	1	triangular	n/a

[1]The literature range was identified from review of Bowie et al., (1985) as well as others (Auer and Canale, 1982; Biggs, 1990; Borchardt, 1996; Bothwell, 1985; Bothwell, 1988; Bothwell and Stockner, 1980; Chapra, 1997; Chaudhury et al., 1998; Cushing et al., 1993; Di Toro, 1980; Drolc and Koncan, 1999; Fang et al., 2008; Hill, 1996; Horner et al., 1983; Kannel et al., 2006; Klarich, 1977; Knudson and Swanson, 1976; Lohman and Priscu, 1992; Ning et al., 2000; Park and Lee, 2002; Peterson et al., 2001; Rutherford et al., 2000; Shuter, 1978; Stevenson, 1990; Tomlinson et al., 2010; Turner et al., 2009; Van Orden and Uchrin, 1993; Watson et al., 1990).
[2] Determined from multiple seston measurements in summer of 2007 (*n*=15)
[3] Minimum and maximum taken from the literature range initially and then refined through Bayesian inference [see (Dilks et al., 1992; Reckhow and Chapra, 1999; Stow et al., 2007)].
[4] From light-dark bottle experiments in August 2007 (*n*=4); min/max not temperature corrected.
[5] COV for subsistence quota calculated from Shuter (1978).
[6] According to Di Toro (1980) these values have very strong covariance with one another and were evaluated as such.

14.4 Uncertainty Propagation

We used the nutrient addition scenario in **Section 13.2** corresponding to (1) induced pH greater than 9.0 and (2) benthic algal biomass in the wadeable region ≥150 mg Chl*a* m^{-2} which resulted in TN and TP criteria of 655 and 55 µg L^{-1} in the upper river and 815 and 95 µg L^{-1} in the lower river for uncertainty propagation. From review of the literature, it appears as if 1,000-2,000 model runs are sufficient to established acceptable model output distributions (Brown and Barnwell, 1987; Jehng-Jung and Bau, 1995; Melching and Yoon, 1996; Vemula et al., 2004), and consequently, we used a simulation of 2,000 runs in our analysis (*n*=2,000). Sampling of the PDFs was completed using Latin Hypercube techniques in Q2K and random sampling in AT2K (note: the UNCAS module for AT2K was developed following NSTEPS review and thus less time was spent on its development). Identical parameter distributions and variance were used in each assessment of model uncertainty.

14.5 Results

The results of the Monte Carlo uncertainty analysis are shown in **Figure 14-4**. Overall, the output variance is quite interesting as both maximum daily pH and benthic algal accumulation (in the wadeable region) are fairly symmetric, but do show dispersion about the central tendency. The standard deviation is approximately 0.10-0.2 S.U. for pH over the longitudinal profile meaning that at least 68.2% of the simulations are within 0.2 pH units of the most probable outcome. Nearly 50-75% of all model realizations are below the stated pH criteria at the critical evaluation point thus we can be confident that between these percentiles, the proposed criteria would maintain uses in the river regardless of uncertainty in model forcings or parameterization used.

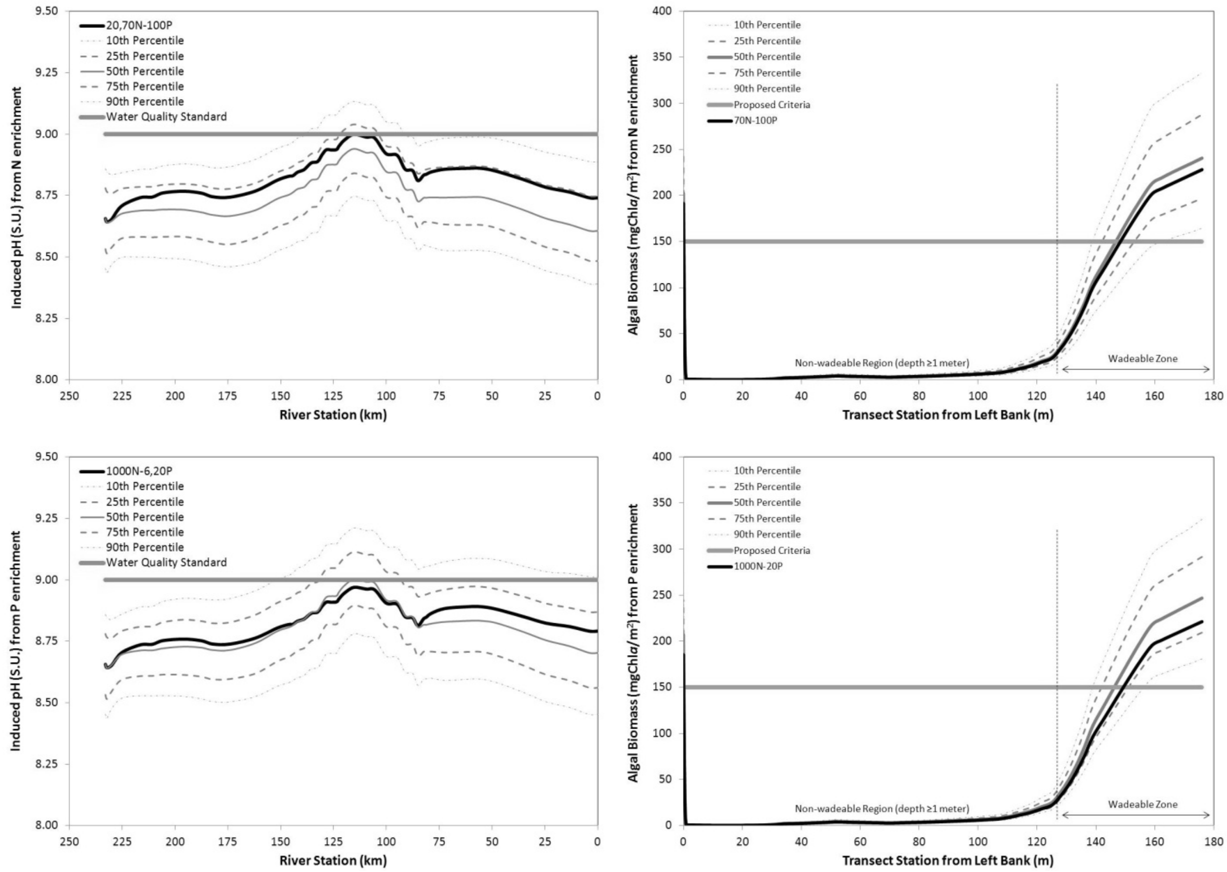

Figure 14-4. Estimated model output variance for ecological response endpoints.
(Top left/right panel). Uncertainty associated with pH and benthic algae predictions from the Q2K and AT2K models for the nitrogen enrichment scenario. (Bottom left/right panel). Same but for phosphorus enrichment. The 10^{th}, 25^{th}, 50^{th}, 75^{th}, and 90^{th} percentiles are shown along with the water quality standard and criteria run.

That said, there are differences between that of the error propagation runs and the calibrated model (for an individual nutrient level). For example, the model is oriented more toward the upper response regions for TN in comparison to TP, meaning that the criteria for TN if anything are more protective. However the TP calibration behaves more like the median uncertainty run which suggests the model behaves on average like the median of the simulated distribution which is as to be expected. In both cases the simulation drifts quickly from upstream boundary condition and the cause of this alteration is believed to be a function of changes in headwater flow and variability in incoming tributary, pH, temperature, suspended solids loads, and of course uncertain rate coefficient distributions, that cause general drift in the longitudinal pH profile. Regardless of the case, uncertainty in induced pH change will not have a significant impact on the derived nutrient criteria. Hence our recommendation is to use the previously identified thresholds for the river (**Section 13.0**).

Interpretation of wadeable benthic biomass is slightly more difficult. In this instance, uncertainty declines with depth ranging from nearshore regions that have large uncertainty (biomasses at the upper and lower quartiles range from about 200-300 mg Chl*a* m^{-2} with standard deviation of approximately 70 mg Chl*a* m^{-2}) to non-wadeable regions where uncertainty is negligible (due to light limitation). Median uncertainty approximates that of the calibrated model, however tails are much wider. For example the mean wadeable biomass at the 75^{th} percentile is around 190 mg Chl*a* m^{-2}, far greater than the standard.

Similarly, in both simulations, the calibrated model falls in the lower half of the response region so that approximately 50% of the Monte Carlo simulations exceed nuisance algal levels. Thus uncertainty in the benthic algae computation is apparent. In this regard we propose a margin of safety (MOS) be used to criteria determination to counter uncertainty with respect to algal biomass. While this is typically done numerically (through selection of a lower criteria), we have no way to ascertain a defensible MOS in this instance. As such, we recommend that a monitoring program be instituted to identify algal trends as the river moves closer to the proposed criteria (i.e., due to the fact that observational evidence is more reliable than numerical experiments in refining the proposed criteria). Details regarding the monitoring program and water quality standards refinement are expounded on in **Section 16.0**.

Finally, output variance for TN and TP was found to be inconsequential (**Figure 14-5**). Percentiles expand in the downstream direction as a consequence of model rates (and associated loadings) which is an artifact of our decision to not perturbate nutrient loads in the analysis. It should also be noted that the calibration response is at the lower end of TN and TP concentrations, between the 25th and 50th. Hence the model is calibrated towards the lower end of the plausible range which means the criteria are, if anything, conservative.

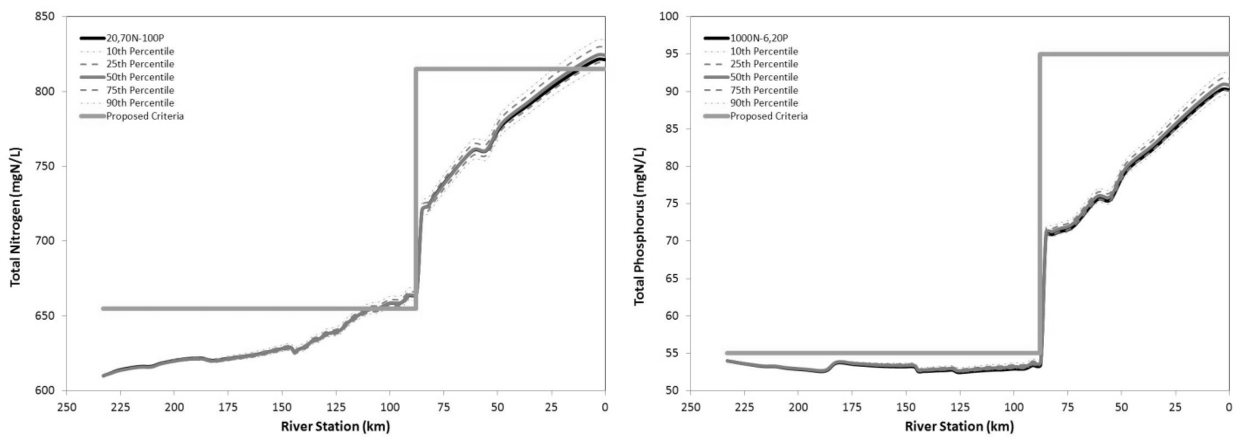

Figure 14-5. Estimated model output variance around computed nutrient criteria.
(Left panel). Output variance for TN according to the uncertainty runs detailed previously. (Right panel). Same but for TP. The changing variance in total nutrients is related to the decision did not perturbate nutrient loads as part of the uncertainty analysis (i.e., they were already iterated as part of the nutrient addition scenarios). It should also be noted that the proposed criteria are shown in the plots, but that the model run used to evaluate the uncertainty does not directly correspond with this condition; meaning that we used one of the 10 incremental runs described previously complete this analysis and thus criterion will not exactly match.

Overall, variance is lower than expected due to the fact that rate uncertainties are less influential on total nutrients than individual species (i.e., rates govern cycling between pools but still sum to the total) and that Bayesian inference techniques effectively narrow the range of allowable rate distributions from the broader literature array (thereby decreasing the allowable range in computed criteria). In both instances uncertainty is skewed to the right but criteria are near the upper end of the response region for pH, the middle of the range for benthic algae, and in the lower end of the range for nutrients. In this regard, we believe the proposed criteria are reasonably well-founded.

Based on these findings, DEQ is recommending that the numeric nutrient criteria identified in **Section 13.0** remain unaltered. These are listed below for reference:

- Upper river: 655 µg TN L^{-1} and 55 µg TP L^{-1}
- Lower river: 815 µg TN L^{-1} and 95 µg TP L^{-1}

Recommendations to prevent future water quality excursions are contingent on the assumption that the uncertainty of the nutrient outcomes is well-understood. While we believe this to be generally true, there are some caveats. First, we have demonstrated that for pH nearly 75% of all of the model realizations (including both parameter and rate uncertainty) associated with total N and P concentrations will maintain water quality standards. This in itself is useful, however, it is important to note that the expected response is based on assumed boundary condition and rate coefficient distributions which sometimes relied only on the literature or Bayesian inference. Thus the associated response region may have been truncated by such procedures.

With regard to benthic algae, uncertainty is much larger and more difficult to quantify. Due to lack of a better way to ascribe the wide range in simulated response, we have recommended a 5-year monitoring program be instated as the basis of our MOS for benthic algae. Monte Carlo outcomes generally suggest nuisance algal responses could manifest at greater than suggested levels (under different model parameterizations and loading conditions) and to accommodate such uncertainty, and allow proactive management of the river, such an approach is necessary (which happens to be feasible only because of the current nutrient status and assimilative capacity of the river). Still our model estimates have the greatest likelihood of being correct (according to expectation theory) and thus we acknowledge that while the uncertainty analysis is not without limitation, it is useful in understanding the relative magnitude of potential model outcomes.

15.0 COMPARISONS WITH OTHER NUTRIENT CRITERIA METHODS

Non-modeling or empirical methods for nutrient criteria development were also reviewed in conjunction with the mechanistic analysis to develop an understanding of commonly-reported nutrient, algae, DO, and pH relationships from the literature (as well as ecoregional numeric nutrient criteria recommendations from EPA). Historical data from the lower Yellowstone River (albeit limited) were also used to construct nutrient-algae relationships as suggested by EPA. The results of this these comparisons are presented below.

15.1 CRITERIA RECOMMENDATIONS FROM THE LITERATURE

There is growing consensus regarding nutrient thresholds and responses, and appropriate strategies for numeric nutrient criteria development in wadeable streams and rivers (Dodds and Welch, 2000; Snelder et al., 2004; Suplee et al., 2007; EPA, 2000b). In many of these efforts, total nitrogen (TN) and phosphorus (TP) concentrations are proposed to minimize nuisance algal growth, dissolved oxygen deficiencies, or other undesired water quality responses. Concentrations are on the order of 300-3000 µg TN L^{-1} and 20-300 µg TP L^{-1} according to the peer-reviewed studies in **Table 15-1**.

Proposed limits tend to be in agreement with values suggested for western Montana, and in some cases, were specifically developed for the area, e.g., the voluntary criteria recommendations for the Clark Fork River (Dodds et al., 1997) or percentile based approaches for wadeable streams in Montana (Suplee et al., 2007). However, the applicability of these studies toward larger and more turbid deep rivers, specifically the Yellowstone River, is debatable due to the differences outlined in **Section 2.0**.

Table 15-1. Examples of numeric nutrient criteria in the literature.

Author(s)	Location	Outcome or Recommendations
Dodds et al., (1997)	Montana, USA and data from 200 rivers worldwide	Mean targets of 350 µg TN L^{-1} and 30 µg TP L^{-1} total to keep benthic biomass ≤150 mg Chl*a* m^{-2}
Appendix A of Suplee et al., (2008)	Wadeable plains streams in the northern plains regions of eastern Montana	Suggested criterion of 1,120 µg TN L^{-1} to assure maintenance of dissolved oxygen levels above 5.0 mg L^{-1} (i.e., the state DO water quality standard)
Sheeder and Evans (2004)	Pennsylvania, USA	Suggested criteria of 2,010 µg TN L^{-1} and 70 µg TP L^{-1} based on data compilation from watersheds where biological uses were attained
Dodds and Welch (2000)	Multiple locations, USA and New Zealand	Suggests criteria of 250-3000 µg TN L^{-1} and 20-415 µg TP L^{-1} to limit benthic algae <200 mg Chl*a* m^{-2}, limits for oxygen deficit and pH excursion unknown
Biggs (2000b)	Periphyton Guidelines for New Zealand	1.0-26.0 µg SRP L^{-1} and 10-295 µg L^{-1} soluble inorganic nitrogen (SIN) in order to maintain benthic algal growth in wadeable streams and rivers to no more than 120-200 mg Chl*a* m^{-2} and 35 g AFDM m^{-2}. Author indicates that criteria should be chosen within the ranges based on the likely number of days that will pass between scouring high flow events.

Table 15-1. Examples of numeric nutrient criteria in the literature.

Snelder et al., (2004)	Mesotrophic rivers on the South Island, New Zealand	Proposed criteria of 59.8 µg L^{-1} soluble inorganic nitrogen (SIN) and 5.7 µg L^{-1} soluble reactive phosphorus (SRP) to keep benthic biomass <200 mg Chl*a* m^{-2}
Dodds et al., (Dodds et al., 2006)	Multiple locations, USA and New Zealand	Saturation points in nutrient-algal biomass correlations are identified. Above the saturation point, algal biomass is not likely to be controlled; thus, the saturation points represent potential criteria. These were 27 µg TP L^{-1} and 367 µ TN L^{-1}.

15.2 Ecoregional Recommendations from EPA

Level III Ecoregion ambient water quality criteria recommendations have also been proposed by EPA (2001). Those suggested for the Northwestern Great Plains region are shown in **Table 15-2**. Suggested values may or may not be appropriate for the area due to the fact that much of the water in the Yellowstone River originates from two other ecoregions, the Wyoming Basin and Middle Rockies ecoregions (**Figure 15-1**). Criteria recommendations for those regions are also shown.

Table 15-2. Level III ecoregion ambient water quality criteria recommendations.

Nutrient Parameters[1]	Northwestern Great Plains	Middle Rockies	Wyoming Basin
Total Nitrogen (µg L^{-1})	560	120	380
Total Phosphorus (µg L^{-1})	23	10	22

[1]Using historical data and reference sites, 25th percentile

15.3 Historical Nutrient-Algae Relationships on the Yellowstone River

Historical nutrient-algae data were also compiled for the lower river (i.e., Forsyth to Sidney) to identify the relationship between water column nutrient concentration and algal biomass. Results indicate that the amount of information available to make such determinations is sparse (i.e., very infrequent biomass monitoring), and that nutrient concentrations generally increase in the downstream direction without associated changes in algal density. Ambient water quality concentrations also rarely meet the N & P ecoregional criteria recommendations. Hence, either the small number of samples evaluated on the Yellowstone River is too small, or the proposed ecoregional criteria are inadequate.

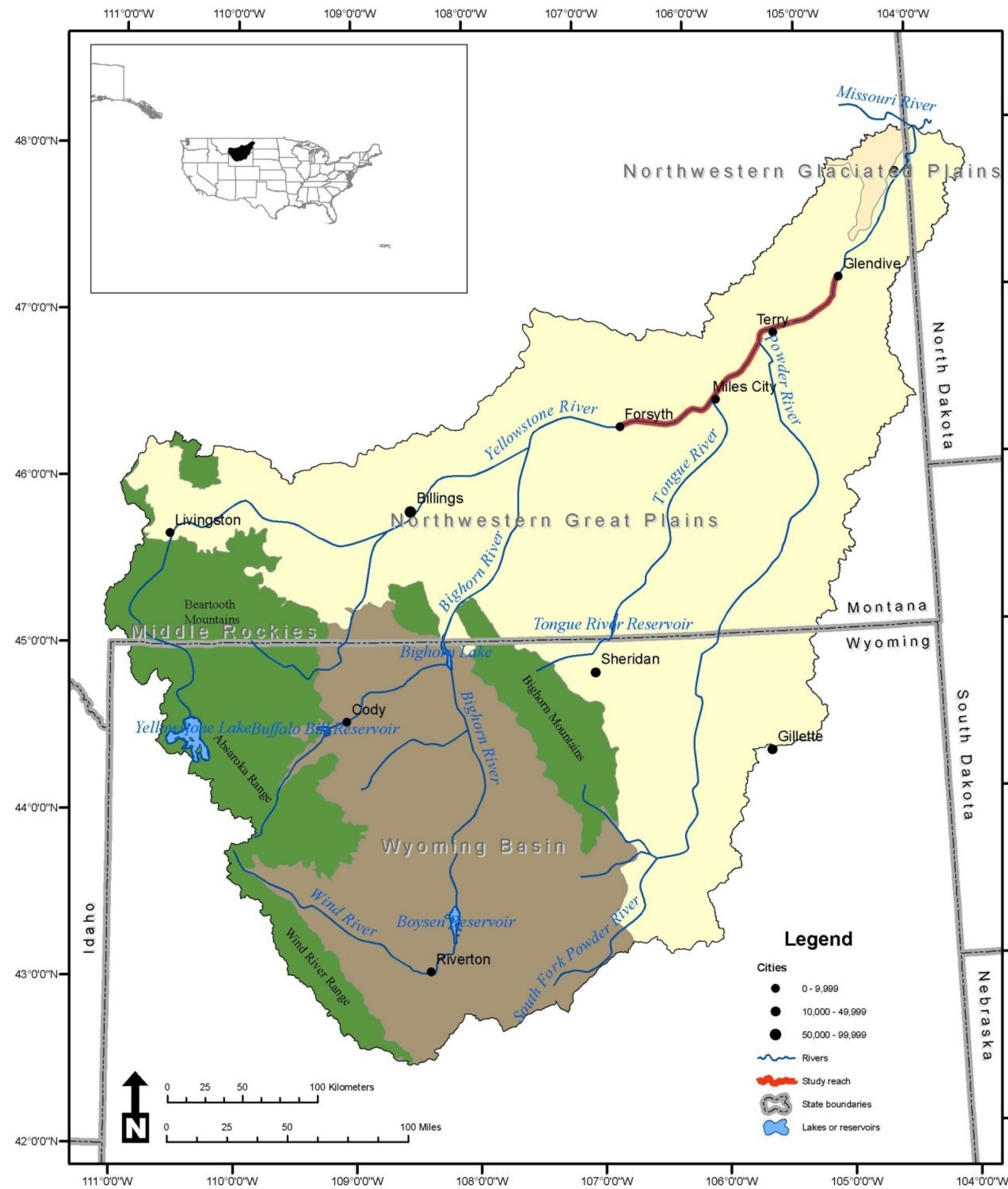

Figure 15-1. Level III ecoregions in relation to water quality criteria recommendations.
The study reach is located in the center of the Northwestern Great Plains ecoregion. However, much of the water flowing into the river originates from the Middle Rockies and Wyoming Basin and ecoregion. Hence it is difficult to determine which criteria recommendations from **Table 15-2** should really apply to the section in question.

From paired nutrient-algae data that were available on the lower river (i.e., Forsyth to Sidney, within the same week during the low-flow August- September period) we found that TN and TP explain 34% and 1% of the variance in benthic biomass, respectively, using log-linear regression. When extrapolated to a concentration reflective of nuisance biomass (as defined by 150 mg Chl*a* m^{-2}), threshold nutrient concentrations would be 505 µg L^{-1} of TN and 45 µg L^{-1} of TP to limit nuisance alga (**Figure 15-2**) (note that extension of the regression beyond the data is not recommended by DEQ and we suggest the TP regression not be considered at all). The analysis excludes data in the shallower reaches of the river near Billings and Laurel, and also does not consider the differences in USGS and DEQ collection methodologies (Porter et al., 1993).

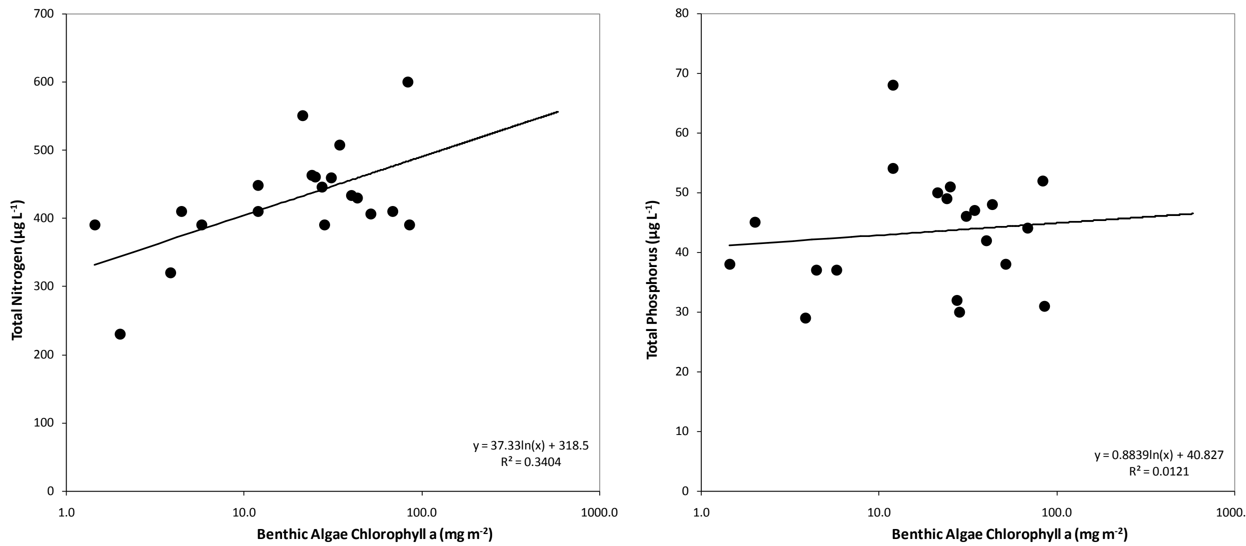

Figure 15-2. Relationship between nutrients and benthic biomass on the lower Yellowstone River. (Left panel) Relationship between TN and benthic algae. (Right panel) Same but for TP. Data shown for the months of August and September (data from Forsyth to Sidney, MT).

Clearly coefficients of determination (r^2) for the regressions are weak, which is typical. Correlation coefficients between river nutrient concentrations and benthic algal biomass are usually no better than about 0.4 (Chételat et al., 1999; Dodds et al., 1997; Dodds et al., 2002; Dodds et al., 2006). This stems partly from the noise of the actual field and analytical techniques, but more so from the fact that the life cycles of benthic algae (growth, death, and senescence) are variable and can greatly alter the total nutrients in aquatic settings over short periods. Finally, advection physically translates information downstream which means algal biomass measured at one point may better correlate with nutrients elsewhere (e.g., upstream). Thus, these correlations should be used as initial estimates only. Interestingly, the result are comparable to ecoregional recommendations and illustrate the difficulty in using regression analysis to describe patterns between two variables whose linkage are not conservative relative to one another. Extension of the analysis using multivariate regression provides little improvement of the predictive power of the equation. The r^2 increases slightly (36%) albeit the adjusted r^2 actually declines which suggests there is little improvement in explanation of variance with the addition of multiple degrees of freedom.

15.4 Summary of Findings

Three independent methods were used to provide comparative estimates of numeric nutrient criteria endpoints for the lower Yellowstone River. This included a review of nutrient, algae, DO, and pH relationships from the scientific literature, ecoregional recommendations from EPA, and analysis of historical nutrient-algae relationships on the river itself. Generally, there are a large range of plausible outcomes for which criteria could potentially exist, which is further confounded by limitations such as available data, spatial transferability, and uncertainty in data methods.

A matrix of these outcomes is presented in **Table 15-3**, which collectively illustrate the need for the modeling study and the apparent difference that can result when using site-specific, as opposed to large dataset empirical approximations.

Table 15-3. Summary of outcomes from varying approaches to assess numeric nutrient criteria.
Recommendations for the lower Yellowstone River.

Source	TN Outcome ($\mu g\ L^{-1}$)	TP Outcome ($\mu g\ L^{-1}$)
Literature range	300-3,000	20-300
Level III ecoregional recommendation	560	23
Historical nutrient-algae data	514	43
Site-specific water quality model[1]	**655 / 815**	**55 / 95**

[1]Big Horn River to Powder River / Powder River to state-line, respectively, this study.

15.5 Expert Elicitation Regarding Findings

Anonymous reviews from EPA's Nutrient Scientific Technical Exchange Partnership & Support (NSTEPS) were completed as part of this project to satisfy peer review requirements/expert elicitation and are in **Appendix D** (along with DEQ's responses to these comments). Finally, a public comment period was also open through November 30, 2011 for which very few responses were provided.

16.0 SUMMARY AND RECOMMENDATIONS

An alternative approach toward numeric nutrient criteria development was established via this project. It consisted of: (1) use of mechanistic water quality models to determine the stressor-response relationship between nutrients and key water quality endpoints (DO, pH, benthic algae, etc.), (2) derivation of nitrogen and phosphorus criteria endpoints for a large river using those tools, (3) evaluations of whether modeled criteria are consistent with other nutrient endpoint techniques, and (4) compilation of our findings such that other States or Tribes can make informed decisions about large rivers in their regions.

The work was completed on a 232.9 km segment of the Yellowstone River in eastern Montana from Forsyth to Glendive, MT (Waterbody ID MT42K001_010 and MT42M001_012) with corroboration of a much larger reach (586 km). In the focus area, we developed criteria for two distinct reaches: (1) the Bighorn River to the Powder River (for which our model characterizes approximately half of the reach); and (2) Powder River to state-line (which has a similar extent). Different water quality parameters led to different nitrogen and phosphorus criteria (other large rivers may be similar). The distinction comes from longitudinal changes in river variables like depth, turbidity, and light.

In the upper and less turbid reach of the lower Yellowstone River (Forsyth to Powder River), river pH proved to be the most sensitive water quality variable. An induced pH shift >9.0 indicated impairment. Thus in this region, a large proportion of the river could respond photosynthetically to increased nutrients. The lower river was less sensitive and therefore near-shore nuisance algae (<150 mg Chl*a* m^{-2}) were most important. Both Q2K and AT2K were essential in making these determinations. Based on these findings, it was recommended that criteria be set at 655 µg TN L^{-1} and 55 µg TP L^{-1} from the Big Horn River to the Powder River confluence, and 815 µg TN L^{-1} and 95 µg TP L^{-1} from the Powder River confluence to state line to prevent unacceptable variation in pH or nuisance algae.

Findings were also compared with existing information in the literature to identify the applicability of the estimate in the context of previous studies. Because the Yellowstone River is deep and moderately turbid/light limited, our criteria recommendations are higher than typically suggested for wadeable streams and rivers in either the scientific literature, or from the EPA. This is a function of two factors. First, the criteria found in the literature were mainly developed for wadeable streams which are shallow. Secondly, wadeable streams are often less turbid than larger rivers. Hence light-limitation was an important component of this study and we integrated its effect into river management. Such a consideration makes the transfer of wadeable stream empirical approaches to large rivers undesirable, and the use of mechanistic models very appealing. Finally, we suggest that a concerted national effort to gather data on large rivers be conducted, including the use of modeling and experimental research. This should include work by fishery biologists to learn more about the effects of dense algal mats on 0+ age fishes which use the shallow near-shore margins of large rivers as nursery grounds.

16.1 FOLLOW-UP FOR THIS WORK

Lastly, we recognize that despite our best efforts, the criteria in this document are imperfect. Uncertainty is inherent within all water resource systems, embedded within the science and engineering we use to describe them. We have acknowledged and quantified this uncertainty to the extent possible through error analysis and implementation of modeling best-management practices. However, this does not preclude the possibility that such criteria may need to be re-visited in the future. We are fortunate enough in this instance that we will have the opportunity to analyze water quality data, do model post-audits, and adjust management objectives and criteria over time (if necessary) as the river moves closer to the suggested criteria. As a consequence, we recommend further surveying/sampling of the Yellowstone River and additional computer modeling in the form of model post audits at periodic intervals based on the newly acquired data.

Triennial monitoring of the lower Yellowstone River is one possible approach to accommodate uncertainty in the benthic algae predictions. This is consistent with the Clean Water Act which requires states to review water quality standards every 3 years. The model-derived criteria described in this document will eventually become standards once adopted by the state. Thus periodic reviews will be inevitable. As such, it is probably not necessary to do model post-audits, or collect additional corroboratory data on such a tightly defined schedule. Proper development of the lower Yellowstone River model relied on the collection of field data during low-flow years (e.g., 2000, 2007) near to the design flow (14Q5). Because, these low flows do no occur every year (they occur at least statistically about every five years), we recommend that a sampling plan for key model parameters and endpoints (e.g., pH, wadeable region benthic algae) be developed that could then in turn be implemented when future low-flow baseflows do occur.

17.0 REFERENCES

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APPENDIX A - QUALITY ASSURANCE PROJECT PLAN (QAPP) AND SAMPLING AND ANALYSIS PLAN (SAP)

USING A COMPUTER WATER-QUALITY MODEL TO DERIVE NUMERIC NUTRIENT CRITERIA FOR A SEGMENT OF THE YELLOWSTONE RIVER

Quality Assurance Project Plan (QAPP)

Prepared for:

MONTANA DEPARTMENT OF ENVIRONMENTAL QUALITY
Water Quality Standards Section, Water Quality Planning Bureau
P.O. Box 200901
Helena, MT 59620-0901

Approvals

______________________________	______________
Michael Suplee (WQ Standards Section)	Date
______________________________	______________
Kyle Flynn (Data Management Section)	Date
______________________________	______________
Michael Van Liew (Data Management Section)	Date
______________________________	______________
Bob Bukantis (WQ Standards Section Supervisor)	Date
______________________________	______________
Michael Pipp (Data Management Section Supervisor)	Date
______________________________	______________
Rosie Sada (WQ Monitoring Section Supervisor)	Date
______________________________	______________
Mark Bostrom (QA Officer)	Date

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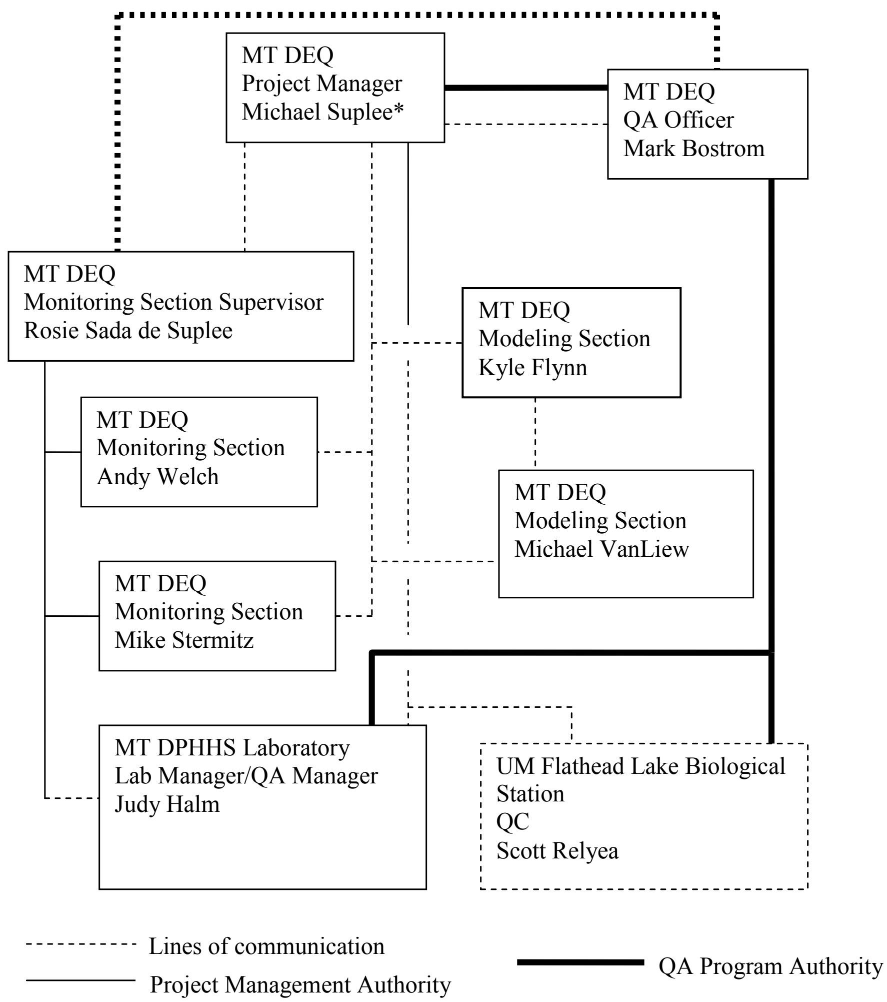
APPENDICES
Appendix A *Data Collected on the Yellowstone River, Aug. 2006*

1.0 Project/Task Organization

This document presents the research quality assurance project plan (QAPP) for collecting and analyzing data from a segment of the lower Yellowstone River. This work is being undertaken for the purpose of developing a computer water-quality model. As such, in addition to quality assurance descriptions for field-collected data, detailed descriptions of how the computer model will be calibrated and validated are also provided herein. Field data collection and model setup/calibration-verification will be done by staff of the Montana Department of Environmental Quality (DEQ). Analysis of samples will be undertaken by the University of Montana Flathead Lake Biological Station and the Montana Department of Public Health and Human Services Environmental Laboratory. Michael Suplee, Ph.D., will provide overall project oversight for this study. The following chart shows the roles of the various entities and their relationship to one another.



* In the field, Suplee will have general management authority for sampling decisions affecting the crew.

2.0 Introduction

2.1 Background

In Montana, designated beneficial uses of state surface waters include growth and propagation of fish and associated aquatic life, drinking water, agriculture, industrial supply and recreation (ARM 17.30.621 through 629). Eutrophication, or the over enrichment of waterbodies by nutrients (usually nitrogen [N] and phosphorus [P]), can cause nuisance algal growth, alter aquatic communities and result in undesirable water-quality changes that can impair these beneficial uses (Freeman, 1986; Arruda and Fromm, 1989; Welch, 1992; Dodds et al., 1997). Since 2001, the Montana Department of Environmental Quality (DEQ) has been working to develop numeric nutrient criteria for surface waters. The intent of numeric nutrient criteria is to protect waterbodies and their associated beneficial uses from the adverse effects of eutrophication. DEQ has made good progress in nutrient criteria development for wadeable streams and small rivers of the state by integrating stressor-response and reference-based approaches (Varghese and Cleland, 2005; Suplee et al., 2007). However, criteria development for large rivers (e.g., Yellowstone, Missouri rivers) has not yet been undertaken. Herein, we propose an approach to developing numeric nutrient criteria for a large river segment using a mechanistic, computer water-quality model. This differs from the methods DEQ has used thus far for wadable streams.

2.2 Problem Definition

Montana DEQ believes that a nutrient-criteria derivation technique for large rivers (defined loosely here as river segments with a Strahler order $\geq$ 7, 1:100,000 scale; Strahler, 1964) should differ from DEQ's wadeable-stream approach because (1) the ability to identify "reference" watersheds for the state's large rivers, per the wadeable-stream methods outlined in Suplee et al. (2005), is infeasible, and (2) using reference "segment-sheds" for large rivers (Fig. 1), per proposed EPA methods (M. Paul, personal communication) may not sufficiently address cumulative affects from upstream of the reference segment-shed. Without being able to identify reference watersheds for these large systems, setting benchmarks based *only* on reference segment-sheds becomes highly debatable. Further, in the absence of reference one is left with the task of defining a water quality impact without the benefit of knowing what un-impacted looks like.

Because of the issues outlined above, we believe that a reasonable way to proceed toward developing nutrient criteria for large rivers is to identify the valued ecological attributes of the system of concern, clearly state how these relate to beneficial uses, and then determine when those attributes have been impacted, via simulation modeling. Valued ecological attributes are defined as ecosystem characteristics that directly or indirectly contribute to human welfare (Stevenson 2006), and are closely allied with beneficial uses. Determining when valued ecological attributes/beneficial uses have been impacted can be difficult, and requires both value judgments and scientific understanding. The more clearly an impact threshold to a valued ecological attribute/beneficial-use can be defined, the more defensible will be the nutrient criteria that prevent the impact.

We propose developing numeric nutrient criteria on a large river segment through mechanistic water-quality modeling by considering two specific valued ecological attributes that can be directly linked to beneficial uses. Because there are clear impact thresholds for the following, we intend to model these on the Yellowstone River:

1. Dissolved oxygen levels, which are required by state law to be maintained ≥ 5 mg/L in order to protect aquatic life and fishery uses (early life stages; DEQ 2006a).
2. Benthic algae levels, which should be maintained below a nuisance threshold {ARM 17.30.637(1)(e)}to protect recreation uses. Based on a 2006 DEQ scientific public opinion survey addressing when the recreational use of rivers & streams becomes impacted by excess benthic algae, algae levels should be kept below 150 mg Chl *a*/m^2 (Larix 2006; also see study results at: http://www.umt.edu/watershedclinic/algaesurveypix.htm.).

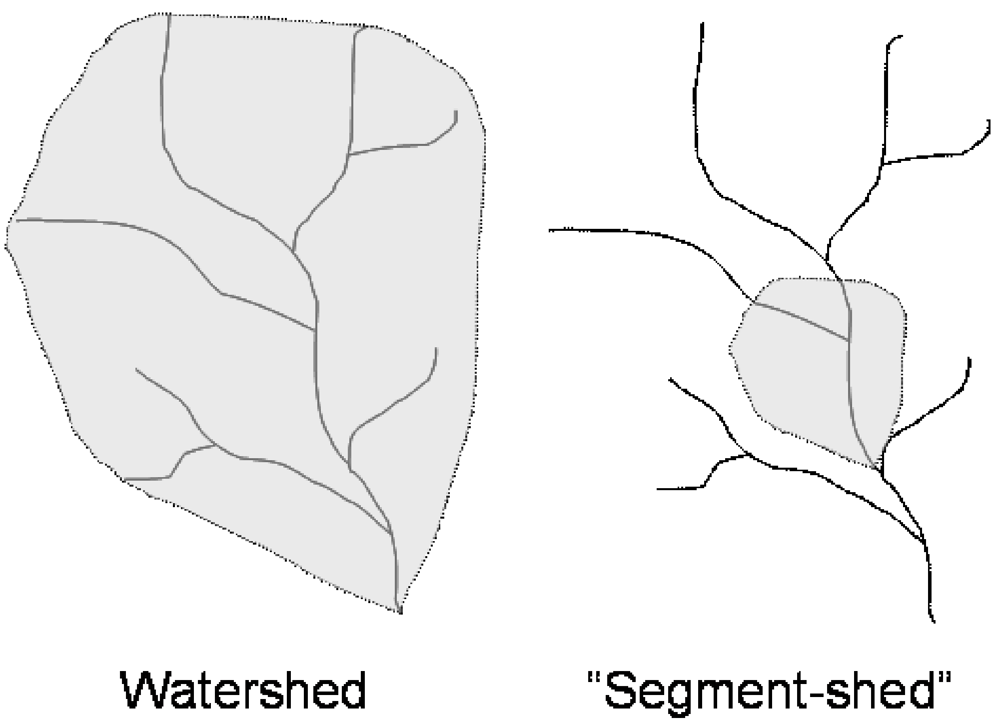


Figure 1 – Conceptual diagram illustrating the watershed versus segment-shed ideas. The segment-shed is recommended for considering the area contributing to land cover/land use information above a large river site.

The QUAL2K model was selected by DEQ for the Yellowstone project due to its frequent use in dissolved oxygen (DO) modeling and its ability to simulate benthic algae levels (Drolc and Koncan, 1996; Chaudhury et al., 1998; Chapra, 2003, USGS SMIC 2005). Although the benthic component of the model has not been well reported on in the literature, empirical relationships between river nutrient concentrations and benthic algae density have been reported (e.g., Dodds et al. 1997). Butcher (2006) reported that the default parameters in computer models like QUAL2K need to be adjusted to come in to alignment with the empirical results of published studies (e.g., Dodds et al., 1997). DEQ acknowledges that there may be inconsistencies between mechanistic models and empirical nutrient-algae relationships, and we will carefully assess this during model development. To help cross-check the modeled criteria, two other nutrient criteria development techniques will be considered. First, a quasi-reference approach will be used whereby the modeled criteria will be compared to nutrient concentrations from an upstream reach of the Yellowstone River perceived to have minimal water quality impacts ("comparison" site; Suplee, 2004). Second, the model output nutrient concentrations will be compared to concentrations from river and stream empirical models (Dodds et al., 1997; Dodds et al., 2006). These efforts will help cross-check the model output results.

Based on preliminary discussions among the principle authors of this QAPP (Suplee, Flynn and Van Liew, DEQ), it was decided to undertake the modeling work on a segment of the lower Yellowstone River. The segment was selected because it has a minimal number of point sources, a fairly well established gaging network, and fairly characteristic non-point source impacts. Further, Miles City (within the study reach) is currently in the planning phase of upgrading its wastewater treatment plant. As part of this upgrade, Miles City is very interested in potential future numeric nutrient criteria that may apply to the Yellowstone River. To assure that this segment of the Yellowstone River was appropriate for the project, reconnaissance trips by DEQ staff were undertaken along the river from August 14th – 19th 2006, February 7th – 8th 2007, and June 21st-22nd, 2007. During these trips notes were taken on the accessibility of various locations along the reach, candidate locations to install monitoring equipment were identified, and field measurements of stream velocity, DO, temperature and sediment oxygen demand (SOD) were made.

3.0 Project/Task Description

3.1 Primary Question, Objectives and River Reach Description

The project outlined in this QAPP is designed to answer the following question:

> *In a segment of the lower Yellowstone River, what are the highest allowable concentrations of nitrogen and phosphorus which will not cause benthic algae to reach nuisance levels and/or dissolved oxygen concentrations to fall below applicable State water quality standards?*

As described previously, DEQ intends to use a computer model that will answer this question. The Yellowstone River segment to be modeled will extend from the Rosebud West fishing access site (FAS) at 46.2646 N latitude, 106.6959 W longitude (just upstream of USGS gage 06295000Yellowstone River at Forsyth, MT), to the old Bell Street Bridge at 47.1055 N latitude, 104.7198 W longitude, which is at the same location as USGS gage 06327500, Yellowstone River at Glendive, MT (Fig 3.1).

Once the model is calibrated and validated (Chapra, 2003; Wells, 2005) for this reach, DEQ will simulate a critical low-flow condition (i.e., 7Q10) during which nuisance algae growth and depressed DO concentrations are likely to be most severe. We will then vary N and P concentrations in the model to affect changes in the DO and algae-level outputs from the model. The highest input N (dissolved organic N, NO_3, and NH_4) and P (dissolved organic P and inorganic P) concentrations that do not cause nuisance algae growth and/or exceedences of the DO standard under these low-flow conditions can be used as the numeric nutrient criteria for this river segment during the base flow period. Total to soluble nutrient ratios — as currently manifested in the river — will be used to derive total nutrient criteria concentrations, which are the end goal of this project. If a single nutrient (e.g., N) is clearly limiting in the river, the Redfield ratio (Redfield, 1958) will be used to set the accompanying, non-limiting nutrient criterion.

In order to accurately calibrate & validate the model, DEQ intends to measure a large number of factors that directly or indirectly influence DO and benthic algae density in the river. These include forcing functions such as meteorology and hydrology, and state/rate data, which are described in subsequent sections. Our basic assumption is that direct measurement of key parameters will increase the confidence in the model predictions and reduce the uncertainty in model parameters and coefficients (Melching and Yoon, 1996; Barnwell et al., 2004). The modeled criteria can also be compared to nutrient concentrations from the upstream comparison site on the Yellowstone River perceived to have minimal water quality impacts, and to results from applicable empirically-derived models (Dodds et al. 1997; Dodds et al. 2006).

3.2 Project Design

3.2.1 Model Selection

The criteria for selecting a model were (A) relative simplicity and (B) its ability to answer our question and yield adequate accuracy (Krenkel and Novotny, 1979; Chapra, 2003). QUAL2K, MIKE11, WASP, and CE-QUAL-W2 were all considered. QUAL2K was ultimately selected by DEQ due to frequency in application for TMDL planning and dissolved oxygen modeling (Drolc and Koncan, 1996; Chaudhury et al., 1998; Rauch et al., 1998; Chapra, 2003, USGS SMIC, 2005), endorsement by the EPA (EPA, 2005) and because it offers relative simplicity as a one-dimensional steady-state model (e.g., it assumes the channel is well mixed vertically and longitudinally and meteorology, hydrology, and hydraulics remain constant during the simulated time-step). QUAL2K can also be run in a quasi-dynamic mode to simulate diurnal DO and temperature variations (Mills et al., 1986; Chapra and Pelletier, 2003). The other models that were considered are fully dynamic, but are more complex and require more data input, and one (MIKE11) is proprietary. QUAL2K is also able to simulate benthic algae growth, a key parameter of interest in this study, which its predecessor (QUAL2E) could not.

DEQ measured DO and temperature during the summer 2006 reconnaissance trip to verify that basic modeling assumptions such as complete mixing (vertically and laterally) would not be violated at any of the sites visited. The results of the field work are documented as part of this QAPP (Appendix A) and clearly show that the initial model assumptions are satisfactory. In addition, the steady state flow assumption was evaluated using the anticipated headwater flow at the Forsyth USGS gage. Over a one week period from August 15-22 (the anticipated period for modeling) flow changed 6% of the period of record. This is considered acceptable for steady-state modeling.

3.2.2 Model Development and General Design

Seven major river subreaches, which comprise the entire Yellowstone River study reach, were identified for model development. Each of the seven major subreaches will be further subdivided based on hydrology, hydraulics, known water quality changes, etc. such that approximately 30-40 total modeling subreaches are anticipated. The seven major subreaches are (Figure 3.1): (1) Rosebud West FAS to the Cartersville Canal return flow, (2) Cartersville Canal return flow to the Tongue River confluence; (3) Tongue River confluence to Kinsey Bridge FAS, (4) Kinsey Bridge FAS to the Powder River/Shirley Main Canal confluence; (5) Powder River/Shirley Main Canal confluence to the O'Fallon Creek confluence, (6) the O'Fallon Creek confluence to eleven miles upstream of Glendive, MT, and (7) eleven miles upstream of Glendive to the Bell Street Bridge in Glendive, MT. A YSI 6600EDS sonde will be deployed at each of these breakpoints and will measure the necessary parameters for water-quality model calibration (temperature, DO, pH, Chl *a*, etc.). Additionally, an upstream site will be located at the Buffalo Mirage FAS just upstream of Laurel, MT. The comparison site is on an upstream segment of the Yellowstone River currently considered to fully support all its uses (2006 Integrated Report), and is near or within the ecotone where the river changes from a cold-water to a warm-water fishery.

Figure 3.1. Yellowstone River QUAL2K Monitoring Locations

Depth-and-width integrated sampling is planned to be coincident with the YSI locations (as well as for major tributaries and the comparison site), and is designed to bracket water quality and other measured parameters at the upstream and downstream ends of each of the seven subreaches. Based on a review of USGS gage sites, DEQ has concluded that only two natural tributaries in the modeling study reach will require monitoring during the "low flow" monitoring period; the Tongue and Powder River. However, any major tributaries that are flowing near their mouths during the synoptic sampling runs (e.g., O'Fallon or Rosebud creeks) will be sampled opportunistically. And because of their likely influence on water quality, several irrigation canals will be sampled. The Cartersville, Kinsey, Shirley, Terry Main and Main canals will be monitored for water withdrawal volume at their upper limits. They will also be sampled for quality/quantity at their confluence (inflows) with the river, when identifiable return points exist, to establish the influence of their return flow. In some cases (e.g., Bonfield FAS, Pirogue Island State Park, Terry Bridge etc.), monitoring sites will also be near the middle of a subreach. Benthic/rate measurements will be completed at these locations along with instantaneous water quality to provide a check to assure no major water quality changes have occurred within the subreach.

Water sample and other data will be collected during two 8-10 day periods in August and September 2007, for the purpose of establishing calibration and validation datasets for the simulated water quality state variables. This split-sample calibration-validation approach is appropriate for a Level 1 confirmation in which the model is tested using different meteorological and boundary conditions from which it was calibrated (Chapra, 2003). This "low-flow" period is considered representative of the critical limiting period where conditions of nuisance algae and/or low dissolved oxygen would limit beneficial uses in the Yellowstone River.

Mills et al. (1986) recommended that sampling occur at points where water quality standards may be violated, in addition to boundary conditions and key tributary breaks. Benthic measurements are planned for downstream of Forsyth, Miles City and Terry, to observe potential responses of the river to WWTP inputs. This has been initiated due to the fact that midday DO concentrations were measured below 5 mg/L during the 2006 field visit (Appendix A) in Miles City, and heavy nuisance algal growth was observed near Miles City at the Roche Jaune FAS.

Other important forcing data necessary for modeling include point source discharges, diffuse sources (non-point), and meteorological data. Municipal permitted point source discharges are located at Forsyth, Miles City, Terry, and near the border of Fallon/Prairie County. Nutrient and other data collected as part of the MPDES permits from point sources will be gathered from the DEQ Permitting and Compliance Bureau. If these are not deemed appropriate for modeling purposes, an additional effort will be made to organize a data collection effort at these point sources over the monitoring period. Non-point source data (e.g. groundwater monitoring) will not be collected as part of this project. Rather, the Montana Bureau of Mines and Geology (MBMG) GWIC database will be consulted to establish quality constituents of groundwater accretion. A cursory review of this database revealed a number of groundwater water-quality sampling locations in Rosebud, Custer, Prairie and Dawson counties.

Meteorological data are being collected at a number of stations independent from this study. Communities along the targeted reach such as Forsyth, Miles City, Glendive, etc. have NOAA or BOR weather stations that provide the necessary data for modeling. Those stations with hourly meteorological observations of either air temperature, wind speed, relative humidity, solar radiation or cloud cover are identified below (see also Figure 3.1):

1. Buffalo Rapids - Terry, MT (BRTM), BOR Agrimet
2. Buffalo Rapids - Glendive, MT (BRGM), BOR Agrimet
3. Glendive AWOS (WBAN 24087), NOAA
4. Miles City Municipal Airport (WBAN 24037, COOP ID 245690), NOAA
5. Forsyth W7PG-10 (AR184), NOAA

3.2.3 Sediment Oxygen Demand Measurements Using Benthic Chambers

Sediment Oxygen Demand in the Yellowstone River, August 2006. Sediment oxygen demand (SOD), or river-water oxygen consumption originating from the sediments, can be an important component of river DO dynamics (Bowman and Delfino, 1980; Matlock et al., 2003). We undertook SOD measurements at two locations in our targeted reach of the Yellowstone River in August 2006, using the sediment-core SOD method (Edberg and Hofsten, 1973). SOD was measured in paired, opaque core samples (Fig. 3.2) collected at the Roche Jaune FAS and the Fallon Bridge FAS. All SOD values were corrected for the water-column oxygen demand (WOD) of the water above the sediment cores (Suplee and Cotner, 1995). At the Roche Jaune FAS the WOD was undetectable, while SOD was (on average) 0.5 g O_2 m^{-2} day^{-1}. However, the greatest proportion of DO demand was probably associated with thick beds of filamentous *Cladophora* at the site (we did not measure DO demand of the *Cladophora*, and no *Cladophora* was present on the sediment cores we collected). At the Fallon Bridge FAS, where no attached *Cladophora* was noted, WOD was 1.1 g O_2 m^{-3} day^{-1} and SOD was (on average) 0.7 g O_2 m^{-2} day^{-1} (CV = 22%). SOD accounted for about 38% of the total DO demand in the river at the Fallon Bridge FAS, when WOD was integrated over the mean river water depth of 1 m.

From these preliminary measurements we concluded that SOD can be a major part of the river's DO dynamics, and should be directly measured for purposes of QUAL2K calibration and validation. Although QUAL2K calculates SOD based on diagenesis of settling organic carbon, temperature, etc., it also allows the user to input supplementary SOD if the model is underestimating measured SOD values (Chapra and Pelletier, 2003).

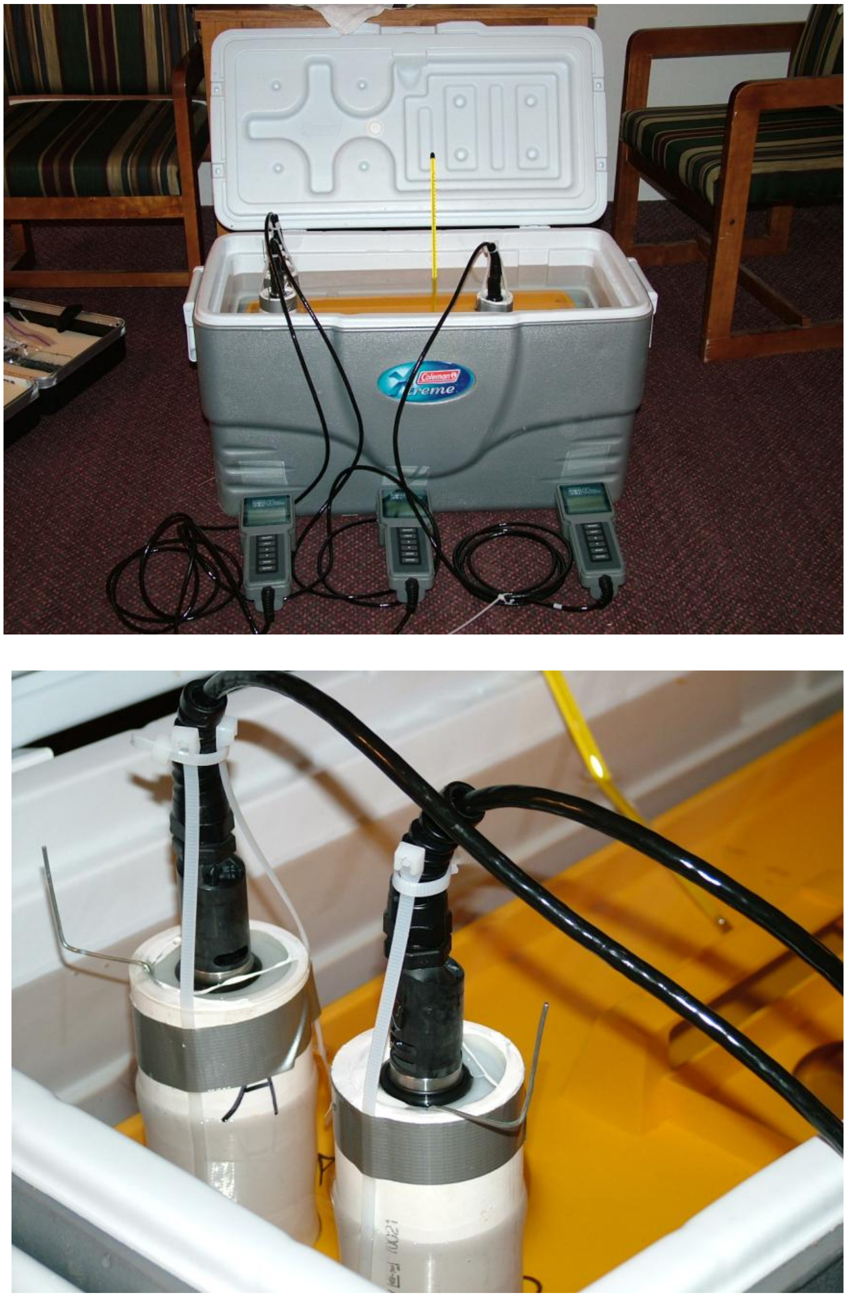


Figure 3.2. Measurement of sediment oxygen demand in sediment core samples, Yellowstone River, August 2006. A. Paired sediment cores in their water bath, with YSI model 85 DO meters attached. The tube on the right only contained river water and was used to measure BOD. B. Close-up of the sealed sediment cores and attached YSI DO probes. The metal wires were attached to paddles used to stir the water above the sediments just prior to taking the DO measurements. Water bath temperature was maintained at the temperature measured in the river during sediment collection.

In Situ Measurement of SOD Using Benthic Chambers, Summer 2007. EPA indicates that *in situ* measurements of SOD are preferable to laboratory sediment-cores techniques (Mills et al., 1986). And although sediment cores were used for the August 2006 reconnaissance, it is also the opinion of Suplee (of this QAPP) that *in situ* SOD methods should be used in 2007, based on past experience measuring SOD (see Suplee and Cotner, 1995; Suplee and Cotner, 2002; Cotner et al., 2004). This is because the bed of the Yellowstone River was comprised of coarse and fine gravel, making the collection of undisturbed sediment cores quite difficult. It is also difficult to simulate flow velocities across the sediments in a sediment core. Simulation of river velocity over the sediments is important to accurate measurement of river SOD (Hickey, 1988; Mackenthun and Stefan, 1998).

We intend to use *in situ* opaque SOD chambers similar in design to that of Hickey (1988; Fig 3.3). His chamber design is specialized for river use and can simulate *in situ* river velocities. Opaque chambers allow for simulation of nighttime SOD, which is the critical time period when river DO is the lowest and which is of most interest to us. A chamber volume/surface ratio (L/m^2) of < 100 generally provides good declines in DO over efficient time frames (2-12 hours), therefore a ratio of 70 will be used for our chambers. The chamber pump will simulate velocities across the sediment ranging from zero to 0.4 m sec^{-1}, which encompasses the range of near-bottom water velocities measured in the river in August 2006 (Appendix B). A flexible skirt of rubber or a similar inert material will be attached around the circumference of the chamber where it interfaces with the sediments. Due to the river bottom's composition, we will probably not be able to press the chambers in to the sediments very deeply, therefore the skirt will help provide an additional seal between the sediments and the enclosed water in the chamber.

Solute Fluxes to be Measured Using the ***In Situ*** *Benthic Chambers*. Di Toro et al. (1990) recommended that if SOD is being measured *in situ*, dissolved methane and ammonia should also be measured, and QUAL2K allows the user to prescribe these fluxes (Chapra and Pelletier, 2003). The flux of total dissolved inorganic carbon (DIC) will also be measured. The sediment DIC flux will be compared to the DO flux in order to calculate the respiratory quotient (RQ; CO_2 flux/O_2 flux), which will show if organic material on the river bottom is being metabolized by largely aerobic or anaerobic processes (Wetzel, 1983; Suplee and Cotner, 2002). This information will be valuable for model calibration.

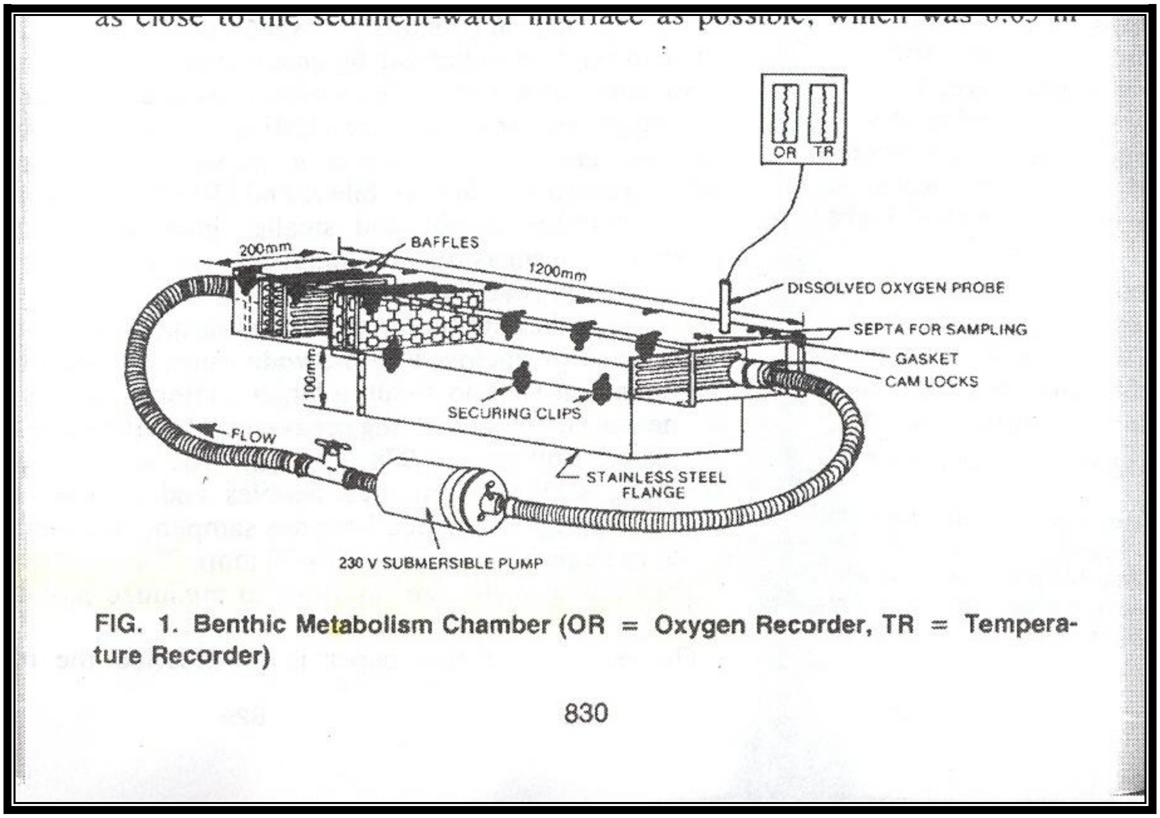


FIG. 1. Benthic Metabolism Chamber (OR = Oxygen Recorder, TR = Temperature Recorder)

Figure 3.3. General diagram of the flow-adjustable SOD chamber proposed for use in the project, from Hickey (1988). The final design will be a modification of this basic layout. For example, a flexible skirt will be added around the circumference of the chamber to assure a good seal to the river bottom in cases where the device cannot be pressed very deeply in to the sediments.

3.2.4 Other Rate Measurements

QUAL2K allows the user to input maximum phytoplankton photosynthesis rates at a given temperature (k_{gp}[T]; Chapra and Pelletier, 2003). These will be measured directly, methods for which are outlined in the SAP. Simulated night-time DO uptake by *Cladophora spp*. will be measured at locations (e.g., Miles City) where dense beds are present and likely influence DO dynamics.

3.2.5 Other Benthic Measurements

Estimate of Algal Growth Cover and Proportion of Applicable Channel SOD. The % river bottom cover by algae and the % river bottom to which SOD measurements apply will be estimated at cross sections of specified sites. Both of these parameters can be prescribed by the user in QUAL2K. During the transect collection of benthic algae, a record will be made at each sampling locale indicating the degree and type of algae coverage. QUAL2K also allows the user to dictate the proportion of river bottom that SOD measurements apply towards, under the assumption that only a proportion of the river bottom is capable of generating a significant SOD.

We will estimate in the field the proportion of the river bottom along the transect that has velocity and depth characteristics similar to the sites where SOD was measured. Our assumption is that areas of high velocity and scouring (e.g., river thalweg) will have lower SOD than the slower, more depositional parts of the river, where SOD measurements will be made. The model will be setup to reflect the values provided by these field-collected coverage estimations.

3.2.6. Water Column Measurements

Most water quality measurements are routine and are adequately detailed in the SAP or existing DEQ QAPPs (e.g., DEQ 2005). However, some non-standard analytical measurements are important to QUAL2K operation and will therefore be completed. QUAL2K prompts the user for the stoichiometry (C:N:P ratio) and mass of suspended organic matter ("seston"; living and detrital organic material), so samples for these will be collected and analyzed. See the SAP for details on sample collection procedures.

Real-time measurements (30 min increments) using YSI 6600 EDS sondes will be recorded at 8 sites, for up to 45 continuous days of monitoring. There are currently no DEQ SOPs for using these instruments in long-term deployment. Therefore, data quality objectives for their use are detailed in Section 4.0.

3.2.7 Meteorological Measurements

According to Troxler and Thackston (1975) and Bartholow (1989), it is possible that the meteorological data collected at airports or in towns on the bluffs above the Yellowstone River by NOAA/BOR may not be representative of conditions at the river. Therefore, an independent weather station unit will be installed by DEQ on a small island in the river within the Fort Keogh Agricultural Experiment Station, near Miles City and its airport weather station. If there are significant differences between the on-river and official Miles City NOAA weather data, the differences can be used to help adjust other official data on other parts of the modeling reaches. An adjustment procedure (Raphael, 1962; Bartholow, 1989) will be based on the assumption that the rest of the Yellowstone study area is fairly homogenous with respect to elevation, aspect and land use.

3.2.8 Hydraulic Measurements

Water-quality models are typically no better than required data (i.e., coefficients), especially the travel time used in their mass transport formulation (Hubbard et al., 1982; Wilson et al., 1986; Barnwell et al., 2004). Accurate representation of model hydraulics is necessary to achieve the model output quality desired for this study (see section 7.3, Model Usability). Several approaches have been proposed for estimation of hydraulic properties used in QUAL2K. Paschal and Mueller (1991) and Ning et al. (2000) utilized velocity measurements in a number of modeling reaches to estimate travel time. Kuhn (1991) and Bilhimer et al. (2006) introduced a dye tracer and used florescence measurements to identify travel time between modeled reaches. Park and Lee (2002) used a formulation of Manning's equation and assume prismatic trapezoidal channel geometry. DEQ will directly measure channel geometry, velocity, and associated roughness coefficients at specified sites. Height and width of the lowhead dam near Forsyth will be obtained for calculation of re-aeration and associated hydraulics.

Preliminary calculation of travel time between Forsyth and Glendive has already been completed using a Microsoft VBA program developed by USGS for the Yellowstone River (McCarthy, 2006). The USGS software indicated a travel time of 2.25 days, which is based on the observed flood wave celerity of two storm events and the ratio of this velocity to most probable base flow velocity. McCarthy (2006) is quick to point out that this estimate could easily be off by a factor of two. A dye tracer study is planned to be completed through the USGS in summer 2008 for validation of computed travel time.

4.0 Quality Objectives and Criteria

4.1. Quality Criteria for Benthic Chamber SOD

In spite of its importance to DO dynamics, SOD measurement is not found in Standard Methods (APHA, 1998); however, there is a significant body of literature on the topic (see review by Bowman and Delfino, 1980). Bowman and Delfino (1980) defined 3 criteria for acceptable SOD measurements: (1) consistency; (2) reproducibility; and (3) efficiency. Consistency refers to the ability of the investigator to adhere to the prescribed SOD measuring technique. Consistency will be addressed by adherence to the techniques outlined in the SAP. Reproducibility addresses replicate variability. We will measure SOD in duplicate chambers at each site, with a CV target of ± 20%, which is considered good (Bowman and Delfino, 1980). WOD (used to correct gross SOD) will be measured via the Winkler method in triplicate 300 ml dark bottles incubated at ambient river temperatures. Efficiency refers to the ability to make a sufficient number of measurements over a relatively short time period. We intend to be able to complete each set of SOD measurements within 2-8 hours of initiation, by assuring that the chambers have a chamber volume/sediment surface ratio of 70. If the longer timeframe (i.e. 8 hrs) is needed, these will be run overnight so that SOD measurement will not consume the working hours required to complete other project tasks.

4.2. Quality Criteria for YSI 6600 EDS Sondes Deployed Long-Term

Long Term Deployment of YSI 6600 EDS Sondes. YSI 6600 EDS sondes will be deployed along the river and continuously record data for up to 45 days. Each instrument will be calibrated in the laboratory prior to deployment, and checked again for instrument drift upon retrieval. The Alliance for Coastal Technologies (ACT) is a third-party organization that carries out performance verification studies for these (and other) instruments in rigorous, long-term field deployments around the U.S. (*see reports and organization information at*: http://www.act-us.info/evaulation_reports.php) We have used their "Performance Verification Statement" reports to develop quality criteria for the sondes that we will deploy on the Yellowstone River. These ACT reports discuss, on a probe-type by probe-type basis, the period of time until biofouling begins to interfere with instrument measurements. Days-to-interference from biofouling vary, but typically fall in the range of 14-35 days; in some cases, however, no interference is noted even after 44 days of continuous deployment (ACT, 2007). To assure quality measurements, the YSI sondes will be checked for biofouling in our study at the approximate midpoint of the study, 25-30 days after initial deployment, and cleaned and recalibrated as needed. Data collected to that point will be down loaded to a laptop for safe keeping.

Instrument drift during the deployment period is an equally important issue, and is addressed below, by measurement type.

Dissolved Oxygen. Accurate DO measurement is key to this study, so DEQ has purchased YSI's ROX™ optical DO sensors. These sensors became available from YSI in 2006 and in testing show no significant drift over 1-2 month deployment timeframes during which they were tested (YSI, 2007). This is a great improvement over the drift observed for YSI's polarographic probes (ACT, 2004). The quality criterion for DO concentration data collected over the sampling period using ROX™ optical sensors is that instrument drift will be ≤ 0.2 mg DO/L, using the single-point, water-saturated air technique.

Turbidity. In an ACT test at 7 sites around the country with deployment times ranging from 29-77 days, instrument drift (5 NTU, initial standard calibration) ranged from 0-17%, with a mean drift of 8% (ACT, 2007). The quality criterion for turbidity data collected over the sampling period in our study is that instrument drift, from initial calibration at 11.2 NTU, will be ≤ 10% (YSI has calibration solution of 11.2 NTU which is as close to the 5 NTU as they provide).

Chlorophyll a. In another ACT test at 5 of the 7 sites mentioned above, Chl *a* (using Rhodamine WT as the initial calibration dye) drift during deployment ranged from 31-63% "pre-cleaning" of the probe, and from 0.8 to 18% (mean 7%) "post-cleaning" of the probe (ACT, 2006). (Keeping this probe clean clearly diminishes drift.) The quality criterion for Chl *a* data collected over the sampling period in our study is that instrument drift from calibration (using Rhodarmine WT) will be ≤ 10%, post-cleaning.

4.3. Quality Criteria for Other Field Measurements

Routine Water Quality Measurements. All quality assurance and quality control (QA/QC) requirements followed by DEQ will be instituted for this project. This includes use of standard site visit forms and chain of custody forms for all samples. The QA/QC requirements for water quality samples, flow measurements, etc. are described in detail in DEQ (2005), and are sufficiently covered that repeating them here is not needed.

Dye Tracer Study. The dye tracer study, if initiated, will be carried out by the USGS and all QA/QC procedures developed and implemented by that agency will be followed.

5.0. Assessment and Response Actions

The QA program under which this project operates includes independent checks obtained for sampling and analysis (i.e., laboratory quality assurance processes). The DEQ QA officer may perform audits of field operations and laboratory activities during the course of the project. The QA officer has the authority to stop work on the project if problems affecting data quality that will require extensive effort to resolve are identified.

Any changes to the SAP which may result after the project is initiated will be documented and included as an addendum to the SAP. Project responsibilities for individuals directly involved in the project are shown in Table 5.1 below. The project manager (Suplee) will communicate all significant changes in field protocols or sampling locations to the modeling staff and the DEQ QA officer, as they arise. The likely impacts of these changes on project success will be discussed on a case-by-case basis, and the project adjusted/modified to continue to meet the objectives in this QAPP, as needed.

Table 5.1. Project Personnel Responsibilities.

Name	Organization	Project Responsibilities
Michael Suplee	MT DEQ	Project Management/data collection
Kyle Flynn	MT DEQ	Model Calibration and Validation
Michael Van Liew	MT DEQ	Model Calibration and Validation
Monitoring Staff 1	MT DEQ	Data Collection
Monitoring Staff 2	MT DEQ	Data Collection

6.0 Data Review, Validation and Verification

6.1 Modeling Analyses - Preliminary Data Compilation and Review

Prior to data use, DEQ will compile all information in a usable format for modeling. The necessary QC will be completed to ensure that DEQ monitoring efforts, as well as ancillary data sources used in the modeling effort (i.e., other agencies), are suitable for modeling purposes. USGS, BOR, and NOAA data (streamflow and weather) will be downloaded from each agency's web site and assembled into individual data files. These data will be reviewed by DEQ for quality factors such as completeness, accuracy, precision, comparability, and representativness (DEQ, 2005). The same will be done for DEQ data. The appropriate conversions will be made, and time-series data will be generated in a format suitable for modeling (e.g., QUAL2K operates in SI units and on an hourly time step [Chapra, 2003]). Additional data aggregation is necessary given the steady-state limitations of the modeling framework. Model boundary conditions such as streamflow and meteorology are allowed to vary diurnally in the model, however they are considered constant for the length of the simulation period. Therefore a reach having a three day travel-time is exposed to three days of different hourly meteorological forcings which must be averaged to achieve representative input data (e.g., by taking the three day average of the 7:00-8:00 a.m. air temperature, 8:00-9:00 a.m. temperature, etc.). This procedure is necessary for all meteorological input (air temperature, wind speed, dewpoint, etc.) and any other water quality constituent that needs to be analyzed diurnally (temperature, DO, nutrient speciation, etc.). Point-source water quality data are allowed to vary sinusoidally based on a specified mean, range, and time of maximum. Associated discharges are considered steady-state for the entire simulation period.

7.0 Validation and Verification Methods

7.1 QUAL2K Model Calibration and Validation

Calibration has become increasingly important with the need for valid and defensible models for TMDL development (Donigian and Huber, 1991; Little and Williams, 1992; Wells, 2005; DEQ, 2006b). Model calibration defines the procedures whereby the difference between the predicted and observed values of the model are brought to within an acceptable range by adjustment of uncertain parameters. Ideally, this is an iterative process whereby deficiencies in the initial parameterization are reviewed in a feedback loop to reformulate and refine the calibration. General information related to model calibration criteria and validation considerations can be found in Thomann (1982); James and Burges (1982); Donigian (1982); ASTM (1984); and Wells (2005). For the purpose of this QAPP (and subsequent modeling efforts) two tests will be utilized to define the sufficiency of the model calibration. These are percent bias and the sum of the squared residuals.

Percent Bias. Percent bias is defined as the consistent or systematic deviation of results from the "true" value (Moore and McCape, 1993) and can be a result of a number of deficiencies in modeling. These include: (1) incorrect estimation of model parameters, (2) erroneous observed model input data, (3) deficiencies in model structure or forcing functions, or (4) error of numerical solution methods (Donigian and Huber, 1991). Percent bias is calculated as the difference between an observed (true) and predicted value as shown below.

$$\%B = \frac{(OBS_i - PRED_i)}{OBS_i} \tag{1}$$

Where:

B = Percent Bias
OBS_i = Observed State Variable
SIM_i = Simulated State Variable

Percent bias will be computed for each calibration location (7 different points in the modeling reach) to evaluate the efficiency of the QUAL2K Yellowstone model. Overall percent bias should approach zero.

Sum of Squared Residuals (SSQ). SSQ is a commonly used objective function for water quality model calibration (Little and Williams, 1992; Chapra, 1997). It compares the difference between the modeled and observed ordinates, and uses the squared differences as the measure of fit. Thus a difference of 10 units between the predicted and observed values is one hundred times worse than a difference of 1 unit. Squaring the differences also treats both overestimates and underestimates by the model as undesirable. The equation for calculation of the sum of least squares is shown below (Diskin and Simon, 1977). SSQ will be used as a criterion for overall model evaluation and will be calculated as the summation of all squared residuals for the seven calibration/validation nodes in the model, as well as for the individual nodes.

$$\text{Minimize } Z=\sum_{i=1}^{i_n}[OBS_i - PRED_i]^2 \qquad (2)$$

Where:

Z = Sum of Least Squares

Model Validation. Validation is defined as the comparison of modeled results with independently derived numerical observations from the simulated environment. The same statistical procedures identified in model calibration will be implemented to the validation dataset. Model validation is, in reality, an extension of the calibration process (Reckow, 2003; Wells, 2005) and is often referred to as confirmation. Its purpose is to assure that the calibrated model properly assesses the range of variables and conditions that are expected within the simulation. Although there are several approaches to validating a model, perhaps the most effective procedure is to use only a portion of the available record of observed values for calibration and the other for validation (Chapra, 1997). This type of split-sample calibration-validation is proposed for the Yellowstone River modeling project. Two periods of representative warm-weather conditions will be evaluated; a calibration period in August 2007, and a validation period in September 2007.

7.2 Model Sensitivity

Sensitivity analysis is a technique that can greatly enhance the model calibration process (Chapra, 2003). It guides the modeler to focus the calibration on the most sensitive model parameters and allows the user to judge the relative magnitude of various model parameters on key state variables. Sensitivity is typically expressed as a normalized sensitivity coefficient (Brown and Barnwell, 1987) in which the percent change in the model input parameter is compared to the change in model output. The equation for calculating the sensitivity of a model parameter is shown below:

$$\text{Normalized Sensitivity Coefficient (NSC)} = \frac{\Delta Y_o / Y_o}{\Delta X_I / X_I} \qquad (3)$$

Where:

ΔY_o = Change in the output variable Y_o
ΔX_i = Change in the input variable X_i

Sensitivity analysis is often accomplished using a one-variable-at-a-time perturbation approach (Brown and Barnwell, 1987; Chapra, 1997). A summary of the normalized sensitivity coefficient (NSC) calculated for the one-variable-at-a-time approach will be included as part of the reporting which will include the parameter modified, the range and increment of modification (e.g. ±10%), percent change in the modeling results, and the calculated NSC. The literature will also be consulted to assess modeling efforts similar in nature to ours (e.g, Paschal and Mueller, 1991; Reckow, 1994; Drolc and Koncan, 1999). More complex computational algorithms are also available, such as first-order error analyses and Monte Carlo simulation. An older version of QUAL2K, QUAL2E-UNCAS offers this functionality. Unfortunately, deficiencies in the benthic algae component of this older model make it less useful (Park and Lee, 2002). DEQ will assess the utility of QUAL2E-UNCAS at a later date, although we have no plans to use it for the Yellowstone River project.

Research has shown that sensitivity analyses by themselves are not adequate for characterizing model uncertainty (Melching and Yoon, 1996). Reckow (1994 & 2003) and Chapra (2003) indicated uncertainty analyses should be considered as a routine part of ecological modeling studies. Uncertainty stems from the lack of knowledge regarding model input parameters (Melching and Yoon, 1996) and the processes the model attempts to describe (Beard, 1994). Potential sources of uncertainty in the Yellowstone QUAL2K model have been identified *a priori* by DEQ and include the following:

(1) Estimation of uncertain model parameters
(2) Uncertainty in observed model input data
(3) Deficiencies in model structure and forcing functions
(4) Mathematic errors in numerical methods

Chapra (2003) indicated that modeling uncertainty is best expressed probabilistically. This is even more critical for this effort since numeric nutrient criteria are being developed. A simplified Monte Carlo approach to address uncertainty analysis is proposed for the Yellowstone QUAL2K modeling, in order to account for the combined effect of parameter sensitivity and parameter uncertainty (i.e., a highly sensitive parameter that is fairly certain can have much less effect on the uncertainty of model output than a much less sensitive parameter that is highly uncertain). Probability density functions (PDFs) will be estimated for model parameters using either the uniform, normal, or triangular distributions identified in Chapra (1997) enabling a confidence interval to be calculated from state variable output. This will provide statistical measure of significance on model prediction uncertainty. The Monte Carlo approach is fully decribed in Brown and Barnwell (1987) and Chapra (1997). It is unclear at this time whether DEQ will attempt to use the older version of QUAL2E-UNCAS for this analyses. It is proposed to be done manually at this time (using only a handful of the most sensitive model parameters).

7.3 Model Usability

Acceptance of Modeling Results. QUAL2K has been shown to be a reliable tool for the prediction of water quality when the conditions in the river are similar to those used to calibrate and validate the model (Drolc and Koncan, 1996). The acceptance of the QUAL2K model will be gauged by DEQ in several ways, including: (1) review of the "goodness of fit" indices described previously, (2) comparison of simulated and observed values against *a priori*, user-specified criteria, and (3) model testing. User specific criteria developed by DEQ for the overall Yellowstone River QUAL2K model are shown in Table 7.1.

Table 7.1. Preliminary Calibration and Validation Criteria for Yellowstone QUAL2K model.

State Variable[1]	Criteria in Percent	Unit Criteria
Temperature	±5%	±1 ºC
Dissolved Oxygen	±10%	±0.5 mg/L
Bottom Algae	±20%	mg/m^2
Chlorophyll a	±10%	µg Chl a /L

[1] *Should meet the minimum of percent or unit criteria*

Model validation testing will be completed per Reckow (2003). Three levels of validation testing are available, although only one is proposed. Level 0 testing involves validation of the model over a period that is almost identical to that of the calibration period. Level 1 testing involves the use of a different meteorology for the calibration and validation runs. Level 2 involves the use of both different meteorology and point source loadings. The Level 1 approach is proposed for the Yellowstone River Project given the fact that numeric nutrient criteria are being developed only for a specified flow regime (e.g. low flow). The credibility of these criteria will hinge on the confidence in the model predictions and the understanding of the associated sensitivity and uncertainty in model parameters.

N and P concentrations indicated by the final model as potential criteria will be compared to the N and P concentrations collected during the same period at the comparison site, and to literature values from empirical nutrient-Chl *a* models. If results of all 3 are within an order of magnitude of each other, the results from the model will be considered reasonable due to the site specific nature of the results and documentation of the calibration-validation procedures. We anticipate that concentrations provided by the upstream comparison site will be lower than the output from the model, given that the comparison site has less turbid, colder water. Modeled results that differ from the comparison site/empirical models by more than an order of magnitude will result in a careful re-analysis of the model input parameters. If after the re-evaluation the results from the mechanistic model still differ considerably from the other two approaches, DEQ will indicate this in the final report and provide discussion as to the likely reasons why, and also provide recommendations as to whether or not the model is an appropriate tool for developing numeric nutrient criteria, and why.

8.0 Special Training/Certification

All project participants will have completed a First Responder first-aid course, and also be certified in CPR. All participants who will work on the boat will have completed a U.S. Coast Guard certification course in ‘Boating Skills and Seamanship’. All individuals who will be using the boat on the Yellowstone River will, prior to beginning work on the Yellowstone River, undertake at least one day of boat-use practice at Hauser Reservoir near Helena, MT.

9.0 Documents and Records

Data generated during this project will be stored on field forms, in laboratory reports obtained from the laboratories and in Excel spreadsheets hosted by DEQ shared network servers (backed up on a daily basis). Site Visit/Chain of Custody forms will be properly completed for all samples. Written field notes, field forms (photo log, site information), and digital photos will be processed by DEQ staff following QA/QC procedures to screen for data entry errors. Data provided by the State Lab and the Flathead Lake Biological Station will be in a SIM-compatible format, and will be readied for import into the DEQ's local STORET database and EPA STORET database by the Montana Department of Environmental Quality. Data will be processed with Excel and with Minitab release 14. ArcView version 9 ArcMap will be used for GIS applications. The GPS coordinate system datum will be NAD 1983 State Plane Montana, in decimal degrees, to at least the fourth decimal. All data generated during this project will be available to the public.

A technical report document will describe the findings of the study and will accompany the QUAL2K model developed for the project. The report will summarize the approaches taken (i.e., this QAPP and the SAP), the results of the model calibration & validation, sensitivity analysis and uncertainty analysis. The nitrogen and phosphorus criteria derived from the model will be compared to literature values and to data from the upstream quasi-reference site, and will be thoroughly discussed in the report. Recommendations will be made in the report as to whether or not the mechanistic modeling approach appears to be a reasonable and useful method.

10.0 Schedule for Completion

Assuming full funding is received, equipment purchases will proceed in late 2006 and spring 2007. Coast Guard boating safety and first aid/CPR courses will be completed either in spring or early summer, 2007. The YSI sondes will be deployed at the first reasonable opportunity when the river begins to approach base flow, probably sometime in late July or early August. Synoptic sampling will occur as two separate events, in August and September 2007, preferably about 20-30 days apart. Water quality and other data should be ready for use by November 2007, at which point the model calibration and validation can begin. The model and its associated report should be completed by May 2008.

11.0 Project Budget

Table 11.1. Projected Budget for Purchases Required to Complete Project.

Item*	#	Vendor	Catalog #	Unit Price	Total	Probable $ Source
Infrastructure Purchases						
16' mod-V Jon boat w/ ourlocks, trailer, clean 2-stroke Evinrude 90 hp 0utboard jet.	1	Local		$13,906.00	$13,906.00	Monitoring Section
Large Sea Anchor	1	Local		$100.00	$100.00	Monitoring Section
Garelick Boat Hook 3.5-8 ft	1	Cabelas	IG-016885	$24.99	$24.99	Monitoring Section
Variable 24" boom, 200 lb cap. Winch/Depth Meter(43 lbs)	1	WILDCO	85-E20	$1,449.00	$1,449.00	Monitoring Section
WeatherHawk Weather Station	1	Ben Meadows	6JF-111372	$1,595.00	$1,595.00	Data Management Section
Honda 1500 Watt 240/120/12 V gasoline generator	1	Local	QT-522130	$650.00	$650.00	Monitoring Section
Lab oven to 200° C	1	Fisher	13-254-29	$894.29	$894.29	Monitoring Section
US DH-95 (29 lb) "Clean" Depth Integrating Sampler‡	1	Rickly Hydro	401-055	$2,100.00	$2,100.00	Monitoring Section
‡This is the "clean" model, also suitable for metals and pestacides sampling. The bronze DH-59 model ($725.00) may be adequete for nutrients.				***Total:***	**$20,719.28**	
Project-Specific Purchases (Equipment)						
SOD Chamber	2	A. R. I.	Custom	$950.00	$1,900.00	Monitoring Section
SOD chamber 12 v power supply & submerisible pump	2	Various	Custom	$645.00	$1,290.00	Monitoring Section
Sonde Deployment Apperatus	8	A. R. I.	Custom	$335.00	$2,680.00	Monitoring Section
1/8" Stainless Steel Cable	1200	Rickly Hydro	106-073	$0.30	$360.00	Monitoring Section
Heavy Duty cable cutter	1	Rickly Hydro	106-186	$97.00	$97.00	Monitoring Section
Multi-cavity Swage Tool	1	Rickly Hydro	106-185	$145.00	$145.00	Monitoring Section
Stirrer Plate	1	Fisher	14-493-120S	$160.14	$160.14	Monitoring Section
Teflon Stirrer bar assortment	1	Fisher	14-511-59	$63.32	$63.32	Monitoring Section
50 ml buret for Winkler titration	1	Fisher	03-765	$135.30	$135.30	Monitoring Section
100 ml volumetric pipette	2	Fisher	13-650-2U	$23.98	$47.96	Monitoring Section
Rubber safety pipet filler bulb	1	Fisher	13-681-51	$22.81	$22.81	Monitoring Section
4 X 6 ring stand	1	Fisher	14-670A	$31.15	$31.15	Monitoring Section
Buret clamp	1	Fisher	05-779	$36.79	$36.79	Monitoring Section
Clamp for YSI sonde (3.5 " grip)	1	Fisher	05-769-8	$27.68	$27.68	Monitoring Section
250 ml Erlenmeyer flasks (case of 6)	1	Fisher	10-041-4B	$96.75	$96.75	Monitoring Section
Wheaton 300 ml BOD bottle (case of 24)	1	Fisher	02-926-27	$217.11	$217.11	Monitoring Section
Wheaton 300 ml Dark BOD bottle (case of 20)	1	Fisher	02-926-89	$288.12	$288.12	Monitoring Section
Wheaton Dark BOD bottle caps (case of 50)	1	Fisher	02-926-7	$31.94	$31.94	Monitoring Section
Wheaton 12-place BOD bottle holder rack	2	Fisher	02-663-103	$30.98	$61.96	Monitoring Section
3-place FisherBrand PVC Vacuum manifold w/ 1/4 in barb	1	Fisher	09-753-39A	$595.43	$595.43	Monitoring Section
47 mm Nalge vacuum filter holder	3	Fisher	09-747	$117.79	$353.37	Monitoring Section
120 v high-capacity vacuum pump w/ guages & regulators, 1/4 in	1	Cole-Parmer	C-07061-40	$369.00	$369.00	Monitoring Section
5.25 gallon Nalgene carboy with built in pour spout	2	Fisher	02-923-15C	$71.71	$143.42	Monitoring Section
Gasoline for boat, generator	60			$3.00	$180.00	Monitoring Section
Misc.	1			$1,000.00	$1,000.00	Monitoring Section
Chemical Supplies						
Alkaline Iodide Azide Reagent (500 ml)	1	Fisher	LC10670-1	$31.13	$31.13	Monitoring Section
Manganese Sulfate Solution (500 ml)	1	Fisher	SM20-500	$29.92	$29.92	Monitoring Section
Concentrated Sulfuric Acid (2.5 L)	1	Fisher	A484-212	$68.75	$68.75	Monitoring Section
Starch indicator, 1%, with salicylic acid preservative	1	State Lab				Monitoring Section
0.01 N Sodium thiosulfate solution (1 L)	1	Fisher	LC25000-2	$17.63	$17.63	Monitoring Section
1 L Rhodamine WT 20% dye solution (sold in 1 gallon jugs)	1	Fisher	NC9250029	$305.00	$305.00	Data Management Section
				Total:	***$10,786.68***	
Laboratory Analytical Costs (includes reps and blanks)†						
Water nutrients, Chl a, seston: 14 sites X 2 (Aug, Sep) X 5% replication, + 14 blanks						
TN	44	FLBS		$13.37	$588.28	Standards Section
TP	44	FLBS		$13.37	$588.28	Standards Section
DON	44	FLBS		$14.37	$632.28	Standards Section
NO2/3	44	FLBS		$12.11	$532.84	Standards Section
Ammonia	44	FLBS		$12.44	$547.36	Standards Section
DOP	44	FLBS		$14.37	$632.28	Standards Section
SRP	44	FLBS		$12.00	$528.00	Standards Section
TIC	44	FLBS		$14.68	$645.92	Standards Section
TSS	44	State Lab		$9.20	$404.80	Standards Section
Turbidity	44	State Lab		$6.90	$303.60	Standards Section
Benthic Chl a	154	State Lab		$25.00	$3,850.00	Standards Section
Phytoplankton Chla	44	FLBS		$15.41	$678.04	Standards Section
Phyto AFDW	44	FLBS		$6.00	$264.00	Standards Section
Seston total C	44	FLBS		$6.00	$264.00	Standards Section
Seston total N	44	FLBS		$6.00	$264.00	Standards Section
Seston total P	44	FLBS		$6.00	$264.00	Standards Section
mmonia (Chmbrs): 3 chmbrs/site X 2 (start, finish) X 7 sites X 2 (Aug, Sep), + 7 blanks	91	FLBS		$12.44	$1,132.04	Standards Section
DIC (Chmbrs): 3 chmbrs/site X 2 (start, finish) X 7 sites X 2 (Aug, Sep), + 7 blanks	91	FLBS		$14.68	$1,335.88	Standards Section
				Total:	***$13,455.60***	
				Grand Total:	***$44,961.56***	
† FLBS prices are as-quoted. There may be a 1.41 multiplier added to each cost if the UM overhead costs apply to each analysis.						

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Appendix A

DATA COLLECTED ON THE YELLOWSTONE RIVER, AUG. 2006

Site Name, Elevation (ft)	Distance From Shore	Site Depth	Temperature (°C)	DO (mg/L)	DO (% SAT)	Saturation	Notes
Far West FAS (frm L bank)	1 m	< 50 cm	25	7.2	96	[DO SAT = 7.5 mg/L]	Bottom
(2480 ft)	3 m	75 cm	23.9	8.9	116	[DO SAT = 7.7 mg/L]	Bottom
8/14/2006; 6:35 PM	6 m	75 cm	23.9	9.3	121	[DO SAT = 7.7 mg/L]	Bottom
	9 m	1 m	23.9	9.3	121	[DO SAT = 7.7 mg/L]	Bottom
	12m	1.1 m	24	9.5	123	[DO SAT = 7.7 mg/L]	Bottom
Kinsey Bridge FAS (frm L bank)	2 m	35 cm	26.4	8.2	109	[DO SAT = 7.5 mg/L]	
(2326 ft)	10 m	45 cm	24.5	8.4	111	[DO SAT = 7.6 mg/L]	Bottom
8/15/2006; 12:10 PM	10 m	0 cm	24.5	8.5	112	[DO SAT = 7.6 mg/L]	Surface
	20 m	55 cm	24.2	8.4	109	[DO SAT = 7.7 mg/L]	Bottom
	20 m	0 cm	24.2	8.4	109	[DO SAT = 7.7 mg/L]	Surface
	30 m	55 cm	24.2	7.9	103	[DO SAT = 7.7 mg/L]	Bottom
	30 m	0 cm	24.2	8.4	109	[DO SAT = 7.7 mg/L]	Surface
	40 m	38 cm	24.2	8.3	108	[DO SAT = 7.7 mg/L]	Bottom
	40 m	0 cm	24.2	8.4	109	[DO SAT = 7.7 mg/L]	Surface
	70 m	45 cm	24.2	8.5	110	[DO SAT = 7.7 mg/L]	Bottom
	70 m	0 cm	24.1	8.5	110	[DO SAT = 7.7 mg/L]	Surface
	77 m	90 cm	24	8.0	104	[DO SAT = 7.7 mg/L]	Bottom
	77 m	0 cm	24	8.4	109	[DO SAT = 7.7 mg/L]	Surface
Bonfield FAS (frm L bank)	6 m	40 cm	26.2	8.4	112	[DO SAT = 7.5 mg/L]	Bottom
(2262 ft)	6 m	0 cm	26.2	8.1	108	[DO SAT = 7.5 mg/L]	Surface
8/15/2006; 2:30 PM	30 m	35 cm	24.5	8.2	108	[DO SAT = 7.6 mg/L]	Bottom
	30 m	0 cm	24.5	8.0	105	[DO SAT = 7.6 mg/L]	Surface
	60 m	70 cm	24.2	8.4	109	[DO SAT = 7.7 mg/L]	Bottom
	60 m	0 cm	24.2	8.5	110	[DO SAT = 7.7 mg/L]	Surface
	80 m	80 cm	24.1	8.5	110	[DO SAT = 7.7 mg/L]	Bottom
	80 m	0 cm	24.1	8.5	110	[DO SAT = 7.7 mg/L]	Surface
	95 m	1.0 m	24.1	8.3	108	[DO SAT = 7.7 mg/L]	Bottom
	95 m	0 cm	24.2	8.5	110	[DO SAT = 7.7 mg/L]	Surface
Bonfield FAS (frm R bank, in boat)	25 m	0.5 m	24.7	8.5	112	[DO SAT = 7.6 mg/L]	Surface
	25 m	1.0 m	24.7	8.5	112	[DO SAT = 7.6 mg/L]	Middle
	25m	1.5 m	24.75	8.5	112	[DO SAT = 7.6 mg/L]	Bottom
Roche Jaune FAS (frm R bank)	15 m	25 cm	22	7.4	93	[DO SAT = 8.0 mg/L]	Bottom
(~2300 ft)	27 m	29 cm	22.1	4.5	56	[DO SAT = 8.0 mg/L]	Bottom, in *Cladophora* beds
8/17/2006; 12:05 PM	27 m	0 cm	22.1	7.7	96	[DO SAT = 8.0 mg/L]	Above *Cladophora* beds
	37 m	32 cm	22.2	4.6	58	[DO SAT = 8.0 mg/L]	Bottom, in *Cladophora* beds
	37 m	0 cm	22.2	7.3	91	[DO SAT = 8.0 mg/L]	Above *Cladophora* beds
	50 m	34 cm	22.2	6.4	80	[DO SAT = 8.0 mg/L]	Bottom, in *Cladophora* beds
	50 m	0 cm	22.1	7.5	94	[DO SAT = 8.0 mg/L]	Above *Cladophora* beds
	80 m	39 cm	22.2	7.4	93	[DO SAT = 8.0 mg/L]	Bottom algal mats thin here
	80 m	0 cm	22.2	7.6	95	[DO SAT = 8.0 mg/L]	Surface
	100 m	58 cm	22.1	7.3	91	[DO SAT = 8.0 mg/L]	Bottom
	100 m	0 cm	22.2	7.6	95	[DO SAT = 8.0 mg/L]	Surface
	110 m	75 cm	22.2	7.6	95	[DO SAT = 8.0 mg/L]	Bottom
	110 m	0 cm	22.2	7.6	95	[DO SAT = 8.0 mg/L]	Surface
Fallon Bridge FAS (from L bank)	10 m	39 cm	21.5	7.6	95	[DO SAT = 8.0 mg/L]	Bottom
2204 ft	10 m	0 cm	21.5	7.4	93	[DO SAT = 8.0 mg/L]	Surface
8/17/2006	25 m	63 cm	21.6	7.3	91	[DO SAT = 8.0 mg/L]	Bottom
	25 m	0 cm	21.7	7.4	93	[DO SAT = 8.0 mg/L]	Surface
	35 m	80 cm	21.7	7.2	90	[DO SAT = 8.0 mg/L]	Bottom
	35 m	0 cm	21.7	7.4	93	[DO SAT = 8.0 mg/L]	Surface
	50 m	51 cm	21.7	7.2	90	[DO SAT = 8.0 mg/L]	Bottom
	50 m	0 cm	21.7	7.6	95	[DO SAT = 8.0 mg/L]	Surface
	60 m	1.0 m	21.7	7.5	94	[DO SAT = 8.0 mg/L]	Bottom
	60 m	0 m	21.7	7.7	96	[DO SAT = 8.0 mg/L]	Surface
Intake FAS (from R bank, in boat)	85 m	50 cm	20.1	8.0	94	[DO SAT = 8.5 mg/L]	Just off the bottom
2072 ft	128 m	0 cm	20.1	8.1	101	[DO SAT = 8.5 mg/L]	Midchannel; 90 cm max depth
8/18/2006; wetted width = 234 m	128 m	80 cm	20.1	8.1	101	[DO SAT = 8.5 mg/L]	Midchannel; 90 cm max depth
Site Name, Elevation (ft)	Distance From Shore	Site Depth	Temperature (°C)	DO (mg/L)	DO (% SAT)	Saturation	Notes
Elk Island WMA (frm L bank)	20 m	45 cm	20.8	7.3	88	[DO SAT = 8.3 mg/L]	Bottom
1939 ft	20 m	0 cm	20.7	7.8	94	[DO SAT = 8.3 mg/L]	Surface
8/18/2006; 1:50 pm.	30 m	70 cm	20.6	7.7	93	[DO SAT = 8.3 mg/L]	Bottom
	30 m	0 cm	20.6	7.6	92	[DO SAT = 8.3 mg/L]	Surface
	40 m	80 cm	20.6	7.7	93	[DO SAT = 8.3 mg/L]	Bottom
	40 m	0 cm	20.6	7.7	93	[DO SAT = 8.3 mg/L]	Surface
	50 m	95 cm	20.6	7.7	93	[DO SAT = 8.3 mg/L]	Bottom
	50 m	0 cm	20.6	7.8	94	[DO SAT = 8.3 mg/L]	Surface
	60 m	1.05 m	20.6	7.9	95	[DO SAT = 8.3 mg/L]	Bottom
	60 m	0 m	20.7	7.9	95	[DO SAT = 8.3 mg/L]	Surface
Seven Sisters WMA (from L bank)	10 m	1.0 m	20.5	8.0	96	[DO SAT = 8.3 mg/L]	Bottom
~1900 ft	10 m	0 m	20.4	8.0	96	[DO SAT = 8.3 mg/L]	Surface
8/18/2006							
Richland Park (frm L bank)	1.0 m	1.5 m	21.2	7.3	88	[DO SAT = 8.3 mg/L]	Bottom
1900 ft	1.0 m	0 m	21.2	7.8	94	[DO SAT = 8.3 mg/L]	Surface
8/18/2006; 6:00 pm							

Site Name, Elevation (ft)	Distance From Shore	Site Depth (ft)	Velocity (m sec^{-1})	Notes
Far West FAS (frm L bank)	~1 m	1.8	0.15	Approx. dist. From shore
(2480 ft)	~3 m	2.35	0.23	Near bottom
8/14/2006; 6:35 PM	~6 m	2.55	0.3	Near bottom
	~9 m	2.75	0.1	Near bottom
	~12m	3	0.22	Near bottom
	~15 m	3.01	0.33	Near bottom
	~17 m	3.25	0.31	Near bottom
	~ 17 m	3.25	0.24	Near bottom
Kinsey Bridge FAS (frm L bank)	10 m	1.1	0.1	Near bottom
(2326 ft)	15 m	1.4	0.18	Near bottom
8/15/2006; 12:10 PM	20 m	1.7	0.18	Near bottom
	25 m	1.7	0.21	Near bottom
	30 m	1.51	0.23	Near bottom
	35 m	1.45	0.15	Near bottom
	40 m	1.35	0.11	Near bottom
	45 m	1.05	0.15	Near bottom
	50 m	0.8	0.26	Near bottom
	55 m	1.7	0.22	Near bottom
	60 m	2.2	0.06	Near bottom
	65 m	2.35	0.06	Near bottom
	70 m	1.3	0.12	Near bottom
	75 m	1.45	0.21	Near bottom
	80 m	2.5	0.28	Near bottom
	85 m	2.45	0.29	Near bottom
Bonfield FAS (frm L bank)	15 m	1	0.08	Near bottom
(2262 ft)	30 m	1	0.07	Near bottom
8/15/2006; 2:30 PM	45 m	1.35	0.11	Near bottom
	60 m	1.65	0.19	Near bottom
	70 m	1.9	0.15	Near bottom
	80 m	2.3	0.19	Near bottom
	90 m	2.6	0.11	Near bottom
	100 m	2.6	0.18	Near bottom
	110 m	2.35	0.38	Near bottom
	115 m	3	0.13	Near bottom
Roche Jaune FAS (frm R bank)	15 m	0.4	0.13	Near bottom
(~2300 ft)	25 m	0.35	0.08	Near bottom
8/17/2006; 12:05 PM	35 m	0.35	0	In *Cladophora* bed
	35 m	0	0.07	Above *Cladophora* bed
	45 m	0.6	0.001	In *Cladophora* bed
	45 m	0	0.07	Above *Cladophora* bed
	55 m	0.45	0.02	In *Cladophora* bed
	55 m	0	0.15	Above *Cladophora* bed
	65 m	0.45	0.07	Near bottom
	75 m	0.45	0.17	Near bottom
	85 m	0.5	0.24	Near bottom
	95 m	0.6	0.34	Near bottom
	105 m	1.05	0.24	Near bottom
	115 m	1.9	0.36	Near bottom
	125 m	2.45	0.34	Near bottom
Site Name, Elevation (ft)	Distance From Shore	Site Depth (ft)	Velocity (m sec^{-1})	Notes
Fallon Bridge FAS (from L bank)	15 m	1.1	0.11	Near bottom
2204 ft	20 m	1.4	0.14	Near bottom
8/17/2006	25 m	1.5	0.14	Near bottom
	30 m	1.95	0.15	Near bottom
	35 m	2.25	0.1	Near bottom
	40 m	2.2	0.03	Near bottom
	45 m	2	0.11	Near bottom
	50 m	1.7	0.12	Near bottom
	55 m	1.55	0.14	Near bottom
	60 m	1.05	0.09	Near bottom
	65 m	1	0.09	Near bottom
	70 m	1.9	0.19	Near bottom
	75 m	3.2	0.18	Near bottom
Intake FAS (from R bank, in boat)	85 m	0	0.54	Surface
2072 ft	85 m	2	0.26	Near bottom
8/18/2006; wetted width = 234 m	128 m	0	0.36	Surface
	128 m	2.6	0.19	Near bottom
Elk Island WMA (frm L bank)	10 m	0	0.14	Surface
1939 ft	10 m	0.85	0.35	Bottom
8/18/2006; 1:50 pm.	20 m	0	0.18	Surface
	20 m	1.6	0.16	Bottom
	30 m	0	0.22	Surface
	30 m	2.05	0.14	Bottom
	40 m	0	0.2	Surface
	40 m	2.55	0.11	Bottom
	50 m	0	0.24	Surface
	50 m	2.6	0.12	Bottom
	55 m	0	0.22	Surface
	55 m	2.9	0.14	Bottom
	60 m	0	0.22	Surface
	60 m	3.15	0.17	Bottom
	65 m	0	0.24	Surface
	65 m	3.15	0.15	Bottom
Seven Sisters WMA (from L bank)	10 m	0	0.17	Surface
~1900 ft	10 m	2.25	0.06	Bottom
8/18/2006	15 m	0	0.28	Surface
	15 m	3.55	0.08	Bottom
Richland Park (frm L bank)	0.6 m	0	0.93	Surface
1900 ft	0.6 m	3	0.18	Bottom
8/18/2006; 6:00 pm				

USING A COMPUTER WATER-QUALITY MODEL TO DERIVE NUMERIC NUTRIENT CRITERIA FOR A SEGMENT OF THE YELLOWSTONE RIVER

Quality Assurance Project Plan-Addendum

Prepared for:

MONTANA DEPARTMENT OF ENVIRONMENTAL QUALITY
Water Quality Standards Section, Water Quality Planning Bureau
P.O. Box 200901
Helena, MT 59620-0901

Prepared By:

Michael Suplee, Ph.D.
Project Manager
Water Quality Standards Section

November 6, 2007

Purpose of this Addendum

During the sampling phase of the Yellowstone River project (July 30 -September 23, 2007), several modifications to the original QAPP were necessary due to realities encountered in the field. This addendum documents these changes. Each section number below refers to the corresponding section in the original QAPP. It is recommended that the reader review the original QAPP prior to reading this document. Explanations as to why the change was needed are provided with each.

Section 3.1 Primary Question, Objectives and River Reach Description

Modifications to the site locations, and rationales for the changes, are shown in Table 3.1. A further explanation is necessary for the Kinsey Bridge FAS modification (Table 3.1). It was intended that the new site (Yellowstone River @ river mile 375) would completely replace the Kinsey Bridge FAS site. However, dropping water levels during the August sampling event created river hazards for the boat, and therefore the YSI was moved downstream to the Kinsey Bridge FAS (which could be accessed by road). Thus, the dataset for the Yellowstone River zone downstream of the Tongue River & Miles City WWTP is in two parts; data collected at river mile 375 (through August 22nd), and data collected at the Kinsey Bridge FAS (August 22nd-September 19th).

Table 3.1 Addendum. Modification of site locations.

Originally Proposed Site	Modification	Explanation
Yellowstone River @ Kinsey Bridge FAS	Yellowstone River @ river mile 375, 5.5 miles upstream of Kinsey Bridge	The original intent of the Kinsey Bridge site was to detect potential influences from the Tongue River and Miles City WWTP. The modified site (river mile 375) was deemed better because it was closer to these river influences (new site was 4 miles downstream of WWTP, Kinsey Bridge was 9.5 miles downstream).
Yellowstone River upstream of Powder River & Shirley Main Canal confluences	Yellowstone River just upstream of Powder River confluence	Dirt road access to site upstream of Powder River had potential (during rain) to render the site impassable for boat & trailer. Boat was required to get upstream of Shirle Main Canal confluence. YSI could be retrieved from modified site without the boat, if required.
Yellowstone River 11 miles upstream of Glendive	Yellowstonr River @ Fallon Bridge FAS	Reaching the Yellowstone River 11 miles upstream of Glendive required either boat travel from Glendive or a local launch site. No local launch was found, and boat travel from Glendive was deemed too hazardous due to rocks and the river's shallowness.

Section 3.2.3 Sediment Oxygen Demand Measurements Using Benthic Chambers

Modifications to SOD Measurement. Measurement of SOD in a river system proved to be very different than what I have experienced in lentic systems. The YSI 6600 sonde dissolved oxygen (DO) data from the first set of duplicated SOD incubations (reviewed in the field) revealed that DO, instead of decreasing over time (as expected), increased instead. As DO increased throughout the day in the river, so too did DO in the chambers. Because the chambers have a skirt that penetrated into the river bottom 10 cm, I believe the DO increase was due to a proportion of river water moving through the coarse gravels of the river bed below the chambers' skirt which then mixed (to some unknown degree) with the water in the chambers. To help control for this, subsequent SOD measurements were carried out with one YSI 6600 sonde in the benthic chamber (experiment) and the other YSI 6600 sonde attached to the outside of the chamber in the flowing river water (control). This arrangement precluded duplicate chamber incubations because we only had the two YSI sondes available.

Other Sediment Fluxes Not Measured. Due to time constraints and the influence of dilution from through-gravel flows into the benthic chambers, we deemed it impractical to measure sediment fluxes of DIC, SRP and ammonia.

Section 4.1 Quality Criteria for Benthic Chamber SOD

Because of the issues described above, we only carried out duplicate SOD chambers once. This single duplicated event will have to suffice for comparison with the *a priori* quality criteria proposed for SOD measurements (CV of ± 20% among duplicates).

Section 4.2 Quality Criteria for YSI 6600 EDS Sondes Deployed Long-Term

Biofouling from Drifting Algae. The QAPP addressed means by which biofouling would be managed (periodic cleaning, use of YSI sondes with automatic wiper functions on the probes). However, the type of biofouling anticipated was growth and colonization on the deployer & sondes, and it resulted that this type of growth was fairly light in the Yellowstone River and the wiper mechanisms were clearly capable of keeping the probe faces clean. The major potential biofouling interference came from drifting filamentous algae. Although the deployers were designed to hydro-dynamically shunt drifting algae around the sondes, in some cases drifting algae was so heavy that a build up of snared algae filaments began to smother the probe-end of the YSI sondes. Notes and photographs were taken during each visit as to the overall status of the deployer/sonde units (e.g., "snared drifting algae light, no problems anticipated"; or "heavy algae accumulation, readings may be interfered with"). These notes will be used to help assess data quality (see below).

YSI data were cross-checked in September using a second, calibrated YSI placed near the deployed YSI at the time it was to take a reading (every quarter hour). These cross-checks were made prior to the time the deployed YSI was cleaned. These data will be used to help identify cases where snared drifting algae or other problems were causing instrument interference.

***A posteriori** Protocols for Screening YSI Sonde Data*. Criteria were developed in Section 4.2 of the QAPP to address anticipated factors that could affect the YSI sonde's data quality (instrument drift, biofouling). However, we did not outline a process for segregating data we have high confidence in from data that may be compromised by biofouling or other problems. Therefore, an *a posteriori* process is here defined, and will be applied to each YSI sonde dataset so that high quality data is retained and used in model development.

A. Data logged while a deployed instrument was out of the water for cleaning will be flagged "R" (data rejected, per Modern STORET).

B. When data drift is outside of the criteria established in the QAPP (criteria were established for DO, turbidity, and Chl *a*), we will flag the data back to the previous known point of calibration with "BD" (Beyond allowable Drift).

C. Data from a deployed YSI sonde will be compared to data from the cross-check YSI sonde. In cases where the crosse-check sonde data differ substantially from the deployed-sonde data, the deployed data will be flagged with the letters "DX" (Differs from Cross-Check). Allowable variation between the cross-check and deployed instruments are as follows:

    a. Dissolved Oxygen: 0.5 mg/L (instrument accuracy = 0.2 mg/L, X 2 instruments, plus 0.1 mg DO/L for spatial variation[1])
    b. pH: 0.5 standard units (instrument accuracy = 0.2, X 2 instruments, plus 0.1 unit for spatial variation[1])
    c. Temperature: 0.4°C (instrument accuracy = 0.15°C, X 2 instruments, plus 0.1°C for spatial variation[1])

D. When field notes indicate that a YSI sonde may have been overwhelmed by snared drifting algae, we will:

    a. Review the dataset immediately before and after the cleaning of the unit. Where there is a sharp shift in measured values following a cleaning, the dataset following the cleaning will be considered the preferable one for modeling purposes.

---

[1] YSI cross-checks were taken prior to identifying the exact location of the deployed YSI, in order to prevent any disturbance to the deployed unit. As such, the cross-check unit was usually only within 1-5 meters of the location of the deployed unit due to limited water clarity. This spatial difference is another source of difference between deployed vs. cross-check measurements. Therefore, it is accounted for (as best possible) with this additional allowable variation factor.

i. When sharp change in data values occurs after a cleaning event, an attempt will be made to determine when the interference began. The dataset will be reviewed from the last point of know status (i.e., initial deployment or previous cleaning) up to the cleaning event where the sharp change was noted. Data review will focus on data types that manifest diel patterns (pH, DO). These will be reviewed for (1) sudden, unexplainable change in the magnitude of the daily patterns inconsistent with the pattern immediately proceeding the change, and (2) large, unexplainable scatter of individual data points inconsistent with the overall diel patterns. Data that meet the conditions in (1) and (2) that have no reasonable explanation (e.g., there was a corresponding spike in turbidity that dampened diel DO variation) will be flagged with "I I" (Instrument Interference).

USING A COMPUTER WATER-QUALITY MODEL TO DERIVE NUMERIC NUTRIENT CRITERIA FOR A SEGMENT OF THE YELLOWSTONE RIVER

Sampling and Analysis Plan

Prepared for:

MONTANA DEPARTMENT OF ENVIRONMENTAL QUALITY
Water Quality Standards Section, Water Quality Planning Bureau
P.O. Box 200901
Helena, MT 59620-0901

Approvals

______________________________ _______________
Michael Suplee (WQ Standards Section) Date

______________________________ _______________
Kyle Flynn (Data Management Section) Date

______________________________ _______________
Michael Van Liew (Data Management Section) Date

______________________________ _______________
Bob Bukantis (WQ Standards Section Supervisor) Date

______________________________ _______________
Michael Pipp (Data Management Section Supervisor) Date

______________________________ _______________
Rosie Sada (WQ Monitoring Section Supervisor) Date

______________________________ _______________
Mark Bostrom (QA Officer) Date

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APPENDICES
Appendix A: *Internal DEQ memo supporting dye tracer study on Yellowstone River*
Appendix B: *Project Equipment Checklist*
Appendix C: *Inventory of Project Activities, Listed by Site*
Appendix D1: *Benthica and Algae Cross-section Measurement Field Form*
Appendix D2: *Sediment Oxygen Demand and Solute Flux Field Form*
Appendix D3: *Phytoplankton Productivity Field Form*

1.0 Introduction and Background Information

The intent of this sampling and analysis plan (SAP) is to support the project detailed in the quality assurance project plan (QAPP) of the same name. Please refer to Section 2.0 "Introduction" of the QAPP for details on the background and rationale for the project.

2.0 Objectives and Design of the Investigation

2.1 Primary Question and Objectives

The project outlined in this SAP is designed to answer the following question:

In a segment of the lower Yellowstone River, what are the highest allowable concentrations of nitrogen and phosphorus which will not cause benthic algae to reach nuisance levels and/or dissolved oxygen concentrations to fall below applicable State water quality standards?

Sampling described herein is intended to support the QAPP, and is intended be completed in 2007. The only exception to this is the dye-tracer study, which will probably be undertaken in summer 2008. If the dye-tracer study is completed in 2008, the results from it will be used to further refine the model, which should be developed by that time.

2.2 Overview of What Will be Measured, Where, and How Often

Table 2.1 provides the description, frequency and location of measurements planned for summer 2007. The plan was developed following recommendations outlined in an EPA manual (Mills et al., 1986). EPA's manual provides guidance on designing monitoring plans intended to work in conjunction with the QUAL2E model. Fig. 2.1 shows the targeted reach of the Yellowstone River, and the types of measurements that will be made at various locations throughout. **This information is also provided Appendix C, listed as activities per site, which should be used during field work to track what has been completed.**

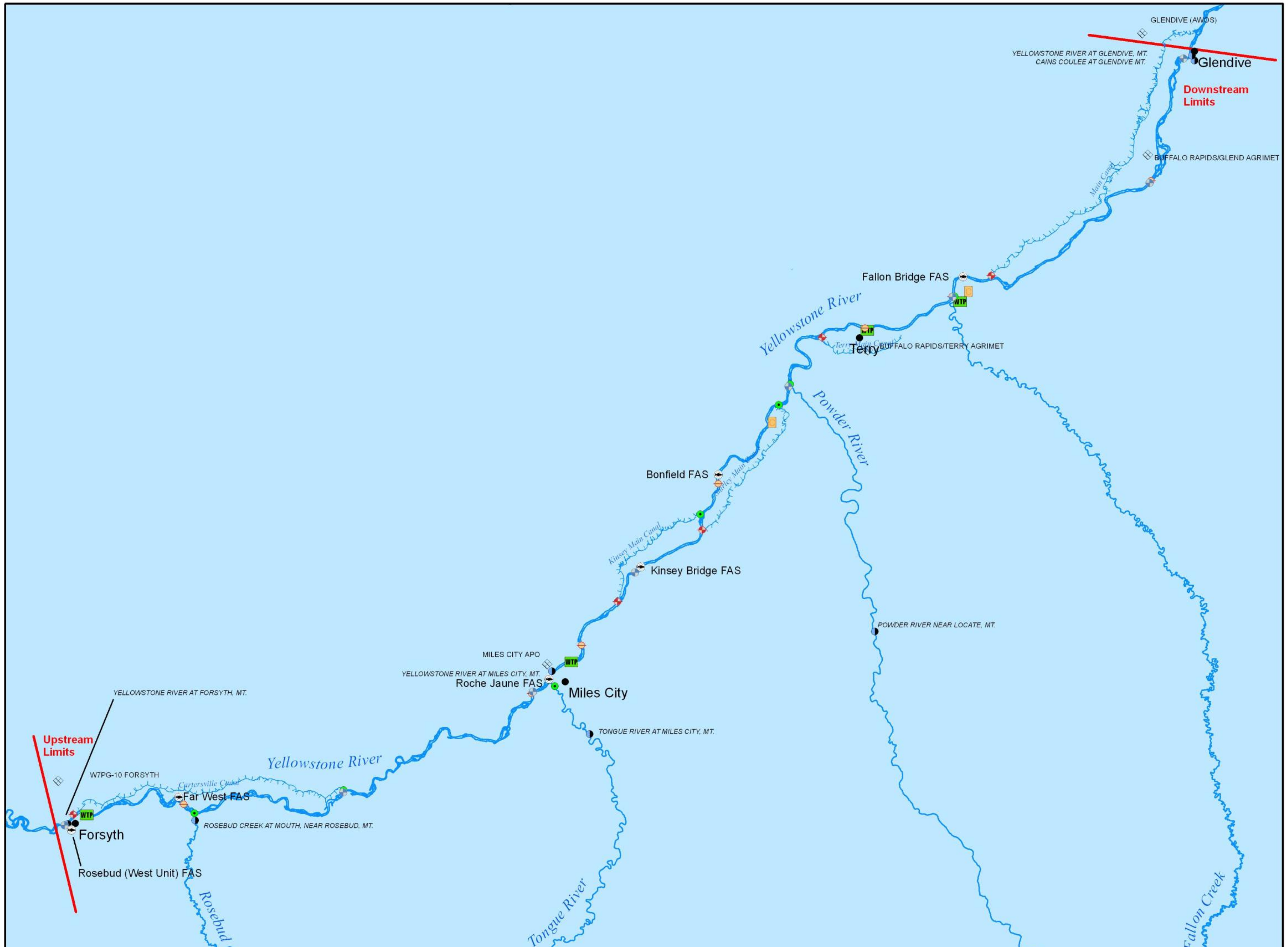

Figure 2.1. Yellowstone River QUAL2K Monitoring Locations

Data Collection Locations

Monitoring Description

- YSI Locations
- Major Withdrawls
- Tributary Inflow/Return Flow
- Permitted CAFO's
- NPDES Permits
- Benthic Measurements/Thermistor Locations
- USGS Gage Sites
- Cities/Towns
- Major Rivers/Streams
- Major Canals
- Study Limits
- Climate Stations
- Fishing Access

N

5 2.5 0 5 10 15 20 Kilometers

Yellowstone River Project Vicinity

Table 2.1 Frequency, Location and Description of Measurements for the Project, Summer 2007.

Measurement (& QUAL2K symbol, if applicable)	Units	How Often Measured	Where Measured
Benthic Chamber Measurements			
Sediment Oxygen Demand (SOD)	$g\ O_2\ m^{-2}\ day^{-1}$	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
Water O_2 Demand to Correct SODs (WOD)	$mg\ O_2\ m^{-3}\ hr^{-1}$	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
Ammmonia flux (J_N)	$g\ N\ m^{-2}\ day^{-1}$	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
Methane flux (J_{CH4}) *Collection and analysis optional*	$g\ O_2\ m^{-2}\ day^{-1}$	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
DIC flux (J_C) — for RQ calculation	$g\ C\ m^{-2}\ day^{-1}$	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
Other Rate Measurements			
Photosynthesis of phytoplankton, via light/dark bottles ($k_{gp[T]}$)	$mg\ C\ m^{-3}\ hr^{-1}$	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
Photosynthesis of bottom-attached *Cladophora*	$mg\ C\ m^{-3}\ hr^{-1}$	Twice (Aug-Sept)	Roche Jaune FAS.
Benthic Measurements			
Benthic algae Chl *a* , and AFDW	$mg\ m^{-2}$	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
% bottom covered by heavy benthic algae at each transect	%	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
% river bottom to which SOD values apply	%	Twice (Aug-Sept)	Far West FAS, u/s of Tongue R. (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive.
Real Time Water Quality Measurements (YSI 6600EDS)			
Dissolved Oxygen (o)	$mg\ O_2/L$	24/7, early Aug to Sept 30	Rosebud (West Unit) FAS, u/s of Carterville Canal return, u/s of Tongue R. (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder R./Shirley Main confluence, upstream of O'Fallon Cr. confluence, 11 miles upstream of Glendive , old Bell St. Bridge in Glendive.
pH	Standard	24/7, early Aug to Sept 30	Rosebud (West Unit) FAS, u/s of Carterville Canal return, u/s of Tongue R. (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder R./Shirley Main confluence, upstream of O'Fallon Cr. confluence, 11 miles upstream of Glendive , old Bell St. Bridge in Glendive.
Temperature	° C	24/7, early Aug to Sept 30	Rosebud (West Unit) FAS, u/s of Carterville Canal return, u/s of Tongue R. (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder R./Shirley Main confluence, upstream of O'Fallon Cr. confluence, 11 miles upstream of Glendive , old Bell St. Bridge in Glendive.
Specific Conductivity	µS/cm	24/7, early Aug to Sept 30	Rosebud (West Unit) FAS, u/s of Carterville Canal return, u/s of Tongue R. (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder R./Shirley Main confluence, upstream of O'Fallon Cr. confluence, 11 miles upstream of Glendive , old Bell St. Bridge in Glendive.
Chl *a* (fluorometric, calibrated to real samples)	µg Chl *a* /L	24/7, early Aug to Sept 30	Rosebud (West Unit) FAS, u/s of Carterville Canal return, u/s of Tongue R. (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder R./Shirley Main confluence, upstream of O'Fallon Cr. confluence, 11 miles upstream of Glendive , old Bell St. Bridge in Glendive.
Turbidity	NTU	24/7, early Aug to Sept 30	Rosebud (West Unit) FAS, u/s of Carterville Canal return, u/s of Tongue R. (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder R./Shirley Main confluence, upstream of O'Fallon Cr. confluence, 11 miles upstream of Glendive , old Bell St. Bridge in Glendive.

Table 2.1, Cont. Frequency, Location and Description of Measurments for the Project, Summer 2007.

Measurement (& QUAL2K symbol, if applicable)	Units	How Often Measured	Where Measured
Water Samples			
Total phosphorus (TP)	μg/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Total nitrogen (TN)	μg/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Nitrate + nitrite (NO_{2+3})	μg/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Total ammonium (NH_4^+)	μg/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Dissolved organic nitrogen (DON)	μg/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Dissolved organic phosphorus (DOP)	μg/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Soluble reactive phosphate (SRP)	μg/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Turbidity	NTU	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
SSC (suspended sediment concentration)	mg/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Total Inorganic Carbon (c_T) Also referred to as Dissolved Inorganic Carbon (DIC)	mole/L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Phytoplankton Chl a (a_p)	μg Chl *a* /L	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Seston C:N:P ratio	dimensionless	Twice (Aug-Sept)	**Yellowstone**: Buffalo Mirage FAS (*Comparison*), Rosebud (West Unit) FAS, u/s of Cartersville Canal return, u/s of Tongue (Ft. Keogh Bridge), Kinsey Bridge FAS, u/s of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Cr. confluence, 11 miles u/s of Glendive, old Bell St. Bridge in Glendive. **Tribs/other**: Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River at mouth, Terry Main Canal return point.
Other Measurements			
PERI-1 diatom population samples		Twice (Aug-Sept)	Buffalo Mirage FAS, Far West FAS, u/s of Tongue (Ft. Keogh Bridge), Pirogue Island State Park, Bonfield FAS, Terry Bridge, upstream of O'Fallon Cr. confluence, 11 miles upstream of Glendive.
Meteorological (wind speed, temp, humidity)	various	Early Aug to Sept 30	Island in Yellowstone R. within the Fort Keogh Ag. Experiment Station.
Water Temperature (Collected hourly via Hobo dataloggers)	° C	Early Aug to Sept 30	Buffalo Mirage FAS, Far West FAS, Pirogue Island State Park, Bonfield FAS, Terry Bridge.
River Width	m	Twice (Aug-Sept)	All sites.
River mean Depth	m	Twice (Aug-Sept)	All sites, except specified benthic sites (See Appendix C).
River velocity	m sec^{-1}	Twice (Aug-Sept)	All other benthic chamber sites.
Flow (DEQ measured)	m^3 sec^{-1}	Twice (Aug-Sept)	**Yellowstone River**: Upstream of Carterville Canal return, upstream of Powder River/Shirley Main Canal confluence, upstream of O'Fallon Creek confluence, 11 miles u/s of Glendive. **Tribs/other**: Cartersville Canal at withdrawal, Cartersville Canal return point, Tongue River nr mouth, Kinsey Main Canal withdrawal, Shirly Main Canal withdrawal, Kinsey Main Canal return point, Shirley Main Canal return point, Powder River nr mouth, Terry Main Canal withdrawal, O'Fallon Cr. near confluence
USGS Discharge	m^3 sec^{-1}	annually - USGS	Yellowstone River at Forsyth, Tongue River at Miles City, Yellowstone River at Miles City, Yellowstone River at Glendive.

3.0 Field Sampling Methods

3.1 Sediment Oxygen Demand, Benthic Chambers, & Solute Fluxes

In Situ Measurement of SOD Using Benthic Chambers, Summer 2007. The chambers will be deployed in pairs at each of the sites indicated in Fig 2.1, Table 2.1 and Appendix C, and will use the YSI 6600EDS sonde and the YSI 85 probe to measure changes in DO and temperature within the chamber.

Chambers will be pressed in to the sediments and then anchored to the bottom using a heavy iron chain wrapped several times around the flexible skirt, so that a good seal between the river bottom and chamber is assured. The chambers will be located on relatively flat sediments in near-shore areas up to 1 meter deep, which can be reached by wading from shore. Based on the near-bottom water velocity measured at the chamber site (using a Marsh-McBirney flow meter, in m sec^{-1}), either the low-flow or high-flow pumps will be selected for attachment to each chamber. After chamber emplacement, within-chamber water will be exchanged with external river water for 2 minutes. The pump will be set on a low-flow setting and its inflow will be disconnected from the chamber so that clean river water can be drawn in and flushed through the chamber. The chamber outflow port will be opened during this time to assure exchange with the external river water. After purging the chamber for 2 minutes, the hose will be reattached and the chamber re-sealed, and the within-chamber water velocity will be adjusted (via the flow-control valve on the pump) to simulate the velocity measured near the river bottom at the site. Periodic checks using the hand-held YSI 85 will be undertaken to monitor chamber DO decline; the incubation will be terminated when a notable decline in DO has occurred.

Changes in the DO of the water within chambers (WOD) will be determined in six 300 ml BOD dark bottles (3 initial, 3 final). The 3 initial bottles will be filled with river water and fixed (Lind 1979) at the time the chambers are emplaced, while the 3 final bottles will be filled and then incubated at ambient river temperatures for the duration of the SOD incubation, then fixed. All 6 will be measured for DO via the Winkler titration method, completing the titration step within 3 days of collection.

The SOD (g O_2 m^{-2} day^{-1}) will be calculated, per Drolc and Koncan (1999), as:

$$SOD = \frac{aV - bV}{S} \quad (1)$$

Where *a* is the slope of the time-DO curve for a chamber with combined sediment & water DO-demand (g O_2 m^{-3} day^{-1}), *b* is the mean slope of the 3 time-DO curves for water in the dark BOD bottles (g O_2 m^{-3} day^{-1}), V is the volume of overlaying water in a chamber interfaced with the sediments (m^3), and S is the area of sediment covered by a chamber (m^2).

Solute Fluxes to be Measured Using the In Situ Benthic Chambers. Ammonia, dissolved inorganic carbon (DIC) and methane fluxes are to be measured in the bethic chambers.

Measurement of methane is, at this writing, optional, as the laboratories identified for the project may not be able to carry out its measurement.

After the chambers have been emplaced, purged and then sealed, water samples for ammonia, methane (*optional*) and DIC will be collected from each chamber at a valve-operated access port using a 60 cc syringe with a luer-lock tip. A second inlet valve will be opened during sample collection to allow an equal volume of river water to enter the chamber and replace that withdrawn during sample extraction. After collection, both valves will be shut. A 2nd set of samples will be collected at the end of the incubation. Concentration change over time for each solute equals the solute's flux.

DIC samples will be carefully filtered using 0.45 µm filters and overflowed in to their sample bottle, without bubbles, until about two sample-bottle volumes have been purged, and then stored without headspace in the bottle on regular ice. **Ammonia** samples will be 0.45 µm filtered, filled to minimize bottle head space, and then frozen on dry ice.

3.2 Other Rate Measurements

Phytoplankton Growth Rates. QUAL2K allows the user to input maximum photosynthesis rates at a given temperature (kgp[T]; Chapra and Pelletier, 2003). Phytoplankton growth rates will be measured using the light-dark bottle technique (Lind, 1979; EPA, 1983; Wetzel and Likens, 1991).

Depth/width integrated water samples (see Section 3.5 on collection of a depth/width integrated water sample) will be used to fill triplicate dark bottles and light bottles. Both light and dark bottles will be incubated *in situ*, under ambient light conditions at or near the water's surface, using the BOD bottle racks, as close to midday as possible. This will provide maximum field-measured photosynthesis rate (EPA, 1983). Incubations will normally be completed within 2-4 hours, at which time the incubation will be terminated by chemical fixation and subsequent DO measured via the Winkler titration method (Wetzel and Likens, 1991; APHA, 1998**). If the titration step of the procedure cannot be completed immediately, place the flocculated & acidified (fixed) samples on ice in the dark for up to a maximum of 3 days.** SEE INSTRUCTIONS ON PAGES 72-77 OF Lind (1979). Samples held in this manner will be warmed to room temperature in the dark prior to completion of the sodium thiosulfate titration step.

Cladophora Influence on DO. Where dense *Cladophora spp*. beds are present, for example the Roche Jaune FAS, DO uptake of *Cladophora* samples will be measured in duplicate 300 ml dark bottles using a YSI model 85 meter. The intent of this measurement is to determine the proportion of DO consumption from the algae relative to the water and sediments, in locations where this alga is obviously a significant nighttime DO sink. DO demand values derived from these measurements can be used to help cross-check outputs from QUAL2K. The calculated rate will be adjusted for the DO change associated with the phytoplankton as measured in the light/dark bottles above.

Blobs of *Cladophora* algae of known mass (squeezed wet weight) will be placed in duplicate dark bottles and the change in DO over time will be measured using a calibrated YSI model 85 meter. The volume occupied by the algae will not exceed about 50% of the bottle. The meter probe will be sealed at the bottle mouth with no air bubbles. Incubations will last 1-2 hrs, or until a 1 mg/L or greater DO drop has been measured. The bottles will be inverted several times prior to taking each DO measurement. Also, the area of river bottom covered by the algal beds will be estimated for a 50 m reach by eye, and the mass of *Cladophora* (squeezed wet weight) m^{-2} in the beds will be measured in 3 locations at the site using the hoop method.

3.3 Other Benthic Measurements

Benthic Algal Chl a, AFDW and Macrophyte DW. Field sampling methods will generally follow, with some exceptions and additions, the DEQ protocols outlined in the draft DEQ Standard Operation Procedure (SOP) manual, "Sample Collection and Laboratory Analysis of Chlorophyll-a", available at: http://www.deq.state.mt.us/wqinfo/monitoring/SOP/sop.asp. Results of the benthic algae sampling will be expressed as chlorophyll *a* (Chl *a*) and AFDW, and the macrophyte biomass as dry weight, in area units (mg m^{-2}).

The longitudinal reach layout described in the DEQ SOP cited above would create unduly long sampling reaches on the Yellowstone River. Instead, we will collect 11 individual samples at equidistant points across transects perpendicular to river flow, at specified sites indicated in Table 2.1 and Appendix C. The hoop, sediment core and template methods will be collected, as appropriate, at equidistant points along each transect.

Algae and macrophytes in hoop samples will be physically separated in the field, and each plant types' Chl *a* and mass will be measured separately in the laboratory. Some transect points will be beyond the reach of a wading person, and instead a boat will be used to collect benthic samples using a Ponar dredge. The boat will be anchored at the sampling point and bottom materials brought up by the Ponar dredge will be subsampled using either the template or sediment core method, as appropriate (the hoop method would not be workable in this situation, and will probably not be applicable in higher velocity areas of the river anyway*). **Use Table 1 of Appendix D1 to record all relevant information for each transect point.***

For diatom community samples, a qualitative composite sample of representative benthic material (PERI-1) from each of the 11transect collection points will be placed in a single 50 cc centrifuge tube, to a volume of 45 ml, and then preserved with formalin (5 ml). Wrap the cap of the tube with Parafilm wax.

Estimate of Algal Growth Cover and Proportion of Applicable Channel SOD. The % river bottom covered by visible algae growth and the % river bottom to which SOD measurements apply will be estimated at the sites specified in Table 2.1 and Appendix C. **During the transect collection of benthic algae**, a record will be made at each of the 11 sampling locales indicating the degree of algae coverage, the substrate class, and the near-bottom water velocity (Table 1, Appendix D1). Based on the information recorded in Table 1, Appendix D1, a final estimate of the % river bottom to which the SOD values apply will be made and recorded in Table 4, Appendix D2.

3.4 Real-Time Water Quality Measurements (YSI 6600EDS)

Data Collected Using the YSI 6600EDS Sondes. Water temperature, pH, DO, specific conductivity, turbidity and Chl *a* concentrations (Table 2.1) will be monitored, for up to six weeks across the study period, using YSI model 6600EDS sondes deployed in the river[1]. The sondes have built-in dataloggers that can be programmed to collect data at pre-defined intervals, and will be set up to take water quality measurements every 30 min or 1 hr. They have a memory capable of storing up to 90 days of logged data, although a YSI representative indicated that 60 days in a more prudent timeframe. YSI's website states that the 6600 sondes have a 75 day battery life at 15 min logging intervals. The sondes will be calibrated in the laboratory according to the manufacturer's instructions (YSI, 2006), and checked again in the field prior to deployment.

Turbidity will be calibrated using the two-point method using 0, 11.2 and 100 NTU standards. **Conductivity** will be calibrated using a 1000 µS/cm standard. The **pH** will be calibrated using the two-point method using pH 7 and 10 standards. **Chl *a*** measurements recorded by the YSI 6600EDS sonde are made using a fluorometric probe, and are relative; that is, to determine the true river Chl *a* values, they must be regressed against laboratory-measured Chl *a* samples, collected separately from the river at the same location[2]. To check instrument drift, the Chl *a* probe will be calibrated in the lab against a 2% Rhodamine WT dye standard (YSI 2006). **DO** will be calibrated, just prior to deployment, in a controlled environment (e.g., hotel room), using the single-point, water-saturated air or air-saturated water method (YSI, 2006).

The sondes are equipped with wipers that periodically clean the sensor surface and these will be activated upon deployment. The sondes may be painted will anti-fouling paint to prevent growth of biofouling aquatic life (YSI, 2006). To minimize problems due to biofouling, the sondes will be checked and cleaned of growth 25-30 days (study midpoint) after the initial deployment. If recalibration is required, as determined from field checks against standard solutions, instrument drift (probe reading vs. standard) will first be recorded prior to re-calibration.

During the sampling runs in mid-August and mid-September, measurements of DO, temperature and specific conductivity will be taken from the boat using a calibrated hand-held YSI (model 85) as near to the deployed sondes as feasible, to cross-check the sondes' data (post deployment). Upon sonde retrieval at the *end* of the project, sonde readings will be compared to laboratory standards for pH, conductivity, etc. to determined instrument drift. DO drift will be checked by using the sonde to measure DO via the single-point, water-saturated air method.

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[1] The YSI placed 11 miles upstream of Glendive is an older model, and because of this it can measure all parameters except turbidity. Also, its DO probe will be the earlier, polaragraphic type, which will be recalibrated after 25-30 days of the initial deployment.

[2] At least 4 Chl *a* water samples will be collected at each long-term sonde deployment site during the study period in order to calibrate the probe measurements. Collection locations and frequency for Chl *a* are shown in Table 2.1; Chl *a* samples procedures for laboratory-analyzed Chl *a* samples are detailed in Section 3.5.

Deployment System for YSI 6600EDS Sondes. During the reconnaissance trip (Aug 2006), we investigated means by which the YSI 6600EDS sondes could be mounted for extended periods in the river (up to 2 months), with some degree of security. The river could not be accessed from the bridge deck of any of the bridges we visited, therefore the sondes will have to be attached to the bridge support columns from the water, or by some other means.

The design shown in Fig. 3.1 was developed for this purpose. The river bottom at all sites in this reach of the Yellowstone River is fairly hard (gravel and sand), and the weighted block of the deployer should not sink in to the bottom any significant distance. The weighted block of the deployer will hold the assembly on the river bottom, and the sonde itself will be maintained in the river flow about 10-15 cm above the bottom. The device should be invisible from shore (except perhaps during very low flows) which should improve security. The brass ID plate embedded on the deployer will say “Water Quality Monitoring Equipment. Property of the State of Montana. If found, please call (406) 444-0831 or (406) 444-5964”. The deployer may be painted with anti-fouling paint to minimize algal and other growth accumulation.

The sonde deployer in Fig. 3.1 will be placed in the river using a boat. A 1/8 inch or smaller stainless steel cable will be looped around the bridge support, or a nearby tree, and then clamped in place with a swage. If no suitable attachment point can be located, an approx. 50 lb block with an eyebolt on it will be placed on the river bottom upstream of the deployer and the sonde deployer will be attached to it. The sonde deployer will then be placed 10-20 m downstream of the bridge support, tree or block, using the boat. The stainless steel cable will allow retrieval of the device as it can be snagged with a grappling hook from the boat. In cases where the device is attached to shoreline trees the cable will be buried, to the extent possible, upon deployment.

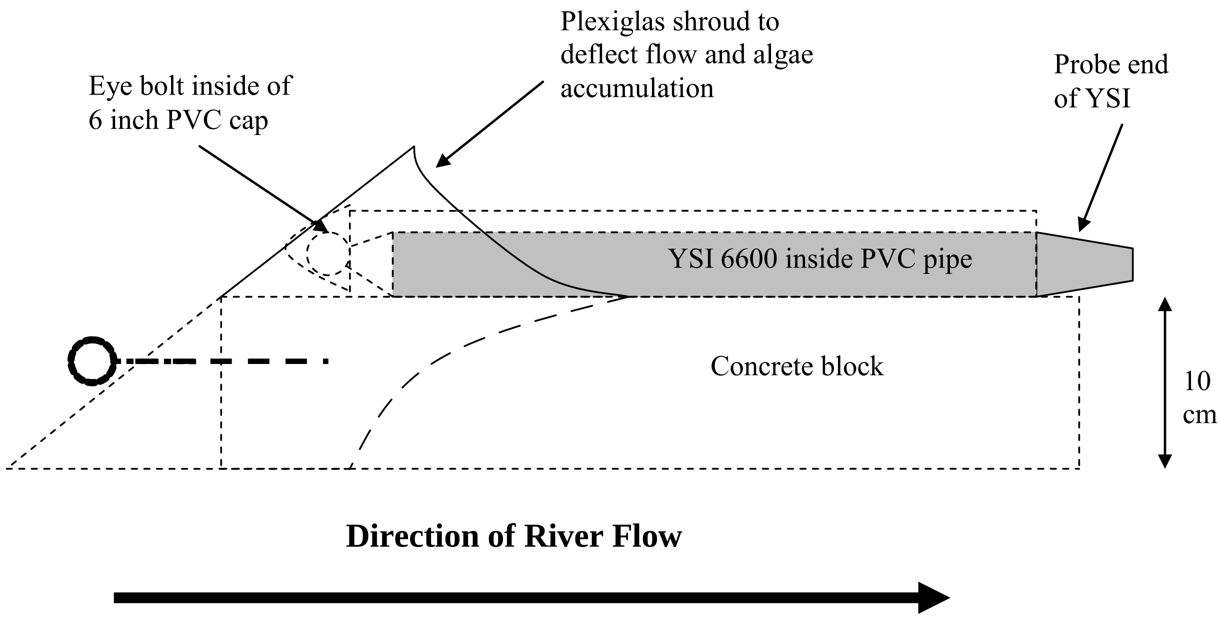

Fig. 3.1. Profile view of the YSI 6600EDS sonde deployment system.

3.5 Water Samples

The majority of nutrient and other water quality parameters shown under the “Water Samples” component of Table 2.1 are routine, and QA/QC guidelines found in DEQ (2005) apply. Because of the width of the Yellowstone River, collecting representative water samples will require depth and width integration techniques rather than simple shore-line grab samples. (Canals will be grab-sampled only.)

A composite water quality sample will be collected concurrent with benthic algae sampling (see Section 3.3) as shown in Figure 3.2 using an equal-width-increment (EWI) sampling technique. At each of the 11 points along a transect, a vertically and horizontally integrated water sample (Wilde et al. 1999) will be collected using a DH48 (wading) or DH95 (boat-mounted) sampler. The 11 samples will be composited into a single carboy and subsamples will be withdrawn for each of water quality parameters of interest (Table 2.1). The plastic carboy will be gently churned (i.e. through light shaking) prior to collection of the samples. For total water-quality measurements (e.g., total P, total N, SSC), phytoplankton Chl *a* and seston, the water in the carboy will be thoroughly shaken and the sub-sample taken immediately.

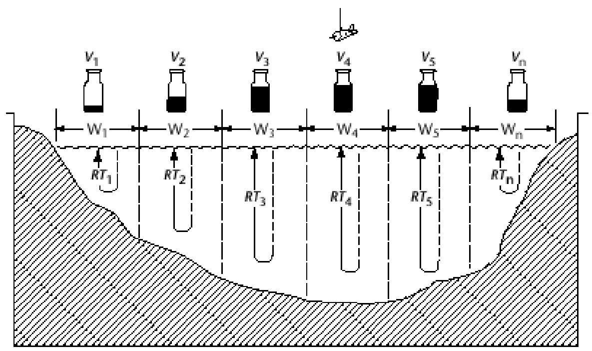

Figure 3.2. Equal Width Increment (EWI) Schematic.

Samples will be preserved and stored per DEQ SOPs (detailed in DEQ's field procedure manual at: G:\WQP\QA_Program\3_Standard Operating Procedures\2-Field Procedures Manual). A copy of the manual will be carried to the field for reference.

Water samples. All dissolved nutrient samples will be field-filtered (0.45 µm). Both total nutrient and soluble nutrient samples will then be frozen immediately on dry ice without additional preservation. (If freezing is not possible, standard DEQ preservation methods with H_2SO_4, etc. will be used. If this scenario arises, submit the preserved nutrient samples to the DPHHS laboratory *only*.**)** **Duplicates will be collected for 5% of all samples. Field/ equipment blanks will be collected at the end of each sampling trip (one in August, one in September).** The DH samplers will be rinsed with 10% HCl and DI water between samplings. Detection limits, appropriate bottle sizes and preservative volumes for each parameter are found in Table 4.0 of DEQ (2005). Sample bottles are as follows:

1. Dissolved nutrients (NO_{2+3}, ammonia, DON, DOP, SRP). 250 ml bottle — 0.45 µm filtered, then on dry ice
2. Total nutrients (TN, TP). 250 ml bottle — dry ice
3. Dissolved Inorganic Carbon. 250 ml bottle — on regular ice
4. Suspended sediment concentration (and Turbidity). 1 L bottle — on regular ice

QUAL2K prompts the user for the stoichiometry (C:N:P ratio) and mass of suspended organic matter ("seston"; living and detrital organic material). Seston will be measured for C, N and P content, dry weight and AFDW. The University of Montana Flathead Lake Biostation is capable of analyzing both CNP samples; the samples will be sent to them after completing the preliminary preparations outlined below. The 1st pair of filters will be analyzed for C & N content using the high temperature induction furnace method (American Society of Agronomy, 1996), and the 2nd pair for total P content using methods outlined in Mulholland and Rosemond (1992).

For CNP samples, dry weight and AFDW will be determined on GF/F filters used to filter known volumes of river water (Section 10300 C; APHA, 1998). (AFDW can be determined from the samples discussed in the next paragraph.) Four samples of known volume will be collected on GF/F filters and stored in 50 cc centrifuge tubes on ice (*not* frozen). **Equal volume of water must be filtered on to each of these filters.** Do not fold. Vacuum on the filters will be kept below 9.0 inches Hg to prevent cell rupture and loss of their contents into the filtrate (Wetzel and

Likens, 1991). At the Water Laboratory in Helena, two of the filters (for C & N analysis) will be placed on a filter holder and rinsed with 10% HCl until they stop fizzing, to remove inorganic carbonates (Niewenhuize et al., 1994). 50 ml tap water will then be pulled through them to remove the acid, and then they will be dried at 105 ^{o}C. The remaining two filters (for P analysis) will be dried directly.

For phytoplankton Chl *a* and AFDW, known volumes of water — which should match the same volume used for the CNP filters— from the shaken carboy will be filtered on to 2 different GF/F filters until a distinct green color is observable on each filter. Vaccum must be held below 9 inches Hg. Filters are folded in half (green side in), put in centrifuge tubes & frozen (dry ice).

3.6 Meteorological Measurements

An independent weather station unit will be installed by DEQ within the Fort Keogh Agricultural Experiment Station, on an island immediately adjacent to the river, near Miles City. The station will measure wind speed and direction, air temperature, and relative humidity and will be used to establish a suitable record for statistical correlation of microclimate, if correction is necessary. The weather station will be of research grade quality, with the following specifications:

1. Air temperature accuracy of ± 0.5 degrees C.
2. Relative humidity accuracy of ± 5 percent.
3. Wind speed accuracy of ± 0.5 m/s.

A Hobo Onset or equivalent station is being purchased by DEQ for the project. Data collected from the DEQ weather station will be compared to the NOAA-FAA data provided by the Miles City Municipal Airport (WBAN 24037, COOP ID 245690) to identify the relative usefulness of data outside of the stream corridor. The sites are approximately one mile away from another.

3.7 Hydrologic Measurements

Discharge will be measured by DEQ at a number of sites during the August and September sampling events to establish the hydrologic balance for the project reach. A calibrated Marsh-McBirney current meter and top-setting wading rod or sounding weight will be used to carry out the velocity-area method (Rantz et al., 1982). Because there will be a combination of wadeable- and boat-accessed measurement points, the procedure for collecting discharge for each type of measurements is shown below.

A. Procedure for Wading Discharge Measurement. See Field Procedures Manual, page 30 (G:\WQP\QA_Program\3_Standard Operating Procedures\2-Field Procedures Manual). In this project, we will determine flow using either (1) the 0.2 and 0.8 measurement points at each subtransect, or (2) the 0.6 depth measurement point, depending on site-specific evaluation of the degree of laminar flow at the site. Sites with even laminar flow *and* limited bottom roughness can be measured using the 0.6 method.

B. Procedure for Boat Discharge Measurements. Visual shoreline references (trees, rocks, bushes, etc.) on each bank, along with a 3X6 ft painted plywood “target”board attached to a post, will be used to assure that measurements are collected along a transect perpendicular to flow. The boat will be positioned to measure depths and velocities by moving to each equidistant point (transect width ÷20) along the transect, and then anchoring in place. A range finder will be used to measure the distance from the boat to the on-shore target board, and a hand-held GPS unit will be used to record the lat and long of the channel midpoint and wetted edges. If the maximum depth in the cross section is less than 3 m and the velocity is low, a rod may be used to measure the depth and support the current meter. For greater depths and velocities, a cable suspension with reel, boat boom, and sounding weight will be used. The Marsh McBirney current meter will be lowered to positions 0.2 and 0.8 of the site depth, and the velocities recorded at each. **If a transect of the Yellowstone River is a combination of boat and wadeable measurements, all points of velocity measurement will be made using the 0.2 and 0.8 method.**

Note: Boat measurements are not recommended where velocities are slower than 0.3 m sec^{-1} or when the boat is subject to the action of wind and waves.

Field staff will observe any rapids along the study reach, as shown on the BLM Yellowstone River Floater's Guide maps, to ascertain if the rapid provides significant re-aeration. For those with significant re-aeration, a water surface slope between upstream and downstream of the rapid will be taken using the laser level, and spot-check DO measurement will be made using the YSI 85 up- and downstream of the rapid.

Digital photographs of the discharge measurement transects will be taken at each site and latitude, longitude and elevation of the sites will be recorded using a hand-held GPS unit. *Canal return points will only be sampled if definable return points can be identified.*

DEQ will use data acquired as part of the USGS's routine monitoring program. USGS has been contacted to ensure that the stations necessary to complete the 2007 field study will be in operation during the 2007 monitoring period (personal communication; P. McCarthy, 2006). USGS data will be acquired in sub daily increments and will serve as the up- and down-stream boundary conditions for the modeling study reach. The following USGS stations will be utilized:

(1) USGS 06295000Yellowstone River at Forsyth, MT (Upstream)

(2) USGS 06309000 Yellowstone River at Miles City, MT

(3) USGS 06308500 Tongue River at Miles City, MT

(4) USGS 06327500 Yellowstone River at Glendive, MT (Downstream)

3.8 Hydraulic Measurements

3.8.1. Dye Tracer Study

See Montana DEQ Field Procedures Manual Section 11.5 Fluorometers (http://www.deq.mt.gov/wqinfo/monitoring/SOP/pdf/11-05.PDF), Hubbard et al. (1982).The following procedures, if undertaken, will be carried out by the USGS. The exact locations of the dye study are in flux because multiple Bureaus within DEQ are cooperating to try to fund the study (see memo, Appendix A). Therefore, the following should be taken as a general plan that will be further refined in the future.

Procedure for Dye Tracer Study A hybrid between the high and low level study approaches proposed by Hubbard et al. (1982) will be completed on the Yellowstone due to the fact that a number of public water supplies are present in the study reach (Forsyth, Miles City and Glendive). The high level approach monitors the dye concentrations at the public water supply intakes to insure that the concentration of dye is less than the maximum levels recommended on the product label while the low level approach fails to do so. It also determines: (1) the travel time of the centroid of dye throughout the modeled reach (using fluorometric techniques) and (2) longitudinal dispersion characteristics of the river by assessing the rate at which the river dilutes the dye . USGS currently maintains two Self-Contained Underwater Fluorescence Apparatus (SCUFA) from Turner Designs in the Helena office. These are proposed for use in the Yellowstone study. Each instrument has a detection limit is 0.04 µg/L for Rhodamine WT dye, provides automatic temperature compensation, and will internally log 11,000 data points at user-defined intervals. SCUFA instrumentation will be leapfrogged in the downstream direction to capture the leading and trailing edges of the dye plume, as well as the peak concentration.

Three unique subreaches will be evaluated as part of the study: (1) Forsyth Bridge (above the diversion) to the Tongue River, (2) Tongue River to the Powder River, and (3) Powder River to the Pacific Railway Bridge in Glendive. Dye will be introduced upstream of Forsyth Bridge at the Myer's Bridge FAS (approximately 47 miles upstream of Forsyth) to ensure complete lateral mixing as well to adequately dilute concentrations prior to arrival at the Forsyth water intake. A single mid channel addition of dye will be used (i.e., 20 liter container of concentrated dye). Length for lateral mixing is calculated as a function of estimated flow velocity (U), channel top width (W), and lateral dispersion coefficient (E_{lat}) for a given flow regime (Hubbard et al., 1982; Chapra, 1997). Lateral mixing distance for the Yellowstone at this site is approximately 40 km

$$L_m = 0.1 \frac{UB^2}{E_{lat}} \tag{2}$$

Rhodamine WT is the preferred dye for tracer studies (Hubbard et al., 1982; Mills et al., 1986; USGS SMIC, 2005), and has been selected for use in this study. Criteria recommended by the Environmental Protection Agency Federal Register Vol. 63, No. 40, National Sanitation Foundation (NSF) Standard 60, and USGS Water Resources Division (Wilson et al., 1986; USGS SMIC, 2005) are 10 µg/L Rhodamine WT for the source water entering a public water supply (prior to treatment and distribution) and 0.1 µg/L in the distribution system. Montana does not have a water quality standard for Rhodamine WT. For this study DEQ will maintain the concentration of Rhodamine WT at or below the levels recommended by the EPA and label instructions. In order to determine the volume of dye necessary to satisfy an adequate endpoint concentration at Glendive, the concentrations at each of the water intakes (Forsyth and Miles City) needs to be determined first to ensure the intakes at are protected, and then that the downstream detection limit is satisfied.

A desired endpoint of 0.25 µg/L near Glendive (well above the SCUFA detection limit of 0.04 µg/L) was identified by DEQ to ensure that photodegradation, biodegradation, adsorption to sediments, or uptake by plants do not cause concentrations to fall below the analytical limits. Smart and Laidlaw (1977) and Turner et al. (1991) indicate that Rhodamine WT is conservative in studies of one week of duration or less (98-100% recovery). Other studies (e.g., Hubbard et al., 1982) indicate significant loss. A margin of safety was therefore selected to ensure detection while still maintaining concentrations well below the EPA, NSF and USGS criteria of 10 µg/L at public water supply intakes. The necessary volume of a 20% Rhodamine WT dye solution required to satisfy these requirements is calculated as follows (Hubbard et al., 1982):

$$V = 2 \times 10^{-3} \left[\frac{Q_m L}{U} \right]^{0.93} C_p \tag{3}$$

Where: (V) is the volume of dye in liters, (Q_m) is the expected or actual discharge in the reach in cubic meters per second, (L) is the distance from injection to sampling point in km, (U) is the mean velocity in m/s, and (C_p) is the peak concentration desired in µg/L. Based on these calculations, a 20 L injection of Rhodamine WT 20% solution near the Myer's FAS (upstream of Forsyth) will achieve the 0.25 µg/L target at Glendive for average August-September flows. These values, of course, will need to be "fine-tuned" as real-time flow data near the time of the field study are compiled. Estimated dye concentrations at critical points in the study reach (e.g. water intakes) are shown in Table 3.2. They are nearly a factor of 10 below the EPA, NSF, and USGS recommended values.

Table 3.2. Estimated Dye Concentrations at Specific Locations along on the Yellowstone River (August-Sept flow regime)

Hydraulic Reach	Upstream Point	Downstream Point	DS Reach Stationing (km) [1]	Mean Q (m^3/s)	Mean U (m/s)	Concentration (µg/L)	
BOUNDARY	---	Myer's FAS	0	205	---	---	
NA-MIXING	Myer's FAS	USGS @ Forsyth	75.5	205	0.91	1.15	
YLW-01	USGS @ Forsyth	USGS @ Miles City	128.7	230	0.91	0.65	
YLW-02	US Tongue River	US Powder River	201.5	235	0.89	0.40	*Estimate
YLW-03	US Powder River	Glendive RR Bridge	310.7	240	0.89	0.25	
	Total Dye Rhodamine WT (20% solution)			20 liters			

[1] *McCarthy (2006); DEQ (2006).*

[2] *Unknown Reach Length*

3.8.2 Channel Dimensions and Related Measurements

Procedure for Velocity and Depth Rating Curve Development. Depth and velocity measurements (in the form of a rating curve) are used to calculate travel time as well as wetted channel dimensions in QUAL2K. DEQ will measure these values in the field to provide model input as well as validation information. At each of the mainstem sites where discharge will be measured (Section 3.7), mean cross-sectional velocity, mean depth, and wetted river width data will already be available. At other specified sites (Appendix C; benthic/rate sites), mean river depth and wetted width will be measured to define the overall hydraulics of the system. Mean river depth will be determined from 11 measurements along each transect site. Wetted width will be measured using a laser range finder. In addition, field measurements from USGS at USGS-gauged sites will be used. Digital photographs of the river at each physical characteristic measurement location will be taken in the up- and down-stream directions. Latitude, longitude and elevation of the sites will be recorded using a hand-held GPS.

One low-head dam is present within the study reach (Fischer, 1999; USFWS, 2002). The Cartersville Diversion Dam (also called Forsyth Diversion Dam) is located near Forsyth and was constructed during the early 1930s utilizing riprap capped with concrete. The dam is over 800 feet in length and spans the entire width of the channel. In order to adequately define velocity and flow depth resulting from this structure, as well as to compute reaeration (Chapra, 2003), height of the diversion dam is a necessary input to QUAL2K for weir computations.

Two measurements will be made at the Forsyth low-head dam (if possible) to identify the average height of the dam: one at the left bank, and one at the right. “As built” drawings will also be consulted. The mean of the left and right banks will be used to determine the average weir height. A metric fiberglass survey rod (or engineers tape) will used to record this measurement. Digital photographs will be taken of the structure and the latitude, longitude and elevation will be recorded using a hand-held GPS. Width will be measured using a laser range finder and will be compared to values measured from aerial photography.

3.9 Boat Usage

Equipment. Because of the river’s depth, a boat will be used for collecting a large number of the measurements outlined above. We will use a 16 ft Jon boat (mod-V hull with tunnel) equipped with an outboard jet. The Jon boat provides a relatively stable platform from which to work, e.g., operating a small winch/boom apparatus to collect benthic samples or measure velocity. Additional equipment for the boat are:

1. Coast Guard approved life preserver for each occupant
2. Two type-IV throwable floatation device
3. Horizontally-mounted fire extinguisher (for fires type A, B and C)
4. Airhorn
5. Flares (visual distress signal)
6. Oars
7. Bailing device, including a bilge pump
8. Winch/boom apparatus for benthic grabs, velocity measurements, etc.
9. Claw-type anchor and mushroom-type anchor with chain and rope
10. Large cleat on bow to secure anchor line
11. Electric anchor cable winch

Boat Operation and Safety Training. ***All field staff in the boat will be required to wear their life preserver at all times.*** All project participants who will operate the boat have completed a boating safety class offered by the U.S. Coast Guard Auxiliary. A copy of the Coast Guard textbook from the course (USCG 2006) will be carried to the field and kept in the boat. Montana boating regulations available at: http://fwp.mt.gov/fishing/regulations/boatrestrictions.html will be reviewed by all project participants who will be in the boat. Participants who will operate the boat will familiarize themselves with the boat & motor operation on a lake or reservoir prior to using the boat on the Yellowstone River.

Intended Usage of Boat. The boat will be launched as close as is reasonably possible to each sampling site. The boat will be anchored in place at points where measurements (velocity, water samples, etc.) are made along transects. One individual on the boat will be assigned as a lookout for other boats on the river at times when the boat is anchored in the river.

4.0 Sample Handling Procedures

Sample storage times are shown in Table 4.0 of the DEQ WQPB QAPP (DEQ 2005). Standard DEQ Water Quality Planning Bureau site visit/chain of custody forms will be used to document and track all samples collected in the project. Samples will be delivered to the Department of Public Health and Human Services Environmental Laboratory (DPHHS laboratory) in Helena, or shipped frozen (or delivered) to the UM Flathead Lake Biological Station. The following samples will be delivered to the Flathead Lake Biological Station for analysis: DIC, dissolved methane (if collected), total N, total P, NO_{2+3}, total NH_3, DON, DOP, SRP, seston CN samples, seston P samples, phytoplankton Chl *a* & AFDW samples. The DPHHS laboratory will receive benthic Chl a samples, and SSC and turbidity samples.

5.0 Laboratory Analytical Measurements

The detection limits of the analyses undertaken by the DPHHS laboratory are detailed in Table 4.0 of the DEQ WQPB QAPP (DEQ 2005). For nutrients and other water quality parameters listed in Table 2.1 of this SAP to be analyzed by the Flathead Lake Biological Station, method detection limits are as shown in Table 5.1, below. Table 5.2 (below) shows the performance characteristics of measurements made by the YSI 6600EDS sondes (YSI, 2006).

Table 5.1. Major Plant Nutrients and their Detection Limits, As Analyzed by the Flathead Lake Biological Station.

Parameter	Units	Analytical Method	Sample Prep/holding time	Detection Limit	Method Reference #
Dissolved inorganic carbon (DIC)	mg/L	Phosphoric acid injection	Filtered asap, stored at 4 degC/ 14 days	0.04 mg/L	13
Total nitrogen (TN)	µg/L	Persulfate digestion	Frozen asap/ 6 months	20 µg/L	7, 8, 9, 10
Nitrate + nitrite (NO_{2+3})	µg/L	Cadmium reduction	Filtered and frozen asap/ 6 months	0.6 µg/L	3, 4
Dissolved organic nitrogen (DON)	µg/L	Persulfate digestion	Frozen asap/ 6 months	17 µg/L	
Total ammonia (NH_3)	µg/L	Automated phenate method	Filtered, frozen asap/ 6 months	5 µg/L*	5, 6
Total phosphorus (TP)	µg/L	Sulfuric acid & persulfate digestion followed by ascorbic acid	Frozen asap/ 6 months	0.4 µg/L	1, 2
Dissolved organic phosphorus (DOP)	µg/L	Sulfuric acid & persulfate digestion followed by ascorbic acid	Frozen asap/ 6 months	3 µg/L	
Soluble reactive phosphate (SRP)	µg/L	Direct ascorbic acid	Filtered, frozen asap/ 6 months	0.3 µg/L	1, 2

* As a result of background ammonia on field filter blanks, the practical detection limit may be approximately 20 µg/L.
All the automated methods are done on a continuous flow instrument (Technicon™ Autoanalyzer™ II)

Method References:

1) Standard Methods for the Examination of Water and Wastewater, 17th Edition (1989), p.4-177 Method 4500-P E. and p. 4-170 Method 4500-P B.
2) Technicon Autoanalyzer II Industrial Method No. 155-71W Ortho Phosphate in Water and Seawater, adapted (Ted Walsh, U. of Hawaii, Personal communication, 1988).
3) Standard Methods for the Examination of Water and Wastewater, 16th Edition (1989). p4-135, Method 4500-NO3- E.
4) Technicon™ Autoanalyzer™ II Industrial Method No. 158-71W/B, revised Aug 1979.
5) Standard Methods for the Examination of Water and Wastewater, 17th Edition (1989), Method 4500 H. pg 4-111 - 4-128.
6) Technicon™ Autoanalyzer™ II Industrial Method No. 154-W/B, revised January 1978, Ammonia in Water and Seawater.
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9) Standard Methods for the Examination of Water and Wastewater, 16th Edition(1985), p.400, Method 418 F.
10) Technicon Autoanalyzer II Industrial Method No. 158-71W/B, revised Aug 1979.
11) Standard Methods for the Examination of Water and Wastewater, 20th Edition (1998). P5-20 or 5-24, Method 5310 B or D.
12) Menzel, D. W. and R. F. Vaccaro. 1964. The measurement of dissolved organic and particulate carbon in seawater. Limnology and Oceanography 9:138-142.
13) Operating Procedures Manual for Oceanography International Corporation Total Carbon Analyzer.

Table 5.2. Performance Characteristics of the YSI 6600EDS Sonde

Parameter	Resolution	Accuracy	Range
Water Temperature	0.01 °C	± 0.15 °C	-5 to 45 °C
pH	0.01 units	± 0.2 units	0 to 14 units
DO (mg/L)	0.01 mg/L	± 0.2 mg/L	0 to 50 mg/L
DO (% saturation)	0.1% air sat.	± 2%	0 to 500% air sat.
Specific Conductance	0.001 mS/cm	± 0.5% of reading	0 to 100 mS/cm
Chlorophyll a	0.1 µg Chl *a* /L	none given*	0 to 400 µg Chl *a* /L
Turbidity	0.1 NTU	2 NTU	0 to 1000 NTU
Battery Life	90 days at 20 °C, 15 min logging intervals w turbidity and Chl a on.		

**In vivo* measurements will only be as accurate as the laboratory samples against which they are calibrated.

6.0 Quality Assurance and Quality Control Requirements

Quality assurance and quality control (QA/QC) requirements for some of the more unique procedures in the SAP (e.g., benthic SOD chambers, long-term YSI sonde deployment) have been outlined in the project QAPP. All other standard QA/QC requirements followed by DEQ (DEQ 2005) will be instituted for this project.

7.0 Data Analysis, Record Keeping, and Reporting Requirements

Data logged in the YSI 6600EDS sondes will be downloaded to a DEQ computer via the EcoWatch for Windows program provided by YSI. Data generated during this project will be stored on field forms, in laboratory reports obtained from the laboratories and in Excel spreadsheets hosted by DEQ shared network servers (backed up on a daily basis). Site Visit/Chain of Custody forms will be properly completed for all samples. Written field notes, field forms (photo log, site information), and digital photos will be processed by DEQ staff following QA/QC procedures to screen for data entry errors. Data provided by the DPHHS laboratory and the Flathead Lake Biological Station will be in a SIM-compatible format, and will be readied for import into the DEQ's local STORET database and EPA STORET database by the Montana Department of Environmental Quality. Data will be processed with Excel and with Minitab release 14, Systat version 10 or StatMost for Windows statistics utilities. ArcView version 9 ArcMap will be used for GIS applications. The GPS coordinate system datum will be NAD 1983 State Plane Montana, in decimal degrees, to at least the third decimal (thousandths). All data generated during this project will be available to the public.

8.0 Schedule for Completion

Equipment purchases have proceeded since late 2006. Boating safety and first aide courses were completed by project participants in spring 2007.

Five major trips are scheduled for completing this SAP:

1) Deployment of YSI sondes in late July/early August 2007 (approximately 8 days)

2) Sampling run No. 1 (calibration dataset), 3rd and 4th full weeks of August, 2007 (approximately 10-12 day trip)

3) Check and clean YSI sondes of biofouling, end Aug/start Sept, 2007 (approximately 5 days)

4) Sampling run No. 2 (validation dataset) 3rd and 4th full weeks of September, 2007 (approximately 10-12 days).

5) Retrieval of YSI sondes, late September/early October 2007 (approximately 5 days).

The model and its associated report should be completed by May 2008. Further refinement of the model based on the dye study will be completed after USGS provides the dye study results.

9.0 Project Team and Responsibilities

This project is intended to be carried out by staff of the Montana Department of Environmental Quality. Personnel directly involved in this project are presented in Table 9-1.

Table 9.1. Project Personnel Responsibilities.

Name	Organization	Project Responsibilities
Michael Suplee	MT DEQ	Project Management/data collection
Kyle Flynn	MT DEQ	Model Calibration and Validation
Michael Van Liew	MT DEQ	Model Calibration and Validation
Monitoring Staff 1	MT DEQ	Data Collection
Monitoring Staff 2	MT DEQ	Data Collection

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Appendix A

Memo

To: Jon Dilliard, Bonnie Lovelace, George Mathieus, Todd Teegarden

From: Michael Pipp, Bob Bukantis, Mike Suplee, Kyle Flynn, and Jim Stimson

CC: Joe Meek, Mark Smith, Kate Miller

Date: April 19, 2007

Re: Potential Cooperative Project Opportunity with the USGS

Proposal Overview

The U. S. Geological Survey (USGS) is interested in conducting a dye-tracer study on the Yellowstone River. A study of this kind would be extremely helpful to several DEQ programs and projects. To undertake the study the USGS needs cooperators to help with funding. The USGS would conduct the study and would participate in funding the effort using their own matching funds. They would match funding from other cooperators on a 40:60 ratio. The purpose of this memo is to explain how the proposed dye-tracer study provides critical information for several DEQ programs and to solicit input on possible funding sources from DEQ Bureau Chiefs and Section Managers. An estimate of the cost for the study is being developed at this time through discussions with the USGS and DEQ staff listed above. As soon as estimates are available Michael Pipp and Bob Bukantis will request a brief meeting with you all to discuss funding possibilities.

Dye-Tracer Study and Numeric Nutrient Criteria Development

The Water Quality Standards Section is developing numeric nutrient criteria for all surface waters of the state. Starting in summer 2007, The Section is planning to work in the lower Yellowstone River in order to develop criteria for the lower river. The Section is planning to use a water quality model (QUAL2K) to answer the following question:

> *In a segment of the lower Yellowstone River, what are the highest allowable concentrations of nitrogen and phosphorus which will not cause benthic algae to reach nuisance levels and/or dissolved oxygen concentrations to drop below applicable State water quality standards?*

The highest input of nitrogen and phosphorus concentrations that do not cause nuisance algae growth and/or exceedences of the DO standards under low-flow conditions may be used as the numeric nutrient criteria for this river segment. Our basic assumption is that the underlying mechanistic foundation of the model is sound, but direct measurement of key parameters driving the model will increase the model's accuracy.

Dye-Tracer Study and Nutrient Water Quality Model

Water-quality models are typically no better than the travel time used in their mass transport formulation and several approaches have been proposed in the literature for estimation of reach travel time. The most accurate of these is through dye-tracer and florescence studies, of which MDEQ is proposing for the Yellowstone River. Accurate travel time is crucial in calculating water temperature within the model (i.e. water temperature is extremely sensitive in DO modeling), for correcting temperature dependent rate coefficients, and completing calculations for which a particular segment is influenced by those rate coefficients. Several unique subreaches are proposed as part of the dye-tracer study for the modeling effort. These include: (1) Forsyth Bridge to the Tongue River, (2) the Tongue River to the Powder River, and (3) the Powder River to the Pacific Railway Bride in Glendive. It is believed that the proposed dye-tracer study could be extended upstream (to Billings for example) to characterize travel time/dispersion for public water supply/drinking water purposes.

Dye-Tracer Study and Surface Water Public Water Supplies

In 2004 the Source Water Protection Program wrote a grant to EPA to help fund a USGS study that used flood wave velocity to estimate surface water time of travel along a portion of the Yellowstone River. It was hoped that the flood wave study could be used as a “quick and easy” method to estimate time of travel for the purpose of assessing the potential impact of contaminant spills or releases on public water supplies along the Yellowstone. However, the flood wave study’s conclusions and results can only be validated with the aid of a dye-tracer study as described above. In addition to validating the flood wave study, time of travel and dispersion data generated by the proposed dye study would give the Public Water Supply and the Source Water Protection programs additional information to help assess the threat of potential contaminant spills or releases on the river. The information from the proposed study can be used to better estimate: 1) how long it will take a contaminant plume to reach a public water supply from a give release site, 2) how long it will take for the plume to pass by the water supply’s intake, and 3) the peak concentration that can be expected in the vicinity of the surface water intake. Funding the proposed dye-tracer study would help multiple programs within DEQ.

Appendix B Equipment List

ITEMS FOR WATER SAMPLING

- Field Sheets, Write in Rain Level Survey Book, Labels, Clip Board, Sharpie Pens/pencils
- Plastic Carboys (2)
- 0.45 µm filter catridges
- 60 cc syringes (clean; 25)
- Sample Containers (includes duplicates and extra bottles, and bottles for chamber fluxes)
    - Water sample bottles (develop detailed list)
    - Centrifuge tubes or petri dishes for Chl *a* (benthic and phytoplankton) and CNP samples
    - 1 gallon size ziplock bags
- Preservatives
    - H_2SO_4
    - Formalin (100 ml)
- 47 mm GF/F filters and tweezers
- 47 mm filter apperatus
- Hand vacuum pump
- Centrifuge tubes
- Aluminum foil
- Ice Chests (3) and Ice
- Dry ice
- Portable 12 v/120 v freezer
- DH 48 and associated bottle
- DH 95 boat or bridge mounted sampler, and associated bottle
- Large HDPE plastic jar as an acid bath for DH48 bottles

ITEMS FOR DO WINKLER TITRATIONS

- Manganese sulfate solution
- Alkalie-Azide reagent
- Standard sodium thiosulfate titrant
- Starch indicator solution (eye dropper)
- 10% HCl solution
- DI water
- Concentrated H_2SO_4
- Carboy for waste chemicals (1)
- 100 ml volumetric pipette (2) and bulb
- 50 ml burette with stop-cock
- Ring stand and burette clamp
- Stirrer plate
- 250 Erlenmeyer flasks (4) and stirrer rods
- Ice chest and ice
- 300 ml dark BOD bottles (9) and holder caps
- 300 ml light BOD bottles (9) and holder caps
- Rack to hold BOD bottles (2)

- Lind (1979) book

ITEMS FOR REAL-TIME WATER QUALITY
- Calibrated YSI 6600ED sondes (8)
    - Calibration Solutions (pH)
    - Spare Batteries
    - Clamp for YSI sonde (3.5 " grip)
- YSI deployment apparatus (8)
- SS cable (miminum of total 1,250 ft; can be in roles of 150 or 200 ft)
- Swage tool and swage locks
- Cable cutter
- Shovel
- Heavy blocks with eyebolt for non-bridge deployment
- Laptop with Ecowatch
- Laptop-to-sonde cable
- 650 hand-held YSI with barometer
- 650-to-sonde cable
- Boat hook with special hook on end to catch cables
- HOBO temperature loggers (6)
- Fence posts or bricks to hold data temp loggers
- Zip ties
- Small sledge hammer

ITEMS FOR SAMPLING FROM BOAT/FLOW

- Top Setting Rod (2)
- Marsh McBirney Velocity Meter (2)-lab calibrated (set to m sec^{-1})
- Laser-level, tripod and batteries
- Bushnel Laser Range Finder
- Grey painted plywood "target" (4'` X 6') and fence posts (2)
- Fiberglass survey rod
- Long fiberglass tape (m)
- GPS Unit and batteries
- Hip waders and boots
- Marsh McBirney boat/bridge mountable velocity device
- Ponar grab

ITEMS FOR SOD MEASUREMENT
- Benthic chambers (2)
- 500 GPH pumps (2) and 1800 GPH pumps (2)
- 100 ft special water-tight connector extension cords (2)
- Honda generator
- Safety breaker (110 v)
- Length of heavy chain (2)
- Snorkel and mask, bathing suite and Tevas
- 300 ml dark BOD bottles (6) and caps
- Ice chest (1)

- YSI 85 (2)
- 60 cc syringe (8)-need 10% HCl and DI water rinse between sites

BOAT SPECIFIC ITEMS AND GENERAL ITEMS

- PFDs for each person
- Oars
- Bailing device, additional to bilge pump
- Winch/boom apparatus for benthic grabs, velocity measurements, etc.
- Claw-type anchor and mushroom anchor with chain and rope
- Sea Anchor
- Rope (200 feet)
- Bimini and boat cover
- Grease gun
- 2-cycle oil (4 qts)
- Extra 12 v batteries (2)
- Large cleat on bow to secure anchor line

- Wilderness First Aid kit
- USCG book, First Aid book
- Cell Phone
- Digital Camera
- Calculators
- Electronic depth finder
- 5-10 gallons gasoline
- Weather Station (for initial deployment)

Appendix C. List of activities to be completed at each site. After completing an activity, place an X in the circle. Include dates where indicated.

		Activities to Complete For Sample Run 1, 2																						These activities occur on trips prior to and after sampling runs 1, 2		
		Depth/width integrated Water Samples					Grab Water Samples			Benthic Chambers			Other Benthic Measures				Rate Measurements			Channel Dimensions						
SITE NAME	Sampling Trip No.	Nutrients (dissolved, total)	SSC & turbidity	TIC (DIC)	Phyto Chl *a*	Seston CNP	Nutrients (dissolved, total)	SSC & turbidity	TIC (DIC)	SOD	WOD (Winkler)	Ammonia & DIC flux	Benthic Chl *a* (11 trnscts)	% bottom with heavy algae cover (11 trnscts)	% bottom with matching SOD conditions	PERI-1	Phyto photosynthesis, light/dark bottles (Winkler)	Photosynthesis of *Cladophora* (YSI 85)	YSI 85: Cross-check sondes	Mean Depth	Wetted Width	Flow	Low-head dam dimensions	YSI 6600 EDS sondes: Long-term deploy. In, out	Temperature (HOBO logger) in, out	Weather Station in, out
Yellowstone River Sites																										
Buffalo Mirage FAS	1	O	O	O	O	O										O				O	O				O Date in:_______	
	2	O	O	O	O	O										O				O	O				O Date out:_______	
Rosebud (West Unit) FAS	1	O	O	O	O	O													O	O	O		O	O Date in:_______		
	2	O	O	O	O	O													O	O	O		O	O Date out:_______		
Far West FAS	1									O	O	O	O	O	O	O	O			O	O				O Date in:_______	
	2									O	O	O	O	O	O	O	O			O	O				O Date out:_______	
Upstream of Cartersville Canal return flow	1	O	O	O	O	O													O	O	O	O		O Date in:_______		
	2	O	O	O	O	O													O	O	O	O		O Date out:_______		
Ft. Keogh Bridge, u/s of Tongue R.	1	O	O	O	O	O				O	O	O	O	O	O	O	O		O	O	O			O Date in:_______		O Date in:_______
	2	O	O	O	O	O				O	O	O	O	O	O	O	O		O	O	O			O Date out:_______		O Date out:_______
Rosche Jaun FAS	1																	O			O					
	2																	O			O					
Pirogue Island State Park	1									O	O	O	O	O	O	O	O				O				O Date in:_______	
	2									O	O	O	O	O	O	O	O				O				O Date out:_______	
Kinsey Bridge FAS	1	O	O	O	O	O													O	O	O			O Date in:_______		
	2	O	O	O	O	O													O	O	O			O Date out:_______		
Bonfield FAS	1									O	O	O	O	O	O	O	O				O				O Date in:_______	
	2									O	O	O	O	O	O	O	O				O				O Date out:_______	
Upstream of Power R. confluence & Shirley Main Canal return	1	O	O	O	O	O													O	O	O	O		O Date in:_______		
	2	O	O	O	O	O													O	O	O	O		O Date out:_______		
Terry Bridge	1									O	O	O	O	O	O	O	O				O				O Date in:_______	
	2									O	O	O	O	O	O	O	O				O				O Date out:_______	
Upstream of O'Fallon Creek	1	O	O	O	O	O				O	O	O	O	O	O	O	O		O	O	O	O		O Date in:_______		
	2	O	O	O	O	O				O	O	O	O	O	O	O	O		O	O	O	O		O Date out:_______		
11 miles u/s of Glendive (Floaters Guide river mile 445)	1	O	O	O	O	O				O	O	O	O	O	O	O	O		O	O	O	O		O Date in:_______		
	2	O	O	O	O	O				O	O	O	O	O	O	O	O		O	O	O	O		O Date out:_______		
Old Bell Street Bridge, Glendive	1	O	O	O	O	O													O	O	O			O Date in:_______		
	2	O	O	O	O	O													O	O	O			O Date out:_______		
Tributaries & Irrigation Canals																										
Cartersville Canal at withdrawal	1																			O	O	O				
	2																			O	O	O				
Rosebud Cr, confluence w Yellowstone	Opportunistic						O	O	O											O	O	O				
Cartersville Canal return point	1						O	O	O											O	O	O				
	2						O	O	O											O	O	O				
Tongue R., confluence w Yellowstone	1						O	O	O											O	O	O				
	2						O	O	O											O	O	O				
Kinsey Main Canal at withdrawal	1																			O	O	O				
	2																			O	O	O				
Shirley Main Canal withdrawal	1																			O	O	O				
	2																			O	O	O				
Kinsey Main Canal return point	1						O	O	O											O	O	O				
	2						O	O	O											O	O	O				
Shirley Main Canal return point	1						O	O	O											O	O	O				
	2						O	O	O											O	O	O				
Powder R., confluence w Yellowstone	1						O	O	O											O	O	O				
	2						O	O	O											O	O	O				
Terry Main Canal at withdrawal	1																			O	O	O				
	2																			O	O	O				
Terry Main Canal return point 1	Opportunistic						O	O	O											O	O	O				
Terry Main Canal return point 2	Opportunistic						O	O	O																	
O'Fallon Cr, confluence w Yellowstone	Opportunistic						O	O	O											O	O	O				
	Opportunistic																									
	Opportunistic																									
Waste Water Treatment Plant Effluent																										
Forsyth WWTP	Access pending						O	O	O													O				
Terry	Access pending						O	O	O													O				
Miles City WWTP	Access pending						O	O	O													O				

Appendix D. Field Forms Specific to the Yellowstone Modeling Project

Date (m/d/y): ____________________

Table 1. Benthic & Algae Cross-Section Data, and Near-bottom Velocity Values.

	Channel Dimensions & Substrate			Benthic Chlorophyll *a*			Near-bottom River Velocity[‡]	
	Dist. from wet edge	*Channel Wet Width:* ______ m		Method (Hoop-**H**,	Means (boat-**B**,	Estimated bottom cover	Velocity	Means (boat-**B**,
Transect Locale	(m)	Depth (cm)	Substrate Size (mm)*	Core-**C**, Template-**T**)	wading-**W**)	by visible algae (%)[†]	(m sec^{-1})	wading-**W**)
A (Left wet edge)								
B								
C								
D								
E								
F (midchannel)								
G								
H								
I								
J								
K (Right wet edge)								

* If smaller than sand, write "SILT"; if hardpan, write "HP". † If river bottom not visible, write "INV".

‡ Take velocity measurment as near to the river bottom as practicable.

Apendix D2. Yellowstone Modeling Project: Sediment Oxygen Demand and Solute Flux Field Form

Activity ID: ______________________ Site Name: ______________________________ Date (m/d/y): ____________

Table 1. SOD

Benthic Chamber	Measured Near-bottom veloc.(m sec^{-1})	Water Depth (cm)	Dominant Substrate Class*	*Date* : Incubation Start Time	Starting Conditions: YSI 85 Temp (° C)	Starting Conditions: YSI 85 DO (mg/L)	Intermediate DO checks (YSI 85) No. 1 Time	No. 1 DO (mg/L)	No. 2 Time	No. 2 DO (mg/L)	No. 3 Time	No. 3 DO (mg/L)	Incubation End Time	End Date (m/d/y)	Ending Conditions: YSI 85 Temp (° C)	Ending Conditions: YSI 85 DO (mg/L)
Chamber A:																
Chamber B																

* Clay/Silt (FN), Sand (SA), Gravel-fine (GF), Gravel-coarse (GC), Cobble (CB), Boulder (BL), Hardpan (HP)

Table 2. Solute Fluxes

Benthic Chamber	Solute Flux	Start Date	Start Time	Start Sample Volume (ml)	End Date	End Time	End Sample Volume (ml)
Chamber A:	DIC						
	Ammonia						
Chamber B:	DIC						
	Ammonia						

Table 4. Percent of stream cross-section with equivelant SOD

Overall proportion of X-section with similar substrate*: ____________

Overall proportion of X-section with similar velocity*: ____________

Estimated % Cross-Section With Equivelant SOD: ____________

* Refer to 'Benthic & Algae Cross-Section Measurement Form' for individual values of A through K along the transect.

Table 3. WOD via Winkler Titrations

Dark Bottle Replicate	Bottle No.	Time Sample Fixed	Date Sample Fixed	Normality of sodium thisulfate titrant	Volume sodium thiosulfate titrant (ml)	Volume of Sample Titrated (ml)	DO (mg/L)	Date Sample Run	Thiosulfate Titration Start Vol. (ml)	Thiosulfate Titration End Vol. (ml)
1 Initial										
1 repeat measure										
2 Initial										
2 repeat measure										
3 Initial										
3 repeat measure										
1 Final										
1 repeat measure										
2 Final										
2 repeat measure										
3 Final										
3 repeat measure										

Appendix D3. Yellowstone Modeling Project: Light/Dark Bottle (phytoplankton productivity) Field Form

Activity ID: ____________ Site Name: ____________ Date (m/d/y): ____________

Table 1. Light Bottles, Winkler Titration

Light Bottle Replicate	Bottle No.	Incubation Start Time	Incubation End Time (bottle fixed)	Incubation Date	Normality of sodium thisulfate titrant*	Volume sodium thiosulfate titrant (ml)	Volume of Sample Titrated (ml)	DO (mg/L)†	Date Sample Run	Thiosulfate Start Vol. (ml)	Thiosulfate End Vol. (ml)
1											
1 repeat measure											
2											
2 repeat measure											
3											
3 repeat measure											

Table 2. Dark Bottles, Winkler Titration

Dark Bottle Replicate	Bottle No.	Incubation Start Time	Incubation End Time (bottle fixed)	Incubation Date	Normality of sodium thisulfate titrant*	Volume sodium thiosulfate titrant (ml)	Volume of Sample Titrated (ml)	DO (mg/L)†	Date Sample Run	Thiosulfate Start Vol. (ml)	Thiosulfate End Vol. (ml)
1											
1 repeat measure											
2											
2 repeat measure											
3											
3 repeat measure											

*1N thiosulfate solution = 1 M.

† *Based on the formula of Wetzel and Likens (1991)* :

$$\text{mg } O_2/\text{L} = \frac{(\text{ml titrant})(\text{molarity of thiosulfate})(8000)}{(\text{ml sample titrated})\ (\text{ml of BOD bottle-2 / ml of BOD bottle})}$$

USING A COMPUTER WATER-QUALITY MODEL TO DERIVE NUMERIC NUTRIENT CRITERIA FOR A SEGMENT OF THE YELLOWSTONE RIVER

Sampling and Analysis Plan-Addendum

Prepared for:

MONTANA DEPARTMENT OF ENVIRONMENTAL QUALITY
Water Quality Standards Section, Water Quality Planning Bureau
P.O. Box 200901
Helena, MT 59620-0901

Prepared By:

Michael Suplee, Ph.D.
Project Manager
Water Quality Standards Section

October 24, 2007

Purpose of this Addendum

During the sampling phase of the Yellowstone River project (July 30 -September 23, 2007), several modifications to the original SAP were necesitated by realities encountered in the field. This addendum documents these changes. Each section number below refers to the corresponding section in the original SAP. It is recommended that the reader review the original SAP prior to reading this document. Explanations as to why the change was needed are provided with each.

Section 2.2 Overview of What Will be Measured, Where, and How Often

Modifications to the site locations, and rationales for the changes, are shown in Table 2.1. A further explanation is necessary for the Kinsey Bridge FAS modification (Table 2.1). It was intended that the new site (Yellowstone River @ river mile 375) would completely replace the Kinsey Bridge FAS site. However, dropping water levels during the August sampling event created river hazards for the boat, and therefore the YSI was moved downstream to the Kinsey Bridge FAS (which could be accessed by road). Thus, the dataset for the Yellowstone River zone downstream of the Tongue River & Miles City WWTP is in two parts; data collected at river mile 375 (through August 22nd), and data collected at the Kinsey Bridge FAS (August 22nd-September 19th).

Table 2.1 Addendum. Modification of site locations.

Originally Proposed Site	Modification	Explanation
Yellowstone River @ Kinsey Bridge FAS	Yellowstone River @ river mile 375, 5.5 miles upstream of Kinsey Bridge	The original intent of the Kinsey Bridge site was to detect potential influences from the Tongue River and Miles City WWTP. The modified site (river mile 375) was deemed better because it was closer to these river influences (new site was 4 miles downstream of WWTP, Kinsey Bridge was 9.5 miles downstream).
Yellowstone River upstream of Powder River & Shirley Main Canal confluences	Yellowstone River just upstream of Powder River confluence	Dirt road access to site upstream of Powder River had potential (during rain) to render the site impassable for boat & trailer. Boat was required to get upstream of Shirle Main Canal confluence. YSI could be retrieved from modified site without the boat, if required.
Yellowstone River 11 miles upstream of Glendive	Yellowstonr River @ Fallon Bridge FAS	Reaching the Yellowstone River 11 miles upstream of Glendive required either boat travel from Glendive or a local launch site. No local launch was found, and boat travel from Glendive was deemed too hazardous due to rocks and the river's shallowness.

Section 3.1 Sediment Oxygen Demand, Benthic Chambers & Solute Fluxes

Fewer SOD Measurements Completed. SOD measurements turned out to be very time consuming. Further, Steve Chapra (QUAL2K model developer) indicated to DEQ prior to the start of the field sampling that SOD measurements are not the highest priority in overall model development. Therefore, given the large number of project tasks and shortage of time, SOD measurements were collected only at two sites; Far West FAS, and the 1902 Bridge (upstream of Tongue River site), and only for the August (calibration) dataset.

Modifications to SOD Measurement. Measurement of SOD in a river system proved to be very different than what I have experienced in lentic systems. The YSI 6600 sonde dissolved oxygen (DO) data from the first set of duplicated SOD incubations (reviewed in the field) revealed that DO, instead of decreasing over time (as expected), increased instead. As DO increased throughout the day in the river, so too did DO in the chambers. Because the chambers have a skirt that penetrated into the river bottom 10 cm and a second rubber skirt at the sediment/water interface, I believe the DO increase was due to a proportion of river water moving through the coarse gravels of the river bed below the chambers' skirt which then mixed (to some unknown degree) with the water in the chambers. To help control for this, subsequent SOD measurements were carried out with one YSI 6600 sonde in the benthic chamber (experiment) and the other YSI 6600 sonde attached to the outside of the chamber in the flowing river water (control). This arrangement precluded a duplicate chamber incubation because we only had the two YSI sondes available.

Modification to SOD Calculations. A cursory review of the data collected in the modified manner described above showed that DO rose more slowly inside the chambers than outside. Because of this, the time-DO curve generated from each YSI (inside chamber, outside chamber) can be used to estimate SOD. This will be accomplished by determining the difference in the area under the time-DO curve for three scenarios: assuming no mixing of external water with internal chamber water, assuming 50% mixing, assuming 100% mixing. SOD values will be corrected for WOD proportional to each scenario.

Modifications to WOD Measurement. Rather than measure oxygen demand of the water within the chambers (WOD) in triplicate BOD bottles, they were measured in duplicate (two initial and two final dark BOD bottles). This was required due to the limited time available to run replicate measures of WOD within the 3-day holding time.

Other Sediment Fluxes Not Measured. Due to time constraints and the influence of dilution from through-gravel flows into the benthic chambers, we deemed it impractical to measure sediment fluxes of DIC, SRP and ammonia.

Section 3.2. Other Rate Measurements

Light/dark Phytoplankton Productivity Measurements. Light/dark BOD bottles were used to estimate phytoplankton primary productivity. The SAP indicated that water used to fill the light/dark bottles would be drawn from composite water samples composited via the equal-width-increment (EWI) method. We concluded that the process of compositing the water in the carboy would cause too much change in the initial DO concentration of the water sample to make it suitable for the light/dark bottle tests. Instead, the light/dark BOD bottles were filled at the river's surface in good-flowing water. The bottles were carefully filled to avoid gurgling or bubbling so that the initial DO conditions of the river were maintained.

Influence of Drifting Filamentous Algae on DO. Large quantities of drifting filamentous algae (likely *Cladophora spp.*) were observed in the river, and were potentially a strong influence on diel DO patterns. We undertook measurements of the drifting algae at a Yellowstone River site near Miles City. Drifting algae was quantified in two steps. In the first step, small blobs of the drifting filamentous algae were placed in duplicate dark BOD bottles and the change in DO over time was determined. The changes were corrected for the oxygen demand associated with the water fraction in the bottles. The blobs were then frozen for later analysis of dry weight, AFDW and Chl *a*. This provided a DO uptake per unit mass of drifting algae per unit water volume under simulated nighttime conditions. In the second step, a 0.3364 m^2 screen (built from standard window screening) was placed in the river and allowed to capture filamentous algae that drifted through it. The screen was carefully monitored to make sure that it did not begin to plug and consequently route drifting algae around it. The screen was placed where it extended from the surface to the bottom of the river at a location just upstream of the Miles City USGS gage, so that total river flow at the site would be know. The velocity of the water at the screen was recorded using a Marsh McBirney flow meter. The time of accumulation as well as the total dry weight, AFDW and Chl *a* content of the captured algae was determined. These data will be incorporated into the QUAL2K model to help characterize a DO sink (drifting filamentous algae) not anticipated when the SAP was written.

Section 3.2. Real-Time Water Quality Measurements (YSI 6600EDS)

The sonde deployers built were very similar in design to that shown in Fig. 3.1, except that they were constructed entirely from aluminum and did not have concrete slabs as a component. Also, the YSI sondes were attached directly to the deployers with zipties and were not contained within a PVC pipe as shown. None of the deployers were attached to bridges; instead, they were attached to concrete blocks (140 lbs) located upstream of the deployer by ~60 ft of 1/8" stainless steel cable. All were placed in good flowing water approximatly 3-4 ft deep. The YSIs were maintained 10 cm (4") off the bottom when attached to the deployers.

Section 3.5. Water Samples

Modification to Equal-Width-Increment Method. Due to time constraints imposed by the need to keep sampling timelines on schedule, a modified equal-width-increment (EWI) sampling method was employed. The modified EWI method involved ferrying the jet boat back and forth across a channel transect at low speed, while a sampler sat on the bow and carried out a series of continuous dips using a DH48. The technique did a good job of width integration, but only sampled depth to the full length of the DH48 (about 5 ft). In the few cases, a simple grab sample was collected on the river. In these cases, the boat was brought to the midchannel in fast flow and the carboy was filled at the bow from the surface. All site visit forms indicate whether a grab, modified EWI or EWI method was used.

Additional Water Quality Samples. The following additional water quality samples were collected at various Yellowstone River sites, tributaries & canals, or WWTPs: fixed and volatile solids; common ions including alkalinity; and carbonaceous BOD. Exact records for when and where these data were collected are found in the project site visit forms.

Additional Sampling at Reach Headwaters. For both the calibration (August) and validation (September) datasets, an extra water quality sampling event was undertaken at the study-reach headwater site (Rosebud West FAS @ Forsyth). This was done on the return trip to Helena, after the completion of the main sampling run. It typically took about 10 days to complete a sampling run from Rosebud West FAS to the Bell St. Bridge in Glendive (beginning to end of study reach), and in order to determine if water quality conditions had changed at the reach headwaters during this time a second sampling event was undertaken there.

Section 3.7. Hydrologic Measurements

Flow was only measured in tributaries, canals and WWTPs. No flow was measured by DEQ in the Yellowstone River itself. It was concluded that an accurate measure of flow could not be determined using our jet boat. The river was too wide (usually 300 ft or more) to secure a tag line. The boat could be anchored at intervals across the channel, which worked well for collecting water and benthic samples. However, while at anchor, the boat usually had too much port to starboard swing to allow for accurately flow measurement, so river-flow measurements were abandoned. Flow measured in the tributaries, canals and WWTPS was carried out using the 0.6-depth measurement technique. One exception was the Terry WWTP discharge, where flow out the end of the pipe was very small and a timed bucket fill was employed instead.

Section 8.0 Schedule for Completion

Five field trips were originally planned for this project. However, the length of time required to complete each field trip was longer than anticipated. Also, the cleaning &

maintenance of theYSI 6600 sondes, which was originally planned to occur as a stand-alone event, was incorporated into the calibration and validation data-collection field trips. The modified schedule (excluding travel-out and travel-back days) was as follows:

1) Deployment of YSI sondes: July 31-August 8, 2007

2) Sampling Trip No. 1 (calibration dataset): August 17-28, 2007

3) Sampling Trip No. 2 (validation dataset): September 11-September 23, 2007. In addition to collecting samples for the validation dataset, the YSI 6600 sondes were retreived throughout this time period.

APPENDIX B - FIELD FORMS AND SELECT REGRESSION CALCULATIONS

TOTAL DISCHARGE

Date:	8/17/2007	**Site Visit Code:**	**Y0314**
Waterbody:	Cartersville Ditch	**Station ID:**	**Y17CRTMC03**
Personnel:	A. Welch, J. Drygas		

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	0.98		4.2	0.00	0.000	0.000	0.000	
2	0.98		4.2	2.001	1.150	4.830	9.666	
3	3.28		4.2	2.526	1.970	8.274	20.902	
4	4.92		5.2	1.772	1.970	10.244	18.149	
5	7.22		6	1.444	2.295	13.770	19.878	
6	9.51		5.5	2.165	2.295	12.623	27.332	
7	11.81		5.5	2.165	2.135	11.743	25.427	
8	13.78		5.75	0.853	2.295	13.196	11.257	
9	16.4		5.5	1.115	2.625	14.438	16.105	
10	19.03		5.5	1.214	2.955	16.253	19.729	
11	22.31		5	1.148	2.295	11.475	13.177	
12	23.62		5	1.739	1.475	7.375	12.824	
13	25.26		5	1.345	2.460	12.300	16.545	
14	28.54		4	0.000	3.940	15.760	0.000	
15	33.14		2.6	0.000	2.300	5.980	0.000	
16					0.000	0.000	0.000	
17					0.000	0.000	0.000	
18					0.000	0.000	0.000	
19					0.000	0.000	0.000	
20					0.000	0.000	0.000	
21					0.000	0.000	0.000	
22					0.000	0.000	0.000	
23					0.000	0.000	0.000	
24					0.000	0.000	0.000	
25					0.000	0.000	0.000	
26					0.000	0.000	0.000	
27					0.000	0.000	0.000	
28					0.000	0.000	0.000	
29					0.000	0.000	0.000	
30					0.000	0.000	0.000	
31					0.000	0.000	0.000	
32					0.000	0.000	0.000	
33					0.000	0.000	0.000	
34					0.000	0.000	0.000	
35					0.000	0.000	0.000	

Total Discharge = **210.991** **ft^3/sec**

TOTAL DISCHARGE

Date: 8/17/2007 **Site Visit Code:** Y0315

Waterbody: Tongue River **Station ID:** Y16TONGR03

Personnel: A. Welch, J. Drygas

	Distance on tape or from initial point (ft)	Distance (ft)	Depth (ft)	Velocity (at point) (ft/sec)	Width (ft)	Area (sq. ft.)	Discharge (ft^3/sec)	Comments
1	2.95	0.00	0.000	0.000	0.000	0.000	0.000	
2	4.27	1.32	0.100	0.000	0.985	0.099	0.000	
3	4.92	1.97	0.350	1.017	1.145	0.401	0.408	
4	6.56	3.61	0.700	2.920	1.640	1.148	3.352	
5	8.20	5.25	0.800	3.084	1.640	1.312	4.046	
6	9.84	6.89	0.900	3.412	1.640	1.476	5.036	
7	11.48	8.53	1.000	3.346	1.640	1.640	5.488	
8	13.12	10.17	0.950	3.379	1.640	1.558	5.265	
9	14.76	11.81	1.100	3.478	1.640	1.804	6.274	
10	16.40	13.45	1.300	3.018	1.640	2.132	6.435	
11	18.04	15.09	1.850	3.314	1.645	3.043	10.084	
12	19.69	16.74	2.000	3.609	1.645	3.290	11.873	
13	21.33	18.38	2.100	4.134	1.640	3.444	14.237	
14	22.97	20.02	2.100	4.003	1.640	3.444	13.785	
15	24.61	21.66	1.950	3.806	1.640	3.198	12.171	
16	26.25	23.30	1.850	3.642	1.640	3.034	11.049	
17	27.89	24.94	1.700	3.740	1.640	2.788	10.428	
18	29.53	26.58	1.400	3.281	1.640	2.296	7.533	
19	31.17	28.22	1.250	2.592	1.640	2.050	5.313	
20	32.81	29.86	1.000	1.214	1.640	1.640	1.991	
21	34.45	31.50	0.450	0.262	1.640	0.738	0.194	
22	36.09	33.14	0.000	0.000	15.750	0.000	0.000	
23		FALSE			16.570	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **134.962** **ft^3/sec**

TOTAL DISCHARGE

Date:	8/18/2007	**Site Visit Code:**	**Y0316**
Waterbody:	Kinsey Main Canal	**Station ID:**	**Y17KNSMC02**
Personnel:	A. Welch, J. Drygas		

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	1.97	0.00	0.660	0.000	0.000	0.000	0.000	
2	3.28	1.31	0.660	0.000	1.475	0.974	0.000	
3	4.92	2.95	1.310	0.000	1.640	2.148	0.000	
4	6.56	4.59	1.640	0.000	1.640	2.690	0.000	
5	8.20	6.23	2.300	0.459	1.640	3.772	1.731	
6	9.84	7.87	2.950	0.755	1.640	4.838	3.653	
7	11.48	9.51	3.280	1.312	1.640	5.379	7.058	
8	13.12	11.15	3.770	1.739	0.820	3.091	5.376	
9	13.12	11.15	3.770	0.000	0.000	0.000	0.000	
10	13.12	11.15	3.770	1.509	0.820	3.091	4.665	
11	14.76	12.79	3.610	1.936	1.640	5.920	11.462	
12	16.40	14.43	3.610	1.870	1.640	5.920	11.071	
13	18.04	16.07	3.610	1.804	1.645	5.938	10.713	
14	19.69	17.72	3.610	1.903	1.645	5.938	11.301	
15	21.33	19.36	3.280	1.804	1.640	5.379	9.704	
16	22.97	21.00	2.950	1.575	1.640	4.838	7.620	
17	24.61	22.64	2.300	1.181	1.640	3.772	4.455	
18	26.25	24.28	1.970	0.689	1.640	3.231	2.226	
19	27.89	25.92	1.310	0.230	1.640	2.148	0.494	
20	29.53	27.56	0.000	0.000	12.960	0.000	0.000	
21		FALSE			13.780	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	
					Total Discharge =	**91.528**	**ft^3/sec**	

TOTAL DISCHARGE

Date:	8/18/20073	**Site Visit Code:**	**Y0317**
Waterbody:	Kinsey Main Canal	**Station ID:**	**Y17KNSMC01**
Personnel:	A. Welch, J. Drygas		

	Distance on tape or from initial point (ft)	**Distance** (ft)	**0**	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	2.40	0.00	0.000	0.000	0.000	0.000	0.000	
2	2.46	0.06	1.150	0.000	0.110	0.127	0.000	
3	2.62	0.22	1.640	0.000	0.410	0.672	0.000	
4	3.28	0.88	3.440	0.000	0.740	2.546	0.000	
5	4.10	1.70	3.940	0.000	0.820	3.231	0.000	
6	4.92	2.52	5.090	0.066	0.820	4.174	0.275	
7	5.74	3.34	5.090	0.131	0.820	4.174	0.547	
8	6.56	4.16	5.580	0.098	0.820	4.576	0.448	
9	7.38	4.98	5.910	0.066	0.820	4.846	0.320	
10	8.20	5.80	5.740	0.131	0.820	4.707	0.617	
11	9.02	6.62	5.410	0.131	0.820	4.436	0.581	
12	9.84	7.44	5.580	0.098	0.820	4.576	0.448	
13	10.66	8.26	4.760	0.066	0.820	3.903	0.258	
14	11.48	9.08	3.280	0.033	0.575	1.886	0.062	
15	11.81	9.41	0.000	0.000	4.540	0.000	0.000	
16		FALSE			4.705	0.000	0.000	
17		FALSE			0.000	0.000	0.000	
18		FALSE			0.000	0.000	0.000	
19		FALSE			0.000	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = 3.556 ft^3/sec

TOTAL DISCHARGE

Date:	8/18/2007	**Site Visit Code:**	**Y0318**
Waterbody:	Powder River	**Station ID:**	**Y21PWDRR01**
Personnel:	A. Welch, J. Drygas		

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	6.89	0.00	0.000	0.000	0.000	0.000	0.000	
2	8.86	1.97	0.400	0.361	2.295	0.918	0.331	
3	11.48	4.59	0.600	0.984	3.770	2.262	2.226	
4	16.40	9.51	0.400	1.411	5.745	2.298	3.242	
5	22.97	16.08	0.700	1.476	6.565	4.596	6.783	
6	29.53	22.64	0.450	1.444	6.560	2.952	4.263	
7	36.09	29.20	0.600	1.509	6.560	3.936	5.939	
8	42.65	35.76	0.650	1.870	5.740	3.731	6.977	
9	47.57	40.68	0.800	2.133	4.920	3.936	8.395	
10	52.49	45.60	0.900	2.461	4.920	4.428	10.897	
11	57.41	50.52	1.000	2.034	4.925	4.925	10.017	
12	62.34	55.45	1.150	1.837	4.925	5.664	10.404	
13	67.26	60.37	1.150	1.903	4.920	5.658	10.767	
14	72.18	65.29	1.000	2.165	4.100	4.100	8.876	
15	75.46	68.57	0.900	1.739	4.100	3.690	6.417	
16	80.38	73.49	0.950	1.509	4.920	4.674	7.053	
17	85.30	78.41	0.800	1.214	4.920	3.936	4.778	
18	90.22	83.33	0.550	0.656	4.920	2.706	1.775	
19	95.14	88.25	0.300	0.066	3.940	1.182	0.078	
20	98.10	91.21	0.000	0.000	44.125	0.000	0.000	
21		FALSE			45.605	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **109.221** **ft^3/sec**

TOTAL DISCHARGE

Date: 8/20/2007 **Site Visit Code:** **Y0325**

Waterbody: Forsyth WWTP **Station ID:** **Y17FWWTP01**

Personnel: A. Welch, J. Drygas

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	3.12	0.00	0.000	0.000	0.000	0.000	0.000	
2	2.95	0.17	0.350	0.591	0.165	0.058	0.034	
3	2.79	0.33	0.500	0.558	0.165	0.083	0.046	
4	2.62	0.50	0.650	0.492	0.165	0.107	0.053	
5	2.46	0.66	0.700	0.427	0.160	0.112	0.048	
6	2.30	0.82	0.800	0.197	0.165	0.132	0.026	
7	2.13	0.99	0.800	0.066	0.165	0.132	0.009	
8	1.97	1.15	0.750	0.000	0.165	0.124	0.000	
9	1.80	1.32	0.700	0.098	0.165	0.116	0.011	
10	1.64	1.48	0.700	0.000	0.245	0.172	0.000	
11	1.31	1.81	0.600	0.000	0.330	0.198	0.000	
12	0.98	2.14	0.500	0.000	0.325	0.163	0.000	
13	0.66	2.46	0.400	0.000	0.245	0.098	0.000	
14	0.49	2.63	0.300	0.000	1.230	0.369	0.000	
15	0.00	FALSE	0.000	0.000	1.315	0.000	0.000	
16		FALSE			0.000	0.000	0.000	
17		FALSE			0.000	0.000	0.000	
18		FALSE			0.000	0.000	0.000	
19		FALSE			0.000	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **0.227** **ft^3/sec**

TOTAL DISCHARGE

Date: 9/10/2007 **Site Visit Code:** Y0327

Waterbody: Cartersville Main Ditch **Station ID:** Y17CRTMC03

Personnel: A. Welch, A. Nixon

	Distance on tape or from initial point (ft)	Distance (ft)	Depth (ft)	Velocity (at point) (ft/sec)	Width (ft)	Area (sq. ft.)	Discharge (ft^3/sec)	Comments
1	3.28	0.00	3.440	0.000	0.000	0.000	0.000	
2	4.92	1.64	3.440	0.000	0.820	2.821	0.000	
3	4.92	1.64	3.440	0.000	0.820	2.821	0.000	
4	6.56	3.28	4.100	0.000	1.640	6.724	0.000	
5	8.20	4.92	4.100	0.000	1.475	6.048	0.000	
6	9.51	6.23	6.070	0.984	0.655	3.976	3.912	
7	9.51	6.23	6.070	0.131	0.985	5.979	0.783	
8	11.48	8.20	5.910	0.525	0.985	5.821	3.056	
9	11.48	8.20	5.910	0.886	0.820	4.846	4.294	
10	13.12	9.84	5.910	0.558	0.820	4.846	2.704	
11	13.12	9.84	5.910	1.247	0.820	4.846	6.043	
12	14.76	11.48	5.910	0.197	0.820	4.846	0.955	
13	14.76	11.48	5.910	0.427	0.820	4.846	2.069	
14	16.40	13.12	6.070	1.247	0.820	4.977	6.207	
15	16.40	13.12	6.070	0.295	0.820	4.977	1.468	
16	18.04	14.76	6.070	1.870	0.820	4.977	9.308	
17	18.04	14.76	6.070	1.017	0.825	5.008	5.093	
18	19.69	16.41	6.230	1.706	0.825	5.140	8.768	
19	19.69	16.41	6.230	1.739	0.820	5.109	8.884	
20	21.33	18.05	5.560	0.722	0.820	4.559	3.292	
21	21.33	18.05	6.560	1.640	0.410	2.690	4.411	
22	22.15	18.87	6.890	0.098	0.410	2.825	0.277	
23	22.15	18.87	6.890	1.247	0.735	5.064	6.315	
24	23.62	20.34	6.230	2.428	0.735	4.579	11.118	
25	23.62	20.34	6.230	1.509	10.170	63.359	95.609	
26		FALSE			10.170	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **184.566** **ft^3/sec**

TOTAL DISCHARGE

Date: 9/11/2007 **Site Visit Code:** **Y0328**

Waterbody: Kinsey Main Canal **Station ID:** **Y17KNSMC02**

Personnel: A. Welch, A. Nixon

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	2.30	0.00	1.310	0.000	0.000	0.000	0.000	
2	3.94	1.64	1.640	0.000	2.130	3.493	0.000	
3	6.56	4.26	2.300	0.033	2.130	4.899	0.162	
4	8.20	5.90	3.120	0.689	1.640	5.117	3.525	
5	9.84	7.54	3.610	1.312	1.640	5.920	7.768	
6	11.48	9.18	3.940	1.772	1.640	6.462	11.450	
7	13.12	10.82	4.270	1.903	1.640	7.003	13.326	
8	14.76	12.46	4.270	2.100	1.640	7.003	14.706	
9	16.40	14.10	3.940	2.001	1.640	6.462	12.930	
10	18.04	15.74	3.610	1.804	1.645	5.938	10.713	
11	19.69	17.39	3.280	1.542	1.645	5.396	8.320	
12	21.33	19.03	2.950	1.345	1.640	4.838	6.507	
13	22.97	20.67	2.620	0.853	1.640	4.297	3.665	
14	24.61	22.31	2.300	0.131	1.640	3.772	0.494	
15	26.25	23.95	1.480	0.000	1.230	1.820	0.000	
16	27.07	24.77	1.310	0.000	11.975	15.687	0.000	
17		FALSE			12.385	0.000	0.000	
18		FALSE			0.000	0.000	0.000	
19		FALSE			0.000	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **93.566** **ft^3/sec**

TOTAL DISCHARGE

Date: 9/11/2007 **Site Visit Code:** Y0329

Waterbody: Kinsey Main Canal **Station ID:** Y17KNSMC01

Personnel:

	Distance on tape or from initial point (ft)	Distance (ft)	Depth (ft)	Velocity (at point) (ft/sec)	Width (ft)	Area (sq. ft.)	Discharge (ft^3/sec)	Comments
1	2.30	0.00	1.400	0.098	0.000	0.000	0.000	
2	2.95	0.65	2.300	0.689	0.655	1.507	1.038	
3	3.61	1.31	2.350	1.148	0.660	1.551	1.781	
4	4.27	1.97	2.400	1.214	0.655	1.572	1.908	
5	4.92	2.62	2.600	1.378	0.655	1.703	2.347	
6	5.58	3.28	2.700	1.640	0.655	1.769	2.900	
7	6.23	3.93	2.800	1.575	0.655	1.834	2.889	
8	6.89	4.59	2.700	1.673	0.660	1.782	2.981	
9	7.55	5.25	2.700	1.575	0.655	1.769	2.785	
10	8.20	5.90	2.600	1.411	0.655	1.703	2.403	
11	8.86	6.56	2.600	1.214	0.655	1.703	2.067	
12	9.51	7.21	2.400	1.148	0.655	1.572	1.805	
13	10.17	7.87	2.200	1.181	0.660	1.452	1.715	
14	10.83	8.53	2.000	0.886	0.490	0.980	0.868	
15	11.15	8.85	1.800	0.787	0.325	0.585	0.460	
16	11.48	9.18	0.000	0.000	4.425	0.000	0.000	
17		FALSE			4.590	0.000	0.000	
18		FALSE			0.000	0.000	0.000	
19		FALSE			0.000	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **27.948** **ft^3/sec**

TOTAL DISCHARGE

Date:	9/11/2007	**Site Visit Code:**	Y0330
Waterbody:	O'Fallon Creek	**Station ID:**	Y22OFALC16
Personnel:	A. Welch, A. Nixon		

	Distance on tape or from initial point (ft)	Distance (ft)	Depth (ft)	Velocity (at point) (ft/sec)	Width (ft)	Area (sq. ft.)	Discharge (ft^3/sec)	Comments
1	3.94	0.00	0.000	0.000	0.000	0.000	0.000	
2	4.92	0.98	0.200	0.098	1.310	0.262	0.026	
3	6.56	2.62	0.520	0.394	1.640	0.853	0.336	
4	8.20	4.26	0.700	0.525	1.640	1.148	0.603	
5	9.84	5.90	0.800	0.328	1.640	1.312	0.430	
6	11.48	7.54	0.820	0.328	1.640	1.345	0.441	
7	13.12	9.18	0.920	0.328	1.640	1.509	0.495	
8	14.76	10.82	0.800	0.525	1.640	1.312	0.689	
9	16.40	12.46	0.800	0.328	1.640	1.312	0.430	
10	18.04	14.10	0.680	0.591	1.645	1.119	0.661	
11	19.69	15.75	0.750	0.394	1.645	1.234	0.486	
12	21.33	17.39	0.700	0.394	1.640	1.148	0.452	
13	22.97	19.03	0.650	0.197	1.640	1.066	0.210	
14	24.61	20.67	0.500	0.525	1.640	0.820	0.431	
15	26.25	22.31	0.440	0.262	1.640	0.722	0.189	
16	27.89	23.95	0.300	0.000	1.640	0.492	0.000	
17	29.53	25.59	0.200	0.000	1.310	0.262	0.000	
18	30.51	26.57	0.000	0.000	12.795	0.000	0.000	
19		FALSE			13.285	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **5.879** **ft^3/sec**

TOTAL DISCHARGE

Date:	9/11/2007	**Site Visit Code:**	**Y0331**
Waterbody:	Powder River	**Station ID:**	**Y21PWDRR01**
Personnel:	A. Welch, A. Nixon		

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	7.87	0.00	0.000	0.000	0.000	0.000	0.000	
2	9.84	1.97	0.900	1.050	2.625	2.363	2.481	
3	13.12	5.25	1.050	1.083	3.280	3.444	3.730	
4	16.40	8.53	0.950	1.542	3.285	3.121	4.812	
5	19.69	11.82	0.800	1.050	3.285	2.628	2.759	
6	22.97	15.10	0.700	1.673	3.280	2.296	3.841	
7	26.25	18.38	0.700	1.837	3.280	2.296	4.218	
8	29.53	21.66	0.980	1.608	3.280	3.214	5.169	
9	32.81	24.94	1.050	1.673	3.280	3.444	5.762	
10	36.09	28.22	1.100	1.345	3.280	3.608	4.853	
11	39.37	31.50	1.100	1.804	3.280	3.608	6.509	
12	42.65	34.78	1.200	1.640	3.280	3.936	6.455	
13	45.93	38.06	1.150	1.739	3.280	3.772	6.560	
14	49.21	41.34	0.950	1.837	3.280	3.116	5.724	
15	52.49	44.62	0.900	1.870	3.280	2.952	5.520	
16	55.77	47.90	0.750	1.706	3.285	2.464	4.203	
17	59.06	51.19	0.720	1.575	3.285	2.365	3.725	
18	62.34	54.47	0.500	1.280	3.280	1.640	2.099	
19	65.62	57.75	0.300	0.656	2.540	0.762	0.500	
20	67.42	59.55	0.000	0.000	28.875	0.000	0.000	
21		FALSE			29.775	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **78.919** **ft^3/sec**

TOTAL DISCHARGE

Date: 8/18/2007 **Site Visit Code:** Y0333

Waterbody: Rosebud Creek at Mouth **Station ID:** Y14ROSBC04

Personnel: M. Stermitz, M. Suplee

	Distance on tape or from initial point (ft)	Distance (ft)	Depth (ft)	Velocity (at point) (ft/sec)	Width (ft)	Area (sq. ft.)	Discharge (ft^3/sec)	Comments
1	7.71	0.00	0.100	0.000	0.000	0.000	0.000	
2	9.51	1.80	0.000	0.000	1.560	0.000	0.000	
3	10.83	3.12	0.100	0.000	1.150	0.115	0.000	
4	11.81	4.10	0.200	0.820	1.145	0.229	0.188	
5	13.12	5.41	0.350	1.411	1.150	0.403	0.568	
6	14.11	6.40	0.300	1.542	0.985	0.296	0.456	
7	15.09	7.38	0.500	0.689	0.985	0.492	0.339	
8	16.08	8.37	0.750	1.083	0.985	0.739	0.800	
9	17.06	9.35	0.800	1.706	0.980	0.784	1.338	
10	18.04	10.33	0.650	1.345	0.985	0.640	0.861	
11	19.03	11.32	0.500	1.378	0.985	0.493	0.679	
12	20.01	12.30	0.300	0.098	0.985	0.296	0.029	
13	21.00	13.29	0.300	1.673	0.985	0.296	0.494	
14	21.98	14.27	0.350	1.673	0.985	0.345	0.577	
15	22.97	15.26	0.200	0.131	0.985	0.197	0.026	
16	23.95	16.24	0.200	0.000	0.980	0.196	0.000	
17	24.93	17.22	0.100	0.000	1.310	0.131	0.000	
18	26.57	18.86	0.000	0.000	8.610	0.000	0.000	
19		FALSE			9.430	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = 6.354 ft^3/sec

TOTAL DISCHARGE

Date: 8/20/2007 **Site Visit Code:** **Y0337**

Waterbody: Cartersville Main Canal Return **Station ID:** **Y17CRTMC01**

Personnel: M. Suplee, M. Stermitz

	Distance on tape or from initial point (ft)	Distance (ft)	Depth (ft)	Velocity (at point) (ft/sec)	Width (ft)	Area (sq. ft.)	Discharge (ft^3/sec)	Comments
1	25.92	0.00	0.400	0.000	0.000	0.000	0.000	
2	24.61	1.31	1.200	2.198	1.315	1.578	3.468	
3	23.29	2.63	1.500	2.428	1.315	1.973	4.789	
4	21.98	3.94	1.900	2.461	1.310	2.489	6.125	
5	20.67	5.25	2.000	2.986	1.310	2.620	7.823	
6	19.36	6.56	2.000	2.690	1.315	2.630	7.075	
7	18.04	7.88	2.200	2.920	1.315	2.893	8.448	
8	16.73	9.19	2.200	2.559	1.310	2.882	7.375	
9	15.42	10.50	1.900	2.756	1.310	2.489	6.860	
10	14.11	11.81	1.500	2.428	1.310	1.965	4.771	
11	12.80	13.12	1.300	2.100	1.315	1.710	3.590	
12	11.48	14.44	1.300	1.706	1.315	1.710	2.916	
13	10.17	15.75	1.300	1.673	1.310	1.703	2.849	
14	8.86	17.06	1.000	1.706	1.310	1.310	2.235	
15	7.55	18.37	0.900	1.181	1.315	1.184	1.398	
16	6.23	19.69	0.600	0.459	1.315	0.789	0.362	
17	4.92	21.00	0.700	0.131	1.065	0.746	0.098	
18	4.10	21.82	0.100	0.000	10.500	1.050	0.000	
19		FALSE			10.910	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **70.182** **ft^3/sec**

TOTAL DISCHARGE

Date:	8/23/2007	**Site Visit Code:**	**Y0345**
Waterbody:	Shirley Main Canal	**Station ID:**	**Y17SR YMC01**
Personnel:			

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	3.28	0.00	0.200	0.010	0.000	0.000	0.000	
2	3.94	0.66	0.550	2.780	0.655	0.360	1.001	
3	4.59	1.31	0.400	2.280	0.655	0.262	0.597	
4	5.25	1.97	0.550	2.410	0.660	0.363	0.875	
5	5.91	2.63	0.650	2.630	0.655	0.426	1.120	
6	6.56	3.28	0.650	2.890	0.655	0.426	1.230	
7	7.22	3.94	0.700	0.650	0.655	0.459	0.298	
8	7.87	4.59	0.700	3.030	0.655	0.459	1.389	
9	8.53	5.25	0.550	2.960	0.660	0.363	1.074	
10	9.19	5.91	0.650	2.500	0.655	0.426	1.064	
11	9.84	6.56	0.600	2.550	0.655	0.393	1.002	
12	10.50	7.22	0.500	2.270	0.655	0.328	0.743	
13	11.15	7.87	0.300	2.560	0.655	0.197	0.503	
14	11.81	8.53	0.450	1.850	0.660	0.297	0.549	
15	12.47	9.19	0.500	2.720	0.655	0.328	0.891	
16	13.12	9.84	0.350	2.350	0.655	0.229	0.539	
17	13.78	10.50	0.400	2.470	0.660	0.264	0.652	
18	14.44	11.16	0.550	2.160	0.655	0.360	0.778	
19	15.09	11.81	0.600	2.650	0.655	0.393	1.041	
20	15.75	12.47	0.600	2.120	0.655	0.393	0.833	
21	16.40	13.12	0.200	1.300	0.510	0.102	0.133	
22	16.77	13.49	0.200	1.030	6.560	1.312	1.351	
23		FALSE			6.745	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **17.666** **ft^3/sec**

TOTAL DISCHARGE

Date:	8/26/2007	**Site Visit Code:**	**Y0346**
Waterbody:	O'Fallon Creek at Hwy 10 Bridge	**Station ID:**	**Y22OFALC16**
Personnel:	R. Sada, M. Suplee		

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	25.92	0.00	0.000	0.000	0.000	0.000	0.000	
2	24.61	1.31	0.100	0.000	1.315	0.132	0.000	
3	23.29	2.63	0.000	0.000	1.315	0.000	0.000	
4	21.98	3.94	0.200	0.130	1.310	0.262	0.034	
5	20.67	5.25	0.300	0.200	1.310	0.393	0.079	
6	19.36	6.56	0.400	0.360	1.315	0.526	0.189	
7	18.04	7.88	0.500	0.430	1.315	0.658	0.283	
8	16.73	9.19	0.600	0.390	1.310	0.786	0.307	
9	15.42	10.50	0.500	0.490	1.310	0.655	0.321	
10	14.11	11.81	0.650	0.430	1.150	0.748	0.321	
11	13.12	12.80	0.450	0.460	1.150	0.518	0.238	
12	11.81	14.11	0.450	0.300	1.310	0.590	0.177	
13	10.50	15.42	0.300	0.390	3.945	1.184	0.462	
14	19.70	6.22	0.400	0.430	1.315	0.526	0.226	
15	7.87	18.05	0.300	0.230	6.570	1.971	0.453	
16	6.56	19.36	0.400	0.390	1.310	0.524	0.204	
17	5.25	20.67	0.350	0.330	1.310	0.459	0.151	
18	3.94	21.98	0.350	0.200	1.315	0.460	0.092	
19	2.62	23.30	0.300	0.100	1.315	0.395	0.039	
20	1.31	24.61	0.200	0.000	11.650	2.330	0.000	
21	0.00	FALSE	0.000	0.000	12.305	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = 3.577 ft^3/sec

TOTAL DISCHARGE

Date:	9/15/2007	**Site Visit Code:**	**Y0364**
Waterbody:	Cartersville Canal	**Station ID:**	**Y17CRTMC01**
Personnel:	R. Sada, M. Suplee		

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	28.87	0.00	0.000	0.000	0.000	0.000	0.000	
2	27.56	1.31	0.200	0.000	1.310	0.262	0.000	
3	26.25	2.62	0.350	0.000	1.315	0.460	0.000	
4	24.93	3.94	0.800	0.262	1.315	1.052	0.276	
5	23.62	5.25	1.250	0.492	1.310	1.638	0.806	
6	22.31	6.56	1.400	0.820	1.310	1.834	1.504	
7	21.00	7.87	1.550	1.115	1.310	2.031	2.264	
8	19.69	9.18	1.600	1.148	1.315	2.104	2.415	
9	18.37	10.50	1.800	0.984	1.315	2.367	2.329	
10	17.06	11.81	2.050	1.115	1.310	2.686	2.994	
11	15.75	13.12	2.300	1.312	1.310	3.013	3.953	
12	14.44	14.43	2.450	1.542	1.315	3.222	4.968	
13	13.12	15.75	2.550	1.575	1.315	3.353	5.281	
14	11.81	17.06	2.550	1.444	1.310	3.341	4.824	
15	10.50	18.37	2.350	1.312	1.310	3.079	4.039	
16	9.19	19.68	2.100	1.280	1.315	2.762	3.535	
17	7.87	21.00	1.800	1.312	1.315	2.367	3.106	
18	6.56	22.31	1.550	1.214	1.310	2.031	2.465	
19	5.25	23.62	1.300	0.820	1.310	1.703	1.396	
20	3.94	24.93	1.000	0.623	1.315	1.315	0.819	
21	2.62	26.25	0.200	0.000	12.465	2.493	0.000	
22		FALSE			13.125	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **46.974** **ft^3/sec**

TOTAL DISCHARGE

Date: 9/11/2007 **Site Visit Code:** **Y0365**

Waterbody: Tongue River **Station ID:** **Y16TONGR03**

Personnel: A. Welch, A. Nixon

	Distance on tape or from initial point (ft)	Distance (ft)	Depth (ft)	Velocity (at point) (ft/sec)	Width (ft)	Area (sq. ft.)	Discharge (ft^3/sec)	Comments
1	4.92	0.00	0.000	0.000	0.000	0.000	0.000	
2	6.56	1.64	0.820	1.280	4.100	3.362	4.303	
3	13.12	8.20	1.750	2.297	6.565	11.489	26.390	
4	19.69	14.77	1.300	2.001	6.565	8.535	17.078	
5	26.25	21.33	1.500	2.001	6.560	9.840	19.690	
6	32.81	27.89	1.250	2.001	6.560	8.200	16.408	
7	39.37	34.45	0.950	1.837	6.560	6.232	11.448	
8	45.93	41.01	0.900	1.870	6.560	5.904	11.040	
9	52.49	47.57	0.900	1.673	6.565	5.909	9.885	
10	59.06	54.14	0.750	1.706	6.565	4.924	8.400	
11	65.62	60.70	0.700	1.772	6.560	4.592	8.137	
12	72.18	67.26	0.700	1.903	6.560	4.592	8.739	
13	78.74	73.82	0.900	1.936	6.560	5.904	11.430	
14	85.30	80.38	0.800	2.100	6.560	5.248	11.021	
15	91.86	86.94	0.900	1.903	6.565	5.909	11.244	
16	98.43	93.51	0.700	1.673	6.565	4.596	7.688	
17	104.99	100.07	0.600	1.739	6.560	3.936	6.845	
18	111.55	106.63	0.600	1.608	6.560	3.936	6.329	
19	118.11	113.19	0.500	1.509	8.200	4.100	6.187	
20	127.95	123.03	0.400	1.214	9.845	3.938	4.781	
21	137.80	132.88	0.400	1.050	9.845	3.938	4.135	
22	147.64	142.72	0.200	0.591	7.380	1.476	0.872	
23	152.56	147.64	0.600	0.558	3.445	2.067	1.153	
24	154.53	149.61	0.000	0.000	73.820	0.000	0.000	
25		FALSE			74.805	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **213.203** **ft^3/sec**

TOTAL DISCHARGE

Date: 9/12/2007 **Site Visit Code:** **Y0368**

Waterbody: Rosebud Creek **Station ID:** **Y14ROSBC05**

Personnel: A. Welch, A. Nixon

	Distance on tape or from initial point (ft)	Distance (ft)	Depth (ft)	Velocity (at point) (ft/sec)	Width (ft)	Area (sq. ft.)	Discharge (ft^3/sec)	Comments
1	5.91	0.00	0.000	0.000	0.000	0.000	0.000	
2	6.56	0.65	0.300	0.328	0.820	0.246	0.081	
3	7.55	1.64	0.470	0.066	0.985	0.463	0.031	
4	8.53	2.62	0.500	0.066	0.980	0.490	0.032	
5	9.51	3.60	0.500	0.000	0.985	0.493	0.000	
6	10.50	4.59	0.730	0.197	0.985	0.719	0.142	
7	11.48	5.57	0.870	0.722	0.820	0.713	0.515	
8	12.14	6.23	1.000	1.083	0.660	0.660	0.715	
9	12.80	6.89	0.950	1.247	0.655	0.622	0.776	
10	13.45	7.54	0.800	1.214	0.655	0.524	0.636	
11	14.11	8.20	0.700	0.755	0.655	0.459	0.346	
12	14.76	8.85	0.500	1.083	0.655	0.328	0.355	
13	15.42	9.51	0.470	0.984	0.820	0.385	0.379	
14	16.40	10.49	0.300	0.623	0.985	0.296	0.184	
15	17.39	11.48	0.220	0.459	0.985	0.217	0.099	
16	18.37	12.46	0.150	0.131	1.310	0.197	0.026	
17	20.01	14.10	0.000	0.000	6.230	0.000	0.000	
18		FALSE			7.050	0.000	0.000	
19		FALSE			0.000	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **4.317** **ft^3/sec**

TOTAL DISCHARGE

Date: 9/12/2007 **Site Visit Code:** **Y0369**

Waterbody: Forsyth WWTP **Station ID:** **Y17WWTP01**

Personnel: A. Welch, A. Nixon

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	0.66	0.00	0.000	0.000	0.000	0.000	0.000	
2	0.98	0.32	0.100	0.000	0.325	0.033	0.000	
3	1.31	0.65	0.200	0.000	0.330	0.066	0.000	
4	1.64	0.98	0.330	0.427	0.330	0.109	0.047	
5	1.97	1.31	0.500	0.459	0.330	0.165	0.076	
6	2.30	1.64	0.350	0.656	0.325	0.114	0.075	
7	2.62	1.96	0.300	0.591	0.325	0.098	0.058	
8	2.95	2.29	0.300	0.459	0.330	0.099	0.045	
9	3.28	2.62	0.300	0.066	0.330	0.099	0.007	
10	3.61	2.95	0.200	0.131	0.330	0.066	0.009	
11	3.94	3.28	0.200	0.066	0.330	0.066	0.004	
12	4.27	3.61	0.100	0.000	0.325	0.033	0.000	
13	4.59	3.93	0.000	0.000	1.475	0.000	0.000	
14		0.66			1.965	0.000	0.000	
15		FALSE			0.330	0.000	0.000	
16		FALSE			0.000	0.000	0.000	
17		FALSE			0.000	0.000	0.000	
18		FALSE			0.000	0.000	0.000	
19		FALSE			0.000	0.000	0.000	
20		FALSE			0.000	0.000	0.000	
21		FALSE			0.000	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

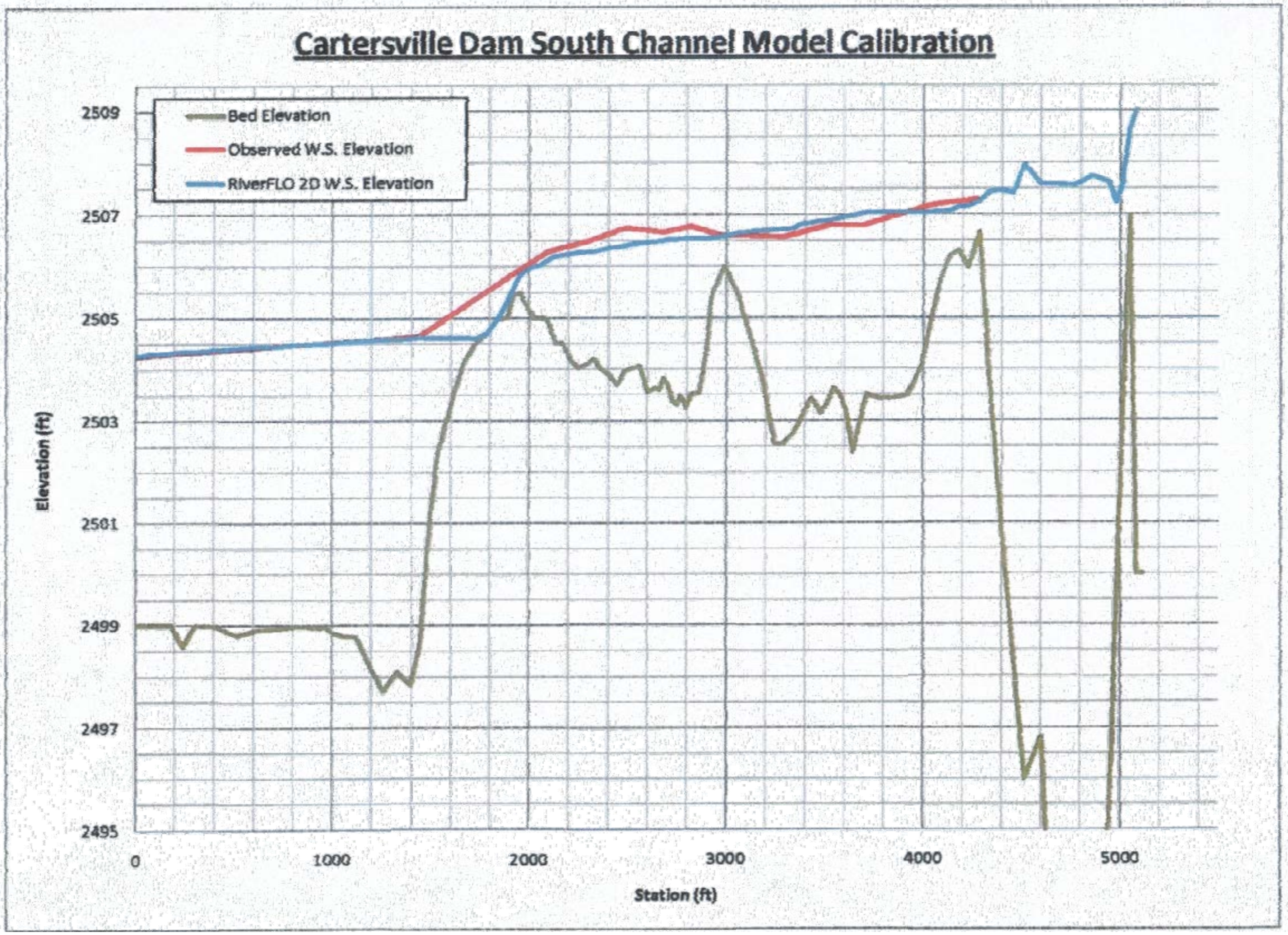
Total Discharge = **0.319** **ft^3/sec**

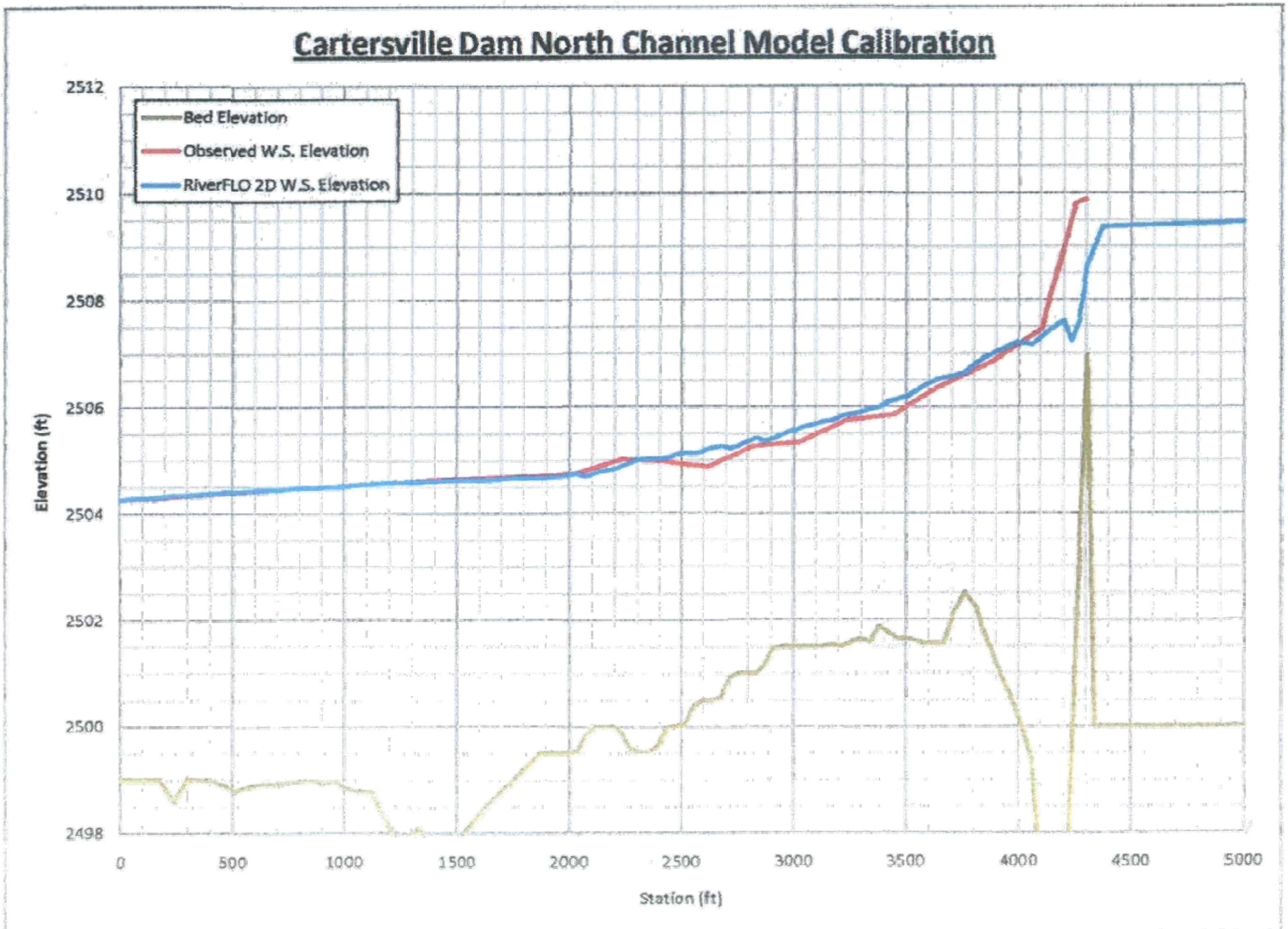
TOTAL DISCHARGE

Date:	9/16/2007	**Site Visit Code:**	**Y0376**
Waterbody:	Shirley Main Canal	**Station ID:**	**Y17SRYMC01**
Personnel:	R. Sada, M. Suplee		

	Distance on tape or from initial point (ft)	**Distance** (ft)	**Depth** (ft)	**Velocity (at point)** (ft/sec)	**Width** (ft)	**Area** (sq. ft.)	**Discharge** (ft^3/sec)	**Comments**
1	1.80	0.00	0.000	0.000	0.000	0.000	0.000	
2	2.89	1.09	0.400	0.164	1.085	0.434	0.071	
3	3.97	2.17	0.500	0.459	1.080	0.540	0.248	
4	5.05	3.25	0.500	0.427	1.085	0.543	0.232	
5	6.14	4.34	0.450	0.459	1.085	0.488	0.224	
6	7.22	5.42	0.450	0.262	1.080	0.486	0.127	
7	8.30	6.50	0.400	0.591	1.080	0.432	0.255	
8	9.38	7.58	0.350	0.853	1.085	0.380	0.324	
9	10.47	8.67	0.450	1.050	1.085	0.488	0.513	
10	11.55	9.75	0.450	1.640	1.080	0.486	0.797	
11	12.63	10.83	0.500	1.673	1.080	0.540	0.903	
12	13.71	11.91	0.500	1.903	1.085	0.543	1.032	
13	14.80	13.00	0.550	1.673	1.085	0.597	0.998	
14	15.88	14.08	0.600	2.461	1.080	0.648	1.595	
15	16.96	15.16	0.650	2.625	1.080	0.702	1.843	
16	18.04	16.24	0.700	2.723	1.085	0.759	2.068	
17	19.13	17.33	0.700	2.362	1.085	0.760	1.794	
18	20.21	18.41	0.550	2.395	1.080	0.594	1.423	
19	21.29	19.49	0.550	1.772	1.085	0.597	1.057	
20	22.38	20.58	0.300	0.262	9.745	2.924	0.766	
21		FALSE			10.290	0.000	0.000	
22		FALSE			0.000	0.000	0.000	
23		FALSE			0.000	0.000	0.000	
24		FALSE			0.000	0.000	0.000	
25		FALSE			0.000	0.000	0.000	
26		FALSE			0.000	0.000	0.000	
27		FALSE			0.000	0.000	0.000	
28		FALSE			0.000	0.000	0.000	
29		FALSE			0.000	0.000	0.000	
30		FALSE			0.000	0.000	0.000	
31		FALSE			0.000	0.000	0.000	
32		FALSE			0.000	0.000	0.000	
33		FALSE			0.000	0.000	0.000	
34		FALSE			0.000	0.000	0.000	

Total Discharge = **16.271** **ft^3/sec**


Cartersville Dam South Channel Model Calibration
Bed Elevation
Observed W.S. Elevation
RiverFLO 2D W.S. Elevation
Elevation (ft)
2509
2507
2505
2503
2501
2499
2497
2495
0
1000
2000
3000
4000
5000
Station (ft)



Cartersville Dam North Channel Model Calibration
Bed Elevation
Observed W.S. Elevation
RiverFLO 2D W.S. Elevation
Elevation (ft)
2512
2510
2508
2506
2504
2502
2500
2498
0
500
1000
1500
2000
2500
3000
3500
4000
4500
5000
Station (ft)

YELLOWSTONE RIVER WATER BALANCE; 8/17-26/2007 - CALIBRATION PERIOD

Map ID	Monitoring Date(s)	Site Name	Data Src (Site Code)	Flow (cfs)[1]	Flow (cms)[1]	Balance (cfs)	Balance (cms)	Groundwater Accretion (cfs)	Groundwater Accretion (cms)	Percentage of Surface Q (%)
1	**Avg for period**	**USGS Yellowstone River at Forsyth**	**USGS (06295000)**	**3,526.0**	**99.849**					
2	Avg for period	Forsyth WTP	MDEQ (from city)	-0.8	-0.022	3,525.2	99.827			
3	17-Aug	Cartersville Irrigation District DVT	MDEQ (Y0314)	-211.0	-5.975	3,314.2	93.852			
4	Avg for period	Forsyth WWTP	City of Forsyth	0.4	0.011	3,314.6	93.864			
5	18-Aug	Rosebud Creek	MDEQ (Y0333)	6.4	0.180	3,321.0	94.044			
6	20-Aug	Cartersville Irrigation District RTN	MDEQ (Y0337)	70.2	1.987	3,391.2	96.031			
7	Avg for period	Baringer Pumping Project DVT	MDEQ (estimated)	-22.4	-0.635	3,368.8	95.396			
8	Avg for period	Baringer Pumping Project RTN	MDEQ (estimated)	0.0	0.000	3,368.8	95.396			
9	Avg for period	Private Irrigation (pumps from YR) DVT	MDEQ (estimated)	-89.8	-2.543	3,278.9	92.853			
10	Avg for period	Private Irrigation (pumps from YR) RTN	MDEQ (estimated)	5.8	0.164	3,284.7	93.017			
11	Avg for period	Miles City WTP	MDEQ (from city)	-3.6	-0.102	3,281.1	92.915			
12	Avg for period	Tongue River	USGS (06308500)	184.6	5.227	3,465.7	98.142			
	Avg for period	Unmonitored Tributaries	MDEQ (estimated)	6.1	0.173	3,471.9	98.316			
	Avg for period	Unmonitored Waste Drains	MDEQ (estimated)	55.0	1.558	3,526.9	99.873			
	Avg for period	Evaporation	NOAA	-21.2	-0.601	3,505.7	99.273			
	Avg for period	**USGS Yellowstone River at Miles City**	**USGS (06309000)**	**3,762.0**	**106.532**	**3,505.7**	**99.273**	**256.3**	**7.259**	**6.9%**
					ADJUST	**3,762.0**	**106.53**	**GAINING REACH**		
13	Avg for period	Miles City WWTP	City of Miles City	1.8	0.052	3,763.8	106.584			
14	18-Aug	Kinsey Irrigation Company DVT	MDEQ (Y0316)	-90.8	-2.572	3,673.0	104.012			
15	Avg for period	T&Y Irrigation District RTN (from Tongue River)	MDEQ (2003 data)	49.7	1.407	3,722.7	105.419			
16	Avg for period	Buffalo Rapids - Shirley Unit DVT	Buffalo Rapids	-114.0	-3.228	3,608.7	102.191			
17	18-Aug	Kinsey Irrigation Company RTN	MDEQ (Y0317)	3.6	0.101	3,612.3	102.292			
18	23-Aug	Buffalo Rapids - Shirley Unit RTN	MDEQ (Y0345)	16.0	0.454	3,628.3	102.746			
19	Avg for period	Powder River	USGS (06326500 adj)	89.0	2.519	3,717.3	105.266			
20	Avg for period	Buffalo Rapids - Terry Unit DVT	Buffalo Rapids	-55.9	-1.584	3,661.4	103.682			
21	Avg for period	Terry WWTP	MDEQ (no effluent)	0.0	0.000	3,661.4	103.682			
	Avg for period	Unmonitored Tributaries	MDEQ (estimated)	8.0	0.227	3,669.4	103.909			
	Avg for period	Unmonitored Waste Drains	MDEQ (estimated)	47.3	1.340	3,716.7	105.249			
	Avg for period	Evaporation	NOAA	-20.1	-0.569	3,696.6	104.680			
	Avg for period	**USGS Yellowstone River nr Terry**	**USGS (6326530)**	**3,860.0**	**109.307**	**3,696.6**	**104.680**	**163.4**	**4.627**	**4.2%**
					ADJUST	**3,860.0**	**109.31**	**GAINING REACH**		
22	Avg for period	Buffalo Rapids - Terry Unit RTN	MDEQ (visually estimated)	0.0	0.000	3,860.0	109.31			
23	26-Aug	O'Fallon Creek	MDEQ (Y0346)	2.9	0.082	3,862.9	109.39			
24	Avg for period	Buffalo Rapids - Fallon Unit DVT	Buffalo Rapids	-72.0	-2.039	3,790.9	107.35			
25	Avg for period	Buffalo Rapids - Fallon Unit RTN	MDEQ (estimated)	0.0	0.000	3,790.9	107.35			
26	Avg for period	Buffalo Rapids - Glendive Unit (I) DVT	Buffalo Rapids	-286.0	-8.099	3,504.9	99.25			
27	Avg for period	Buffalo Rapids - Glendive Unit (II) DVT	Buffalo Rapids	-40.0	-1.133	3,464.9	98.12			
	Avg for period	Unmonitored Tributaries	MDEQ (estimated)	8.6	0.242	3,473.5	98.36			
	Avg for period	Unmonitored Waste Drains	MDEQ (estimated)	62.8	1.778	3,536.2	100.14			
	Avg for period	Evaporation	NOAA	-19.1	-0.541	3,517.1	99.60			
	Avg for period	**USGS Yellowstone River at Glendive**	**USGS (06327500)**	**3,540.0**	**100.245**	**3,517.1**	**99.60**	**22.9**	**0.647**	**0.6%**
					ADJUST	**3,540.0**	**100.25**	**GAINING REACH**		

[1]Values in grey estimated, see supplementary information for estimation methods

IRRIGATED AREA SUMMARY; YELLOWSTONE RIVER

From DNRC Water Resource Surveys
Irrigated acreages checked by KFF 12-21-09

County	Date	Name	Irrigated Area[1] (acres)	Irrigated Area[1] (hectares)	Maximum Irrigated Area[2] (acres)	Maximum Irrigated Area[2] (hectares)
Rosebud	1948	Cartersville Irrigation District	9,021	3,651	10,485	4,243
Rosebud	1948	Baringer Pumping Project	939	380	1,155	467
Rosebud	1948	Private Irrigation (pumps from YR)	487	197	633	256
Custer	1948	T&Y Irrigation District (Tongue Rvr)	8,891	3,598	10,075	4,077
Custer	1948	Private Irrigation (pumps from YR)	2,379	963	3,987	1,614
Custer	1948	Kinsey Irrigation Company	6,205	2,511	6,985	2,827
Custer	1948	Buffalo Rapids - Shirley Unit	2,798	1,132	3,207	1,298
Prairie	1970	Buffalo Rapids - Shirley Unit	1,712	693	1,779	720
Prairie	1970	Buffalo Rapids - Terry Unit	3,167	1,282	3,352	1,357
Prairie	1970	Buffalo Rapids - Fallon Unit	2,974	1,204	3,060	1,238
Prairie	1970	Buffalo Rapids - Glendive Unit	1,535	621	1,576	638
Dawson	1970	Buffalo Rapids - Glendive Unit	12,693	5,137	13,626	5,514

[1]Irrigated area reported at time of water resource survey publication

[2]Maximum irrigated area used for all calculations due to date of publication

IRRIGATION SUMMARY; 8/17-26/2007 - CALIBRATION PERIOD

Major units identified from DNRC Water Resource Surveys

SI Units[1,2,3]

Name	Maximum Irrigated Area (hectares)	Withdrawl (cms)	Main Canal Return Flow (cms)	Estimated Crop ET (cms)	Main Canal Loss (cms)	Waste Drain Return Flow (cms)
Cartersville Irrigation District	4,243	5.975	1.987	2.869	1.444	1.052
Barringer Pumping Project	467	0.635	0.000	0.316		0.070
Private Irrigation (from YR)	1,870	2.543	0.164	1.264		0.435
Kinsey Irrigation Company	2,827	2.572	0.101	1.912	0.580	0.684
Buffalo Rapids - Shirley Unit	2,018	3.228	0.454	1.365	0.506	0.401
Buffalo Rapids - Terry Unit	1,357	1.584	0.000	0.918		0.255
Buffalo Rapids - Fallon Unit	1,238	2.039	0.000	0.837		0.229
Buffalo Rapids - Glendive Unit	6,152	9.232	NA	4.160		1.548

English Units[1,2,3]

Name	Maximum Irrigated Area (acres)	Withdrawl (cfs)	Main Canal Return Flow (cfs)	Waste Drain Return Flow (cfs)	Estimated Crop ET (cfs)
Cartersville Irrigation Diestrict	10,485	211.0	70.2	37.2	101.3
Barringer Pumping Project	1,155	22.4	0.0	2.5	11.2
Private Irrigation (pumps from YI	4,620	89.8	5.8	15.4	44.7
Kinsey Irrigation Company	6,985	90.8	3.6	24.2	67.5
Buffalo Rapids - Shirley Unit	4,986	114.0	16.0	14.2	48.2
Buffalo Rapids - Terry Unit	3,352	55.9	0.0	9.0	32.4
Buffalo Rapids - Fallon Unit	3,060	72.0	0.0	8.1	29.6
Buffalo Rapids - Glendive Unit	15,202	326.0	NA	54.7	146.9

[1]Maximum irrigated area from DNRC Water Resource Surveys

[2]Values in grey estimated, see supplementary information for estimation methods

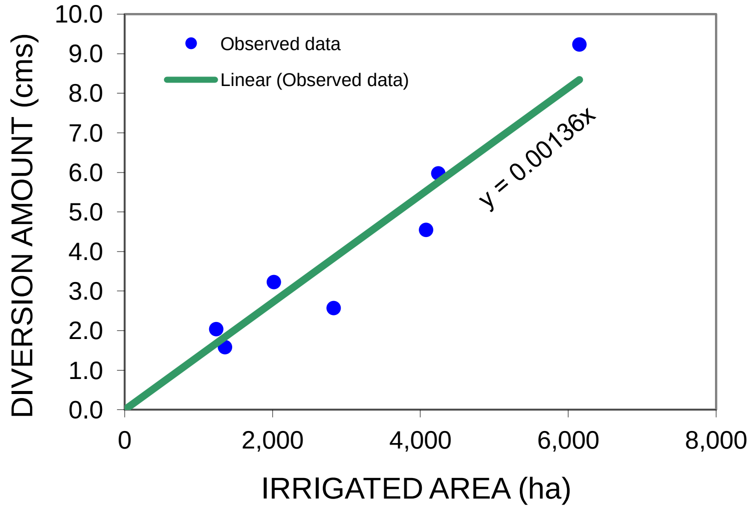
[3]From Kimberly-Penman AgriMet calculations, multiplied by irrigated area

ESTIMATED SURFACE WITHDRAWLS; 8/17-26/2007 - CALIBRATION PERIOD

Major units identified from DNRC Water Resource Surveys

Measured Location	Maximum Irrigated Area[1] (acres)	Maximum Irrigated Area[1] (hectares)	Area Cross-check (NLCD, 2001) (hectares)	Diversion Amount (cfs)	Diversion Amount (cms)
Cartersville Irrigation District	10,484	4,243	3,569	211.0	5.975
Kinsey Irrigation Company	6,986	2,827	2,672	90.8	2.572
Buffalo Rapids - Shirley Unit	4,986	2,018	1,594	114.0	3.228
Buffalo Rapids - Terry Unit	3,353	1,357	1,672	55.9	1.584
Buffalo Rapids - Fallon Unit	3,059	1,238	1,225	72.0	2.039
Buffalo Rapids - Glendive Unit	15,202	6,152	7,168	326.0	9.232
T&Y Irrigation District (Tongue Rvr)	10,074	4,077	2,762	160.6	4.547

Unmeasured Location	Maximum Irrigated Area[1] (acres)	Maximum Irrigated Area[1] (hectares)	Estimated Diversion[2] (cfs)	Estimated Diversion[2] (cms)
Barringer Pumping Project	1,155	467	22.4	0.635
Private Irrigation (pumps from YR)	4,620	1,870	89.8	2.543



[1]Identified from DNRC water resource surveys

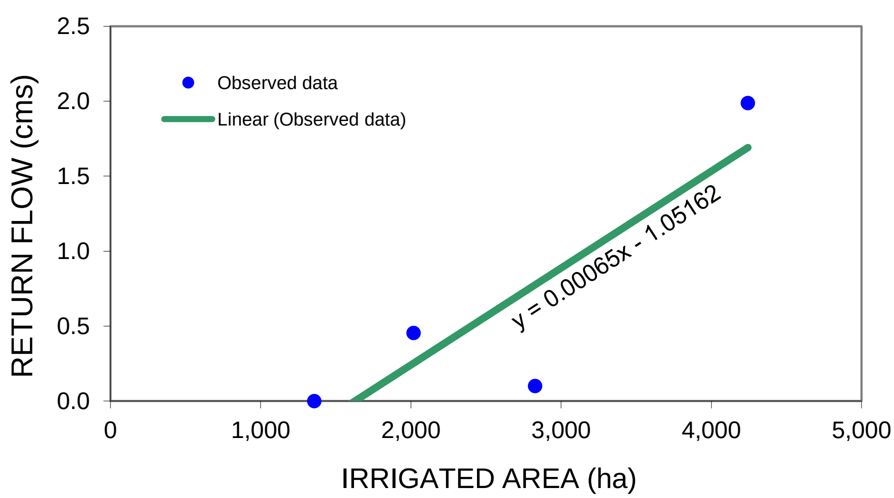
[2]Estimated using regression of irrigated area and measured diversion data

ESTIMATED CANAL RETURN FLOWS; 8/17-26/2007 - CALIBRATION PERIOD

End of canal return flow estimates - based on MDEQ measured values

Measured Location	Maximum Irrigated Area[1] (acres)	Maximum Irrigated Area[1] (hectares)	Main Canal Return Flow (cfs)	Main Canal Return Flow (cms)
Cartersville Irrigation District	10,485	4,243	70.18	1.987
Kinsey Irrigation Company	6,985	2,827	3.56	0.101
Buffalo Rapids - Shirley Unit	4,986	2,018	16.05	0.454
Buffalo Rapids - Terry Unit	3,352	1,357	0.00	0.000

Estimated Location	Maximum Irrigated Area[1] (acres)	Maximum Irrigated Area[1] (hectares)	Estimated Main Canal Return Flow[2] (cfs)	Estimated Main Canal Return Flow[2] (cms)
Barringer Pumping Project	1,155	467	0.0	0.000
Private Irrigation (pumps from YR)	4,620	1,870	5.8	0.164
Buffalo Rapids - Fallon Unit	3,060	1,238	0.0	0.000



[1]Identified from DNRC water resource surveys

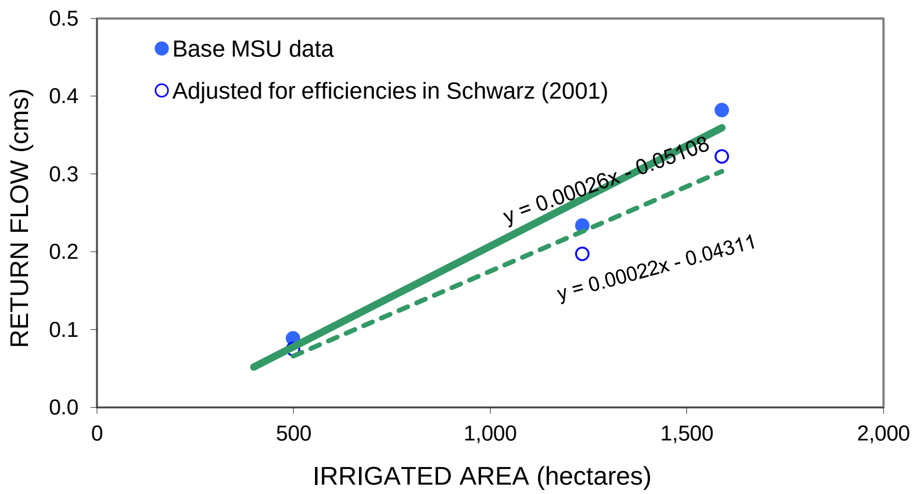
[2]Estimated using regression of irrigated area and measured return flow

ESTIMATED LATERAL WASTE DRAIN RETURN FLOW; 8/17-26/2007 - CALIBRATION PERIOD

Lateral ditch or diffuse return flow estimates - based on MDEQ measured values

Measured Location	Irrigated NLCD Area[1] (acres)	Irrigated NLCD Area[1] (hectares)	Waste Drain Return Flow (cfs)	Waste Drain Return Flow (cms)
Glendive Unit - Clear Creek Drains	3,926	1,589	13.494	0.382
Glendive Unit - Whoopup Creek Drains	1,232	499	3.133	0.089
Glendive Unit - Sand Creek Drains	3,050	1,234	8.253	0.234

Estimated Location	Maximum Irrigated Area[2] (acres)	Maximum Irrigated Area[2] (hectares)	Estimated Waste Drain Return Flow[3] (cfs)	Estimated Waste Drain Return Flow[3] (cms)
Cartersville Irrigation District	10,484	4,243	37.2	1.052
Barringer Pumping Project	1,154	467	2.5	0.070
Private Irrigation (from YR)	4,621	1,870	15.4	0.435
Kinsey Irrigation Company	6,986	2,827	24.2	0.684
Buffalo Rapids - Shirley Unit	4,986	2,018	14.2	0.401
Buffalo Rapids - Terry Unit	3,353	1,357	9.0	0.255
Buffalo Rapids - Fallon Unit	3,059	1,238	8.1	0.229
Buffalo Rapids - Glendive Unit	15,202	6,152	54.7	1.548



[1]Estimated from 2001 NLCD

[2]Identified from DNRC water resource surveys

[3]Estimated using regression of irrigated area and measured waste drain flows

a. use Glendive specific regression for efficiency in Glendive Unit (e.g. 73.7% efficient)

b. adjust to 89.3% efficient for other Buffalo Rapids Units

source: Buffalo Rapids Project (2000)

UNMONITORED TRIBUTARY SUMMARY; 8/17-26/2007 - CALIBRATION PERIOD

DEFINE UNMONITORED EXTENT USING AVAILABE USGS GAGE DATA

	Area (mi^2)	Discharge (cfs)	Area (km^2)	Discharge (cms)
Control Reach 1 - Forsyth to Miles City, MT	1,408	6.1	3,645	0.173
Control Reach 3 - Miles City to Terry, MT	1,682	8.0	4,354	0.227
Control Reach 2 - Terry to Glendive, MT	1,763	8.6	4,564	0.242

**area between Powder River and Terry			278	

**(insignificant - just use previously defined breaks in model)

UNMONITORED TRIBUTARY ESTIMATION; 8/17-26/2007 - CALIBRATION PERIOD

Use drainage area to estimate unmonitored tributary flow to Yellowstone River

CONTROL REACH 1 - USGS Forsyth to USGS Miles City, MT

Description	Gage ID	Area [1] (mi^2)	Area (km^2)	
Yellowstone River at Forsyth MT	6295000	40,146	103,933	
Yellowstone River at Miles City MT	6309000	48,253	124,921	
		8,107	20,988	
Field data representing remaining area(s)				
Rosebud Creek at mouth near Rosebud MT	6296003	1,302	3,371	
Tongue River at Miles City MT	6308500	5,397	13,972	
Unmonitored Drainage Area		1,408	3,645	
Estimated Unmonitored Tributary Flow using August regression[2]			0.173	cms
			6.1	cfs

CONTROL REACH 2 - USGS Miles City to USGS Terry, MT

Description	Gage ID	Area [1] (mi^2)	Area (km^2)	
Yellowstone River at Miles City MT	6309000	48,253	124,921	
Yellowstone River nr Terry MT	6326530	63,447	164,257	
		15,194	39,335	
Field data representing remaining area(s)				
Powder River at Mouth near Terry MT	6326520	13,512	34,981	
Unmonitored Drainage Area		1,682	4,354	
Estimated Unmonitored Tributary Flow using August regression[2]			0.227	cms
			8.0	cfs

CONTROL REACH 3 - USGS Terry to USGS Glendive, MT

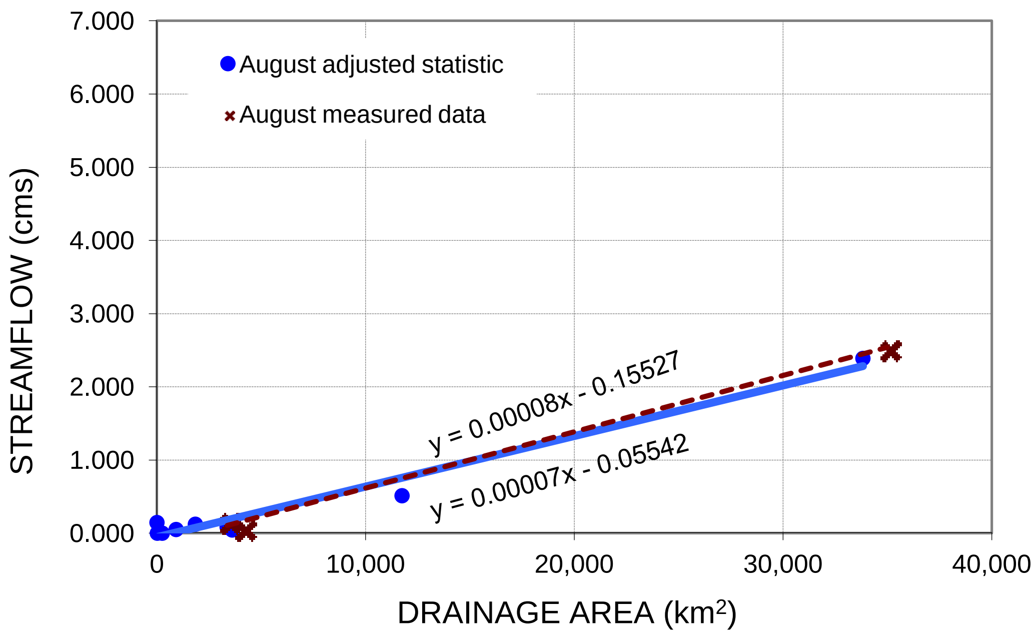
Description	Gage ID	Area [1] (mi^2)	Area (km^2)	
Yellowstone River nr Terry MT	6326530	63,447	164,257	
Yellowstone River at Glendive MT	6327500	66,788	172,906	
		3,341	8,649	
Field data representing remaining area(s)				
O'fallon Creek at mouth (1:24,000 HUC file)	NA	1,578	4,085	
Unmonitored Drainage Area		1,763	4,564	
Estimated Unmonitored Tributary Flow using August regression[2]			0.242	cms
			8.6	cfs

[1]Drainage area reported by McCarthy (2004)

[2]Based on regression of field data and drainage area; checked against August Statistic

Develop regression of drainage area vs. August measured streamflow/flow statistic for Lower Yellowstone River corridor gages

Station ID	USGS Site Name	Drainage Area (miles^2)	Drainage Area (km^2)	Aug[1] Statistic (cfs)	Aug[1] Statistic (cms)	Adj[2] Statistic (cfs)	Adj[2] Statistic (cms)
6296003	Rosebud Creek at Mouth	1,302	3,371	7.2	0.204	3.0	0.084
6296100	Snell Creek nr Hathaway	10.5	27	0.0	0.000	0.0	0.000
6308000	Tongue River near Miles City	4,539	11,751	44.1	1.249	18.1	0.512
6308500	Tongue River at Miles City MT	5,397	13,972	influenced by Tongue River Reservoir			
6309075	Sunday Creek nr Miles City	714	1,848	10.5	0.298	4.3	0.122
6326555	Cherry Creek nr Terry	358	927	4.3	0.122	1.8	0.050
6326952	Clear Creek nr Lindsay	101	261	0.0	0.000	0.0	0.000
6327000	Upper Sevenmile Creek nr Glendive	no august data		---	---	---	---
6327450	Cains Coulee at Glendive	3.7	10	12.5	0.355	5.1	0.146
6326500	Powder River near Locate	13,068	33,831	205.7	5.824	84.3	2.388
6326850	O'Fallon Creek at Mildred	1,396	3,614	3.8	0.106	1.5	0.044



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[1]Calculated statistics based on USGS data through 2007

[2]Adjusted based on ratio of field measured streamflow during 8/17-26, 2007 to Aug. statistic (≈41% of statistic)

Adjustment of August Statistic based on 2007 Streamflow data using active gages

Location		Aug-07 Streamflow	Aug Statistic	
USGS Yellowstone River at Forsyth	USGS (06295000)	3526	8150	0.432638037
USGS Yellowstone River at Miles City	USGS (06309000)	3762	8830	0.426047565
USGS Yellowstone River at Glendive	USGS (06327500)	3540	11600	0.305172414
Powder River near Locate	USGS (06326500)	102	215	0.474418605
Tongue River at Miles City MT	influenced by Tongue River Reservoir			
			AVG	41.0%

BUFFALO RAPIDS IRRIGATION SUMMARY; 8/17-23/2007 - CALIBRATION PERIOD

data provided by Dave Schwarz, Buffalo Rapids Irrigation Unit
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Hydrology\Buffalo Rapids streamflow data
data checked against original emails by KFF 12-19-2009

Shirley Unit

Date	Flow (cfs)	Flow (cms)
6/30 - 8/21	136.8	3.874
8/22 - 9/10	91.2	2.583
AVG[1]	114.0	3.228

Terry Unit

Date	Flow (cfs)	Flow (cms)
6/22 - 8/20	69.9	1.979
8/21 - 8/27	46.6	1.320
AVG[1]	55.9	1.584

Fallon Unit

Date	Flow (cfs)	Flow (cms)
7/13 - 8/29	72	2.039

Glendive Unit

Glendive Canal I

Date	Flow (cfs)	Flow (cms)
6/26 - 8/22	330	9.345
8/23 - 9/17	220	6.230
AVG[1]	286.0	8.099

Glendive Canal II

Date	Flow (cfs)	Flow (cms)
8/14 - 8/28	40	1.133
GLENDIVE I + II	326.0	9.232

[1]Weighted average of number of days in period

BUFFALO RAPIDS IRRIGATION UNIT

8/1/2007 - 9/30/2007

Shirley Unit

Date From	To	Flow (cfs)	Flow (cms)
8/1	8/21	136.8	3.874
8/22	9/10	91.2	2.582
9/11	9/18	45.6	1.291
9/19		91.2	2.582
9/20	9/30	45.6	1.291

Terry Unit

Date From	To	Flow (cfs)	Flow (cms)
8/1	8/20	69.9	1.979
8/21	8/27	46.6	1.320
8/31		69.9	1.979
9/1	9/9	46.6	1.320
9/10	9/18	23.3	0.660

Fallon Unit

Date From	To	Flow (cfs)	Flow (cms)
8/1	8/29	72	2.039
8/30	9/18	48	1.359

Glendive Unit

Glendive Canal I

Date From	To	Flow (cfs)	Flow (cms)
8/1	8/22	330	9.345
8/23	9/17	220	6.230
9/18	9/26	110	3.115

Glendive Canal II

Date From	To	Flow (cfs)	Flow (cms)
8/1	8/13	80	2.265
8/14	8/28	40	1.133

Data provided by Dave Schwartz (Buffalo Rapids Irrigation Unit)

[1]Weighted average of number of days in period

CITY DATA SUMMARY; 8/17-26/2007 - CALIBRATION PERIOD

data provided by Pat Zent - Forsyth and Allen Kelm - Miles City
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\City streamflow data
data checked against original electronic file and emails by KFF 12-17-2009

Forsyth Water Treatement Plant[1]

Date	Flow (mgd)	Flow (cfs)	Flow (cms)
8/17/2007	0.503	0.8	0.022
8/18/2007	0.374	0.6	0.016
8/19/2007	0.489	0.8	0.021
8/20/2007	0.548	0.8	0.024
8/21/2007	0.620	1.0	0.027
8/22/2007	0.481	0.7	0.021
8/23/2007	0.484	0.7	0.021
8/24/2007	0.479	0.7	0.021
8/25/2007	0.394	0.6	0.017
8/26/2007	0.575	0.9	0.025
AVG	0.495	0.8	0.022

Forsyth Wastewater Treatement Plant[2]

Date	Flow (mgd)	Flow (cfs)	Flow (cms)
8/17/2007	0.3034	0.5	0.013
8/18/2007	0.2218	0.3	0.010
8/19/2007	0.2400	0.4	0.011
8/20/2007	0.2784	0.4	0.012
8/21/2007	0.2530	0.4	0.011
8/22/2007	0.2587	0.4	0.011
8/23/2007	0.3445	0.5	0.015
8/24/2007	0.1936	0.3	0.008
8/25/2007	0.2441	0.4	0.011
8/26/2007	0.2557	0.4	0.011
AVG	0.259	0.4	0.011

Miles City Water Treatement Plant[1]

Date	Flow (mgd)	Flow (cfs)	Flow (cms)
8/17/2007	2.38	3.7	0.104
8/18/2007	2.15	3.3	0.094
8/19/2007	2.09	3.2	0.092
8/20/2007	2.70	4.2	0.118
8/21/2007	2.64	4.1	0.116
8/22/2007	2.25	3.5	0.099
8/23/2007	2.24	3.5	0.098
8/24/2007	2.10	3.2	0.092
8/25/2007	2.05	3.2	0.090
8/26/2007	2.62	4.1	0.115
AVG	2.32	3.6	0.102

Miles City Wastewater Treatement Plant[2]

Date	Flow (mgd)	Flow (cfs)	Flow (cms)
8/17/2007	1.21	1.9	0.053
8/18/2007	1.17	1.8	0.051
8/19/2007	1.14	1.8	0.050
8/20/2007	1.22	1.9	0.053
8/21/2007	1.23	1.9	0.054
8/22/2007	1.17	1.8	0.051
8/23/2007	1.16	1.8	0.051
8/24/2007	1.15	1.8	0.050
8/25/2007	1.17	1.8	0.051
8/26/2007	1.17	1.8	0.051
AVG	1.18	1.8	0.052

Glendive Water Treatement Plant

Not Required
Outflows DS of study reach

Glendive Water Treatement Plant

Not Required
Outflows DS of study reach

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[1]From monthly reporty of finished clearwell effluent
[2]Provided by City of Forsyth and Miles City

DNRC STREAMFLOW SUMMARY; 8/17-26/2007 - CALIBRATION PERIOD

data provided by Larry Dolan, DNRC
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\DNRC streamflow data
T&Y diversion on Tongue River (no data collected in 2007); use data from 2005 (similar streamflow year)

Date	Flow (cfs)	Flow (cms)
8/17/2005	166.5	4.714
8/18/2005	166.5	4.715
8/19/2005	154.8	4.384
8/20/2005	152.8	4.328
8/21/2005	152.6	4.322
8/22/2005	158.9	4.499
8/23/2005	163.0	4.616
8/24/2005	164.5	4.659
8/25/2005	163.5	4.630
8/26/2005	162.5	4.601
AVG	160.6	4.547

DETERMINE DIFFERENCE BETWEEN USGS AND DNRC

Miles city

8/17/2005	166	A
8/18/2005	179	A
8/19/2005	203	A
8/20/2005	212	A
8/21/2005	213	A
8/22/2005	189	A
8/23/2005	207	A
8/24/2005	157	A
8/25/2005	144	A
8/26/2005	139	A
AVG	180.9	

Above T&Y

8/17/2005	336	A
8/18/2005	330	A
8/19/2005	328	A
8/20/2005	324	A
8/21/2005	321	A
8/22/2005	305	A
8/23/2005	297	A
8/24/2005	289	A
8/25/2005	283	A
8/26/2005	279	A
AVG	309.2	

T&Y-Miles	Actual Diversion
170.0	166.5
151.0	166.5
125.0	154.8
112.0	152.8
108.0	152.6
116.0	158.9
90.0	163.0
132.0	164.5
139.0	163.5
140.0	162.5
128.3	0.0

MDEQ DISCHARGE MEASUREMENT SUMMARY; 8/17-26/2007 - CALIBRATION PERIOD

data from MDEQ field measurements 8/17-26, 2007
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\MTDEQ streamflow data
data checked against original field sheets by KFF 12-21-2009

Location	Date	Site Visit Code	Discharge (cfs)	Crew
Cartersville Canal DVT	8/17/2007	Y0314	211.0	A. Welch, J. Drygas
Tongue River	8/17/2007	Y0315	135.0	A. Welch, J. Drygas
Kinsey Main Canal DVT	8/18/2007	Y0316	90.8	A. Welch, J. Drygas
Kinsey Main Canal RTN	8/18/2007	Y0317	3.6	A. Welch, J. Drygas
Powder River	8/18/2007	Y0318	109.2	A. Welch, J. Drygas
Forsyth WWTP	8/20/2007	Y0325	0.2	A. Welch, J. Drygas
Rosebud Creek	8/18/2007	Y0333	6.4	M. Stermitz, M. Suplee
Cartersville Canal RTN	8/20/2007	Y0337	70.2	M. Suplee, M. Stermitz
Shirley Main Canal RTN	8/23/2007	Y0345	16.0	M. Stermitz, M. Suplee
O'Fallon Creek	8/26/2007	Y0346	2.9	R. Sada, M. Suplee

MSU STREAMFLOW SUMMARY; 8/17-26/2007 - CALIBRATION PERIOD

data provided by Holly Sessoms, MSU
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\MSU streamflow data
diffuse irrigation returns (lateral canals off main canal)

	Irrigation Waste Drain Location and Streamflow (cfs)							
Date	Clear Cr #1	Clear Cr #2	Clear Cr #3	Whoopup Cr #1	Whoopup Cr #2	Sand Cr	Sand Cr #2	Sand Cr #3
8/17/2007	5.13	5.11	0.94	1.71	1.73	2.58	0.72	3.46
8/18/2007	5.26	5.04	1.37	1.92	1.17	3.38	0.66	2.99
8/19/2007	5.45	4.81	1.13	1.85	1.09	2.38	0.71	2.78
8/20/2007	5.20	3.90	1.60	1.73	1.17	2.57	0.75	2.94
8/21/2007	5.76	4.81	2.62	1.89	1.16	3.16	0.80	2.99
8/22/2007	6.06	4.87	3.06	1.93	1.19	3.30	0.74	2.94
8/23/2007	5.85	5.38	4.12	1.96	1.22	3.33	0.69	2.88
8/24/2007	6.33	5.10	4.24	1.88	1.36	5.69	0.42	3.89
8/25/2007	6.75	4.53	4.72	2.06	1.31	9.91	0.53	3.50
8/26/2007	6.49	4.35	4.98	1.62	1.38	7.82	0.58	3.43
AVG	5.83	4.79	2.88	1.85	1.28	4.41	0.66	3.18

USGS MEAN DAILY STREAMFLOW SUMMARY; 8/17-26/2007 - CALIBRATION PERIOD

data downloaded from USGS NWIS 10/09 & 10/14 2009
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\USGS streamflow data
data checked against original dv download by KFF 12-16-2009

CALIBRATION - FORSYTH

8/17/2007	3,470	A
8/18/2007	3,510	A
8/19/2007	3,540	A
8/20/2007	3,610	A
8/21/2007	3,620	A
8/22/2007	3,640	A
8/23/2007	3,500	A
8/24/2007	3,410	A
8/25/2007	3,470	A
8/26/2007	3,490	A
AVG	3,526.0	

CALIBRATION - MILES CITY

8/17/2007	3,540	A
8/18/2007	3,610	A
8/19/2007	3,680	A
8/20/2007	3,760	A
8/21/2007	3,810	A
8/22/2007	3,850	A
8/23/2007	3,860	A
8/24/2007	3,800	A
8/25/2007	3,890	A
8/26/2007	3,820	A
AVG	3,762.0	

CALIBRATION - GLENDIVE

8/17/2007	3,330	A
8/18/2007	3,290	A
8/19/2007	3,430	A
8/20/2007	3,470	A
8/21/2007	3,480	A
8/22/2007	3,530	A
8/23/2007	3,600	A
8/24/2007	3,760	A
8/25/2007	3,710	A
8/26/2007	3,800	A
AVG	3,540.0	

CALIBRATION - TONGUE RIVER

8/17/2007	100	A
8/18/2007	150	A
8/19/2007	200	A
8/20/2007	200	A
8/21/2007	210	A
8/22/2007	202	A
8/23/2007	191	A
8/24/2007	209	A
8/25/2007	225	A
8/26/2007	159	A
AVG	184.6	

CALIBRATION - POWDER RIVER

8/17/2007	119	A
8/18/2007	122	A
8/19/2007	110	A
8/20/2007	103	A
8/21/2007	103	A
8/22/2007	95	A
8/23/2007	87	A
8/24/2007	85	A
8/25/2007	93	A
8/26/2007	98	A
AVG	102	
ADJUSTMENT[1]	89.0	

[1]Adjusted to value at mouth - see suplement on adjustment method

YELLOWSTONE RIVER WATER BALANCE; 9/11-20/2007 - VALIDATION PERIOD

Map ID	Monitoring Date(s)	Site Name	Data Src (Site Code)	Flow (cfs)[1]	Flow (cms)[1]	Balance (cfs)	Balance (cms)	Groundwater Accretion (cfs)	Groundwater Accretion (cms)	Percentage of Surface Q (%)
1	**Avg for period**	**USGS Yellowstone River at Forsyth**	**USGS (06295000)**	**4,052.0**	**114.744**					
2	Avg for period	Forsyth WTP	MDEQ (from city)	-0.6	-0.017	4,051.4	114.727			
3	10-Sep	Cartersville Irrigation District DVT	MDEQ (Y0314)	-89.0	-2.519	3,962.4	112.208			
4	Avg for period	Forsyth WWTP	City of Forsyth	0.4	0.011	3,962.8	112.219			
5	12-Sep	Rosebud Creek	MDEQ (Y0333)	4.3	0.122	3,967.1	112.341			
6	15-Sep	Cartersville Irrigation District RTN	MDEQ (Y0337)	47.0	1.330	4,014.1	113.671			
7	Avg for period	Baringer Pumping Project DVT	MDEQ (estimated)	-12.5	-0.355	4,001.6	113.316			
8	Avg for period	Baringer Pumping Project RTN	MDEQ (estimated)	0.0	0.000	4,001.6	113.316			
9	Avg for period	Private Irrigation (pumps from YR) DVT	MDEQ (estimated)	-50.2	-1.421	3,951.4	111.895			
10	Avg for period	Private Irrigation (pumps from YR) RTN	MDEQ (estimated)	11.0	0.311	3,962.4	112.206			
11	Avg for period	Miles City WTP	MDEQ (from city)	-3.1	-0.089	3,959.2	112.117			
12	Avg for period	Tongue River	USGS (06308500)	213.4	6.043	4,172.6	118.160			
13	Avg for period	Unmonitored Tributaries	MDEQ (estimated)	7.5	0.212	4,180.1	118.372			
14	Avg for period	Unmonitored Waste Drains	MDEQ (estimated)	57.1	1.617	4,237.2	119.989			
	Avg for period	Evaporation	NOAA	-16.4	-0.463	4,220.9	119.526			
15	**Avg for period**	**USGS Yellowstone River at Miles City**	**USGS (06309000)**	**4,343.0**	**122.985**	**4,220.9**	**119.526**	**122.1**	**3.459**	**2.9%**
					ADJUST	**4,343.0**	**122.985**	**GAINING REACH**		
16	Avg for period	Miles City WWTP	City of Miles City	1.7	0.048	4,344.7	123.033			
17	11-Sep	Kinsey Irrigation Company DVT	MDEQ (Y0316)	-93.6	-2.650	4,251.1	120.383			
18	Avg for period	T&Y Irrigation District RTN (from Tongue River)	MDEQ (2003 data)	36.7	1.039	4,287.8	121.423			
19	Avg for period	Buffalo Rapids - Shirley Unit DVT	Buffalo Rapids	-50.2	-1.420	4,237.7	120.002			
20	11-Sep	Kinsey Irrigation Company RTN	MDEQ (Y0317)	28.2	0.797	4,265.8	120.799			
21	16-Sep	Buffalo Rapids - Shirley Unit RTN	MDEQ (Y0345)	15.5	0.440	4,281.4	121.240			
22	Avg for period	Powder River	USGS (06326500 adj)	77.9	2.206	4,359.3	123.445			
23	Avg for period	Buffalo Rapids - Terry Unit DVT	Buffalo Rapids	-18.6	-0.528	4,340.6	122.917			
24	Avg for period	Terry WWTP	MDEQ (Y0361)	0.1	0.004	4,340.8	122.921			
25	Avg for period	Unmonitored Tributaries	MDEQ (estimated)	9.5	0.268	4,350.2	123.190			
26	Avg for period	Unmonitored Waste Drains	MDEQ (estimated)	50.0	1.415	4,400.2	124.605			
	Avg for period	Evaporation	NOAA	-15.5	-0.440	4,384.7	124.165			
	Avg for period	**USGS Yellowstone River nr Terry**	**USGS (6326530)**	**4,490.0**	**127.147**	**4,384.7**	**124.165**	**105.3**	**2.983**	**2.4%**
					ADJUST	**4,490.0**	**127.147**	**GAINING REACH**		
27	Avg for period	Buffalo Rapids - Terry Unit RTN	MDEQ (estimated)	0.0	0.000	4,490.0	127.147			
28	26-Aug	O'Fallon Creek	MDEQ (Y0346)	5.9	0.166	4,495.9	127.314			
29	Avg for period	Buffalo Rapids - Fallon Unit DVT	Buffalo Rapids	-48.0	-1.359	4,447.9	125.954			
30	Avg for period	Buffalo Rapids - Fallon Unit RTN	MDEQ (estimated)	0.9	0.027	4,448.8	125.981			
31	Avg for period	Buffalo Rapids - Glendive Unit (I) DVT	Buffalo Rapids	-187.0	-5.295	4,261.8	120.686			
32	Avg for period	Buffalo Rapids - Glendive Unit (II) DVT	Buffalo Rapids	0.0	0.000	4,261.8	120.686			
33	Avg for period	Unmonitored Tributaries	MDEQ (estimated)	10.1	0.285	4,271.9	120.971			
34	Avg for period	Unmonitored Waste Drains	MDEQ (estimated)	59.8	1.693	4,331.7	122.664			
	Avg for period	Evaporation	NOAA	-14.7	-0.415	4,317.0	122.249			
35	**Avg for period**	**USGS Yellowstone River at Glendive**	**USGS (06327500)**	**4,763.0**	**134.878**	**4,317.0**	**122.249**	**446.0**	**12.629**	**9.9%**
					ADJUST	**4,763.0**	**134.878**	**GAINING REACH**		

[1]Values in grey estimated, see supplementary information for estimation methods

IRRIGATED AREA SUMMARY; YELLOWSTONE RIVER

From DNRC Water Resource Surveys
Irrigated acreages checked by KFF 12-21-09

County	Date	Name	Irrigated Area[1] (acres)	Irrigated Area[1] (hectares)	Maximum Irrigated Area[2] (acres)	Maximum Irrigated Area[2] (hectares)
Rosebud	1948	Cartersville Irrigation District	9,021	3,651	10,485	4,243
Rosebud	1948	Baringer Pumping Project	939	380	1,155	467
Rosebud	1948	Private Irrigation (pumps from YR)	487	197	633	256
Custer	1948	T&Y Irrigation District (Tongue Rvr)	8,891	3,598	10,075	4,077
Custer	1948	Private Irrigation (pumps from YR)	2,379	963	3,987	1,614
Custer	1948	Kinsey Irrigation Company	6,205	2,511	6,985	2,827
Custer	1948	Buffalo Rapids - Shirley Unit	2,798	1,132	3,207	1,298
Prairie	1970	Buffalo Rapids - Shirley Unit	1,712	693	1,779	720
Prairie	1970	Buffalo Rapids - Terry Unit	3,167	1,282	3,352	1,357
Prairie	1970	Buffalo Rapids - Fallon Unit	2,974	1,204	3,060	1,238
Prairie	1970	Buffalo Rapids - Glendive Unit	1,535	621	1,576	638
Dawson	1970	Buffalo Rapids - Glendive Unit	12,693	5,137	13,626	5,514

[1]Irrigated area reported at time of water resource survey publication

[2]Maximum irrigated area used for all calculations due to date of publication

IRRIGATION SUMMARY; 9/11-20/2007 - VALIDATION PERIOD

Major units identified from DNRC Water Resource Surveys

SI Units[1,2,3]

Name	Maximum Irrigated Area (hectares)	Withdrawl (cms)	Main Canal Return Flow (cms)	Waste Drain Return Flow (cms)	Estimated Crop ET (cms)
Cartersville Irrigation District	4,243	2.519	1.330	0.990	1.247
Barringer Pumping Project	467	0.355	0.000	0.159	0.137
Private Irrigation (pumps from YR)	1,870	1.421	0.311	0.468	0.550
Kinsey Irrigation Company	2,827	2.650	0.797	0.678	0.831
Buffalo Rapids - Shirley Unit	2,018	1.420	0.440	0.431	0.593
Buffalo Rapids - Terry Unit	1,357	0.528	0.000	0.306	0.399
Buffalo Rapids - Fallon Unit	1,238	1.359	0.027	0.283	0.364
Buffalo Rapids - Glendive Unit	6,152	5.295	NA	1.410	1.809

English Units[1,2,3]

Name	Maximum Irrigated Area (acres)	Withdrawl (cfs)	Main Canal Return Flow (cfs)	Waste Drain Return Flow (cfs)	Estimated Crop ET (cfs)
Cartersville Irrigation Diestrict	10,485	89.0	47.0	35.0	44.0
Barringer Pumping Project	1,155	12.5	0.0	5.6	4.8
Private Irrigation (pumps from YR)	4,620	50.2	11.0	16.5	19.4
Kinsey Irrigation Company	6,985	93.6	28.2	24.0	29.3
Buffalo Rapids - Shirley Unit	4,986	50.2	15.5	15.2	21.0
Buffalo Rapids - Terry Unit	3,352	18.6	0.0	10.8	14.1
Buffalo Rapids - Fallon Unit	3,060	48.0	0.9	10.0	12.9
Buffalo Rapids - Glendive Unit	15,202	187.0	NA	49.8	63.9

[1]Maximum irrigated area from DNRC Water Resource Surveys

[2]Values in grey estimated, see supplementary information for estimation methods

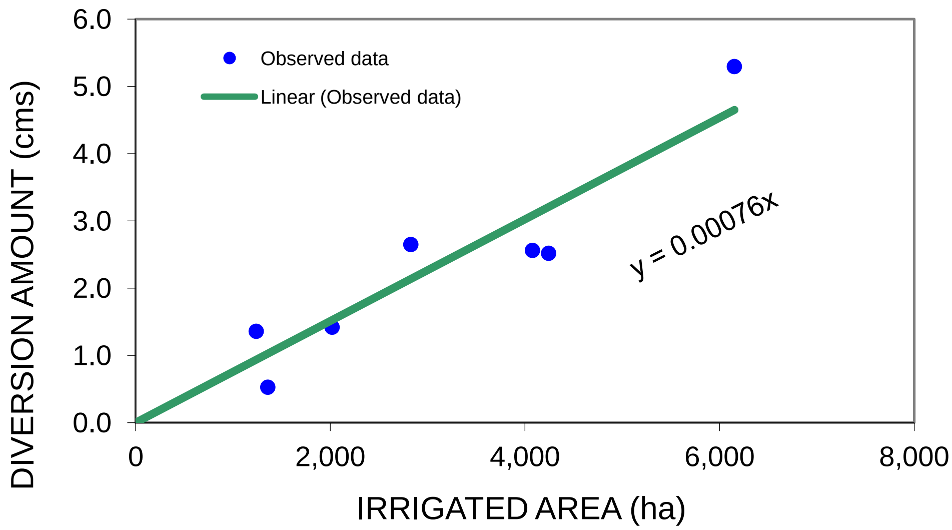
[3]From Kimberly-Penman AgriMet calculations, multiplied by irrigated area

ESTIMATED SURFACE WITHDRAWLS; 9/11-20/2007 - VALIDATION PERIOD

Major units identified from DNRC Water Resource Surveys

Measured Location	Maximum Irrigated Area[1] (acres)	Maximum Irrigated Area[1] (hectares)	Area Cross-check (NLCD, 2001) (hectares)	Diversion Amount (cfs)	Diversion Amount (cms)
Cartersville Irrigation District	10,484	4,243	3,569	89.0	2.519
Kinsey Irrigation Company	6,986	2,827	2,672	93.6	2.650
Buffalo Rapids - Shirley Unit	4,986	2,018	1,594	50.2	1.420
Buffalo Rapids - Terry Unit	3,353	1,357	1,672	18.6	0.528
Buffalo Rapids - Fallon Unit	3,059	1,238	1,225	48.0	1.359
Buffalo Rapids - Glendive Unit	15,202	6,152	7,168	187.0	5.295
T&Y Irrigation District (Tongue Rvr)	10,074	4,077	2,762	90.5	2.562

Unmeasured Location	Maximum Irrigated Area[1] (acres)	Maximum Irrigated Area[1] (hectares)	Estimated Diversion[2] (cfs)	Estimated Diversion[2] (cms)
Barringer Pumping Project	1,155	467	12.5	0.355
Private Irrigation (pumps from YR)	4,620	1,870	50.2	1.421



[1]Identified from DNRC water resource surveys

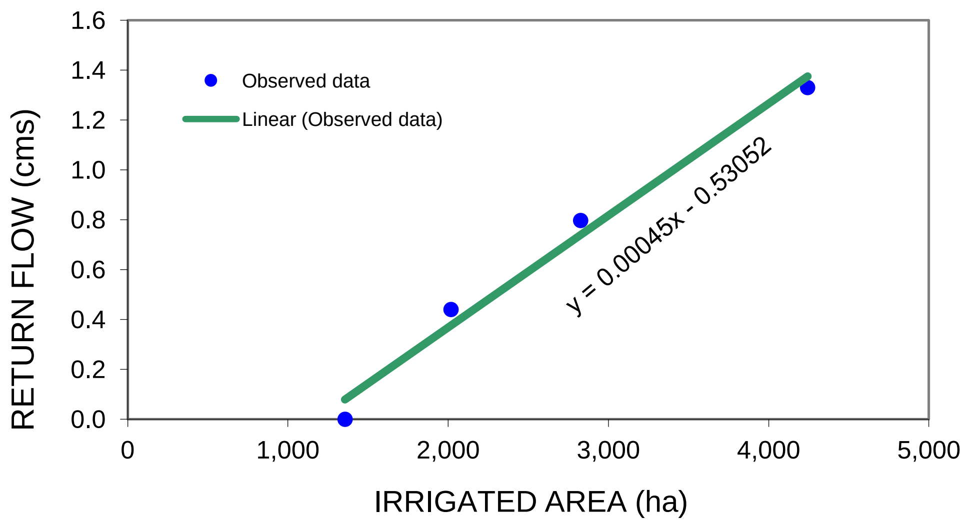
[2]Estimated using regression of irrigated area and measured diversion data

ESTIMATED CANAL RETURN FLOWS; 9/11-20/2007 - VALIDATION PERIOD

End of canal return flow estimates - based on MDEQ measured values

Measured Location	Maximum Irrigated Area[1] (acres)	Maximum Irrigated Area[1] (hectares)	Main Canal Return Flow (cfs)	Main Canal Return Flow (cms)
Cartersville Irrigation District	10,485	4,243	46.97	1.330
Kinsey Irrigation Company	6,985	2,827	28.15	0.797
Buffalo Rapids - Shirley Unit	4,986	2,018	15.55	0.440
Buffalo Rapids - Terry Unit	3,352	1,357	0.00	0.000

Estimated Location	Maximum Irrigated Area[1] (acres)	Maximum Irrigated Area[1] (hectares)	Estimated Main Canal Return Flow[2] (cfs)	Estimated Main Canal Return Flow[2] (cms)
Barringer Pumping Project	1,155	467	0.0	0.000
Private Irrigation (pumps from YR)	4,620	1,870	11.0	0.311
Buffalo Rapids - Fallon Unit	3,060	1,238	0.9	0.027



[1]Identified from DNRC water resource surveys

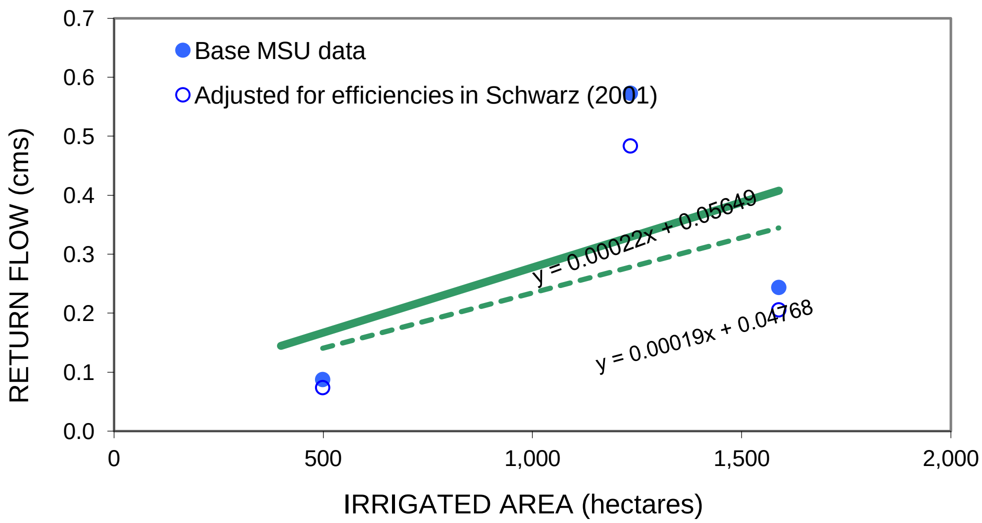
[2]Estimated using regression of irrigated area and measured return flow

ESTIMATED LATERAL WASTE DRAIN RETURN FLOW; 9/11-20/2007 - VALIDATION PERIOD

Lateral ditch or diffuse return flow estimates - based on MDEQ measured values

Measured Location	Irrigated NLCD Area[1] (acres)	Irrigated NLCD Area[1] (hectares)	Waste Drain Return Flow (cfs)	Waste Drain Return Flow (cms)
Glendive Unit - Clear Creek Drains	3,926	1,589	8.607	0.244
Glendive Unit - Whoopup Creek Drains	1,232	499	3.092	0.088
Glendive Unit - Sand Creek Drains	3,050	1,234	20.232	0.573

Estimated Location	Maximum Irrigated Area[2] (acres)	Maximum Irrigated Area[2] (hectares)	Estimated Waste Drain Return Flow[3] (cfs)	Estimated Waste Drain Return Flow[3] (cms)
Cartersville Irrigation District	10,484	4,243	35.0	0.990
Barringer Pumping Project	1,154	467	5.6	0.159
Private Irrigation (pumps from YR)	4,621	1,870	16.5	0.468
Kinsey Irrigation Company	6,986	2,827	24.0	0.678
Buffalo Rapids - Shirley Unit	4,986	2,018	15.2	0.431
Buffalo Rapids - Terry Unit	3,353	1,357	10.8	0.306
Buffalo Rapids - Fallon Unit	3,059	1,238	10.0	0.283
Buffalo Rapids - Glendive Unit	15,202	6,152	49.8	1.410



[1]Estimated from 2001 NLCD

[2]Identified from DNRC water resource surveys

[3]Estimated using regression of irrigated area and measured waste drain flows

a. use Glendive specific regression for efficiency in Glendive Unit (e.g. 73.7% efficient)

b. adjust to 89.3% efficient for other Buffalo Rapids Units

source: Buffalo Rapids Project (2000)

UNMONITORED TRIBUTARY SUMMARY; 9/11-20/2007 - CALIBRATION PERIOD

DEFINE UNMONITORED EXTENT USING AVAILABE USGS GAGE DATA

	Area (mi^2)	Discharge (cfs)	Area (km^2)	Discharge (cms)
Control Reach 1 - Forsyth to Miles City, MT	1,408	7.5	3,645	0.212
Control Reach 3 - Miles City to Terry, MT	1,682	9.5	4,354	0.268
Control Reach 2 - Terry to Glendive, MT	1,763	10.1	4,564	0.285
**area between Powder River and Terry			278	

**(insignificant - just use previously defined breaks in model)

UNMONITORED TRIBUTARY ESTIMATION; 9/11-20/2007 - VALIDATION PERIOD

Use drainage area to estimate unmonitored tributary flow to Yellowstone River

CONTROL REACH 1 - USGS Forsyth to USGS Miles City, MT

Description	**Gage ID**	**Area [1] (mi^2)**	**Area (km^2)**	
Yellowstone River at Forsyth MT	6295000	40,146	103,933	
Yellowstone River at Miles City MT	6309000	48,253	124,921	
		8,107	20,988	
Field data representing remaining area(s)				
Rosebud Creek at mouth near Rosebud MT	6296003	1,302	3,371	
Tongue River at Miles City MT	6308500	5,397	13,972	
Unmonitored Drainage Area		1,408	3,645	
Estimated Unmonitored Tributary Flow using August regression[2]			0.212	cms
			7.5	cfs

CONTROL REACH 2 - USGS Miles City to USGS Terry, MT

Description	**Gage ID**	**Area [1] (mi^2)**	**Area (km^2)**	
Yellowstone River at Miles City MT	6309000	48,253	124,921	
Yellowstone River nr Terry MT	6326530	63,447	164,257	
		15,194	39,335	
Field data representing remaining area(s)				
Powder River at Mouth near Terry MT	6326520	13,512	34,981	
Unmonitored Drainage Area		1,682	4,354	
Estimated Unmonitored Tributary Flow using August regression[2]			0.268	cms
			9.5	cfs

CONTROL REACH 3 - USGS Terry to USGS Glendive, MT

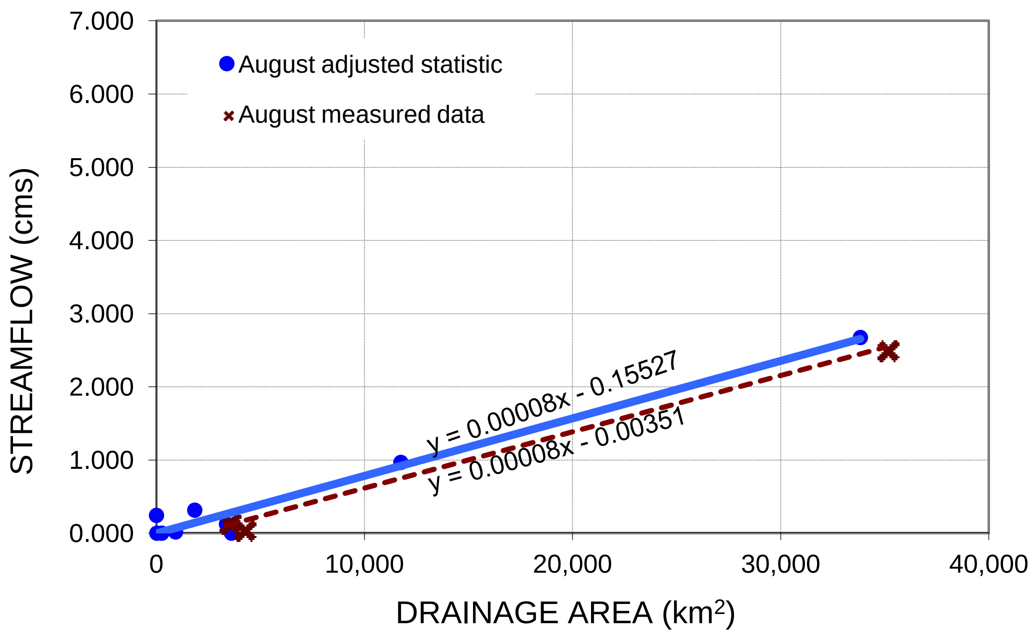
Description	**Gage ID**	**Area [1] (mi^2)**	**Area (km^2)**	
Yellowstone River nr Terry MT	6326530	63,447	164,257	
Yellowstone River at Glendive MT	6327500	66,788	172,906	
		3,341	8,649	
Field data representing remaining area(s)				
O'fallon Creek at mouth (1:24,000 HUC file)	NA	1,578	4,085	
Unmonitored Drainage Area		1,763	4,564	
Estimated Unmonitored Tributary Flow using August regression[2]			0.285	cms
			10.1	cfs

[1]Drainage area reported by McCarthy (2004)

[2]Based on regression of field data and drainage area; checked against August Statistic

Develop regression of drainage area vs. August measured streamflow/flow statistic for Lower Yellowstone River corridor gages

Station ID	USGS Site Name	Drainage Area (miles²)	Drainage Area (km²)	Sept[1] Statistic (cfs)	Sept[1] Statistic (cms)	Adj[2] Statistic (cfs)	Adj[2] Statistic (cms)
6296003	Rosebud Creek at Mouth	1,302	3,371	7.5	0.214	4.4	0.125
6296100	Snell Creek nr Hathaway	10.5	27	0.0	0.000	0.0	0.000
6308000	Tongue River near Miles City	4,539	11,751	58.6	1.659	34.1	0.967
6308500	Tongue River at Miles City MT	5,397	13,972	influenced by Tongue River Reservoir			
6309075	Sunday Creek nr Miles City	714	1,848	19.0	0.539	11.1	0.314
6326555	Cherry Creek nr Terry	358	927	1.0	0.029	0.6	0.017
6326952	Clear Creek nr Lindsay	101	261	0.0	0.000	0.0	0.000
6327000	Upper Sevenmile Creek nr Glendive	no sept data		---	---	---	---
6327450	Cains Coulee at Glendive	3.7	10	14.7	0.417	8.6	0.243
6326500	Powder River near Locate	13,068	33,831	162.0	4.586	94.4	2.674
6326850	O'Fallon Creek at Mildred	1,396	3,614	0.0	0.001	0.0	0.000



[1]Calculated statistics based on USGS data through 2007

[2]Adjusted based on ratio of field measured streamflow during 8/17-26, 2007 to Aug. statistic (≈58% of statistic)

Adjustment of August Statistic based on 2007 Streamflow data using active gages

Location		Sep-07 Streamflow	Sept Statistic	
USGS Yellowstone River at Forsyth	USGS (06295000)	4052	6960	0.582183908
USGS Yellowstone River at Miles City	USGS (06309000)	4343	7800	0.556794872
USGS Yellowstone River at Glendive	USGS (06327500)	4763	7840	0.60752551
Powder River near Locate	USGS (06326500)	102	174	0.586206897
Tongue River at Miles City MT	influenced by Tongue River Reservoir			
			AVG	58.3%

BUFFALO RAPIDS IRRIGATION SUMMARY; 9/11-20/2007 - VALIDATION PERIOD

data provided by Dave Schwarz, Buffalo Rapids Irrigation Unit
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Hydrology\Buffalo Rapids streamflow data
data checked against original emails by KFF 12-19-2009

Shirley Unit

Date	Flow (cfs)	Flow (cms)
9/11 - 9/18	45.6	1.291
9/19	91.2	2.583
9/20-9/30	45.6	1.291
AVG[1]	50.2	1.420

Terry Unit

Date	Flow (cfs)	Flow (cms)
9/10-9/18	23.3	0.660
9/18-9/20	0.0	0.000
AVG[1]	18.6	0.528

Fallon Unit

Date	Flow (cfs)	Flow (cms)
8/30 - 9/18	48	1.359
9/18-9/20	0.0	0.000
AVG[1]	38.4	1.087

Glendive Unit

Glendive Canal I

Date	Flow (cfs)	Flow (cms)
8/23 - 9/17	220	6.230
9/18 - 9/26	110	3.115
AVG[1]	187.0	5.295

Glendive Canal II

Date	Flow (cfs)	Flow (cms)
9/11 - 9/20	0	0.000
GLENDIVE I + II	187.0	5.295

[1]Weighted average of number of days in period

BUFFALO RAPIDS IRRIGATION UNIT
8/1/2007 - 9/30/2007

Shirley Unit

Date			
From	To	Flow (cfs)	Flow (cms)
8/1	8/21	136.8	3.874
8/22	9/10	91.2	2.582
9/11	9/18	45.6	1.291
9/19		91.2	2.582
9/20	9/30	45.6	1.291

Terry Unit

Date			
From	To	Flow (cfs)	Flow (cms)
8/1	8/20	69.9	1.979
8/21	8/27	46.6	1.320
8/31		69.9	1.979
9/1	9/9	46.6	1.320
9/10	9/18	23.3	0.660

Fallon Unit

Date			
From	To	Flow (cfs)	Flow (cms)
8/1	8/29	72	2.039
8/30	9/18	48	1.359

Glendive Unit
Glendive Canal I

Date			
From	To	Flow (cfs)	Flow (cms)
8/1	8/22	330	9.345
8/23	9/17	220	6.230
9/18	9/26	110	3.115

Glendive Canal II

Date			
From	To	Flow (cfs)	Flow (cms)
8/1	8/13	80	2.265
8/14	8/28	40	1.133

Data provided by Dave Schwartz (Buffalo Rapids Irrigation Unit)
[1]Weighted average of number of days in period

CITY DATA SUMMARY; 9/11-20/2007 - VALIDATION PERIOD

data provided by Pat Zent - Forsyth and Allen Kelm - Miles City
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\City streamflow data
data checked against original electronic file and emails by KFF 3-26-2010

Forsyth Water Treatement Plant[1]

Date	Flow (mgd)	Flow (cfs)	Flow (cms)
9/11/2007	0.38	0.6	0.017
9/12/2007	0.39	0.6	0.017
9/13/2007	0.40	0.6	0.017
9/14/2007	0.44	0.7	0.019
9/15/2007	0.40	0.6	0.017
9/16/2007	0.30	0.5	0.013
9/17/2007	0.34	0.5	0.015
9/18/2007	0.43	0.7	0.019
9/19/2007	0.45	0.7	0.020
9/20/2007	0.39	0.6	0.017
AVG	0.393	0.6	0.017

Forsyth Wastewater Treatement Plant[2]

Date	Flow (mgd)	Flow (cfs)	Flow (cms)
9/11/2007	0.2461	0.4	0.011
9/12/2007	0.2531	0.4	0.011
9/13/2007	0.2564	0.4	0.011
9/14/2007	0.2853	0.4	0.013
9/15/2007	0.2569	0.4	0.011
9/16/2007	0.1957	0.3	0.009
9/17/2007	0.2207	0.3	0.010
9/18/2007	0.2757	0.4	0.012
9/19/2007	0.2935	0.5	0.013
9/20/2007	0.2540	0.4	0.011
AVG	0.254	0.4	0.011

Miles City Water Treatement Plant[1]

Date	Flow (mgd)	Flow (cfs)	Flow (cms)
9/11/2007	2.05	3.2	0.090
9/12/2007	1.89	2.9	0.083
9/13/2007	2.00	3.1	0.088
9/14/2007	1.98	3.1	0.087
9/15/2007	2.26	3.5	0.099
9/16/2007	2.41	3.7	0.106
9/17/2007	2.32	3.6	0.102
9/18/2007	1.93	3.0	0.085
9/19/2007	1.77	2.7	0.078
9/20/2007	1.73	2.7	0.076
AVG	2.03	3.1	0.089

Miles City Wastewater Treatement Plant[2]

Date	Flow (mgd)	Flow (cfs)	Flow (cms)
9/11/2007	1.12	1.7	0.049
9/12/2007	1.12	1.7	0.049
9/13/2007	1.09	1.7	0.048
9/14/2007	1.09	1.7	0.048
9/15/2007	1.09	1.7	0.048
9/16/2007	1.11	1.7	0.049
9/17/2007	1.11	1.7	0.049
9/18/2007	1.09	1.7	0.048
9/19/2007	1.11	1.7	0.049
9/20/2007	1.10	1.7	0.048
AVG	1.10	1.7	0.048

Glendive Water Treatement Plant

Not Required
Outflows DS of study reach

Glendive Water Treatement Plant

Not Required
Outflows DS of study reach

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[1]From monthly reporty of finished clearwell effluent
[2]Provided by City of Forsyth and Miles City

DNRC STREAMFLOW SUMMARY; 9/11-20/2007 - VALIDATION PERIOD

data provided by Larry Dolan, DNRC
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\DNRC streamflow data
T&Y diversion on Tongue River (no data collected in 2007); use data from 2005 (similar streamflow year)

Date	Flow (cfs)	Flow (cms)
9/11/2005	116.8	3.308
9/12/2005	109.4	3.098
9/13/2005	105.7	2.993
9/14/2005	99.6	2.820
9/15/2005	94.4	2.674
9/16/2005	88.6	2.509
9/17/2005	81.6	2.311
9/18/2005	75.7	2.144
9/19/2005	68.6	1.942
9/20/2005	64.3	1.822
AVG	90.5	2.562

**return flow measured on 10/1/2003 36.7 cfs

MDEQ DISCHARGE MEASUREMENT SUMMARY; 9/11-20/2007 - CALIBRATION PERIOD

data from MDEQ field measurements 9/10-11, 2007
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\MTDEQ streamflow data
data checked against original field sheets by KFF 3-26-2009

Location	Date	Site Visit Code	Discharge (cfs)	Crew
Cartersville Canal DVT	9/10/2007	Y0327	89.0	A. Welch, A. Nixon
Tongue River	9/11/2007	Y0365	213.2	A. Welch, A. Nixon
Kinsey Main Canal DVT	9/11/2007	Y0328	93.6	A. Welch, A. Nixon
Kinsey Main Canal RTN	9/11/2007	Y0329	28.2	1/0/1900
Powder River	9/11/2007	Y0331	78.9	A. Welch, A. Nixon
Forsyth WWTP	9/12/2007	Y0369	0.3	A. Welch, A. Nixon
Rosebud Creek	9/12/2007	Y0368	4.3	A. Welch, A. Nixon
Cartersville Canal RTN	9/15/2007	Y0364	47.0	R. Sada, M. Suplee
Shirley Main Canal RTN	9/16/2007	Y0376	15.5	R. Sada, M. Suplee
O'Fallon Creek	9/11/2007	Y0330	5.9	A. Welch, A. Nixon

MSU STREAMFLOW SUMMARY; 9/11-20/2007 - CALIBRATION PERIOD

data provided by Holly Sessoms, MSU
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\MSU streamflow data
diffuse irrigation returns (lateral canals off main canal)

	Irrigation Waste Drain Location and Streamflow (cfs)							
Date	Clear Cr #1	Clear Cr #2	Clear Cr #3	Whoopup Cr #1	Whoopup Cr #2	Sand Cr	Sand Cr #2	Sand Cr #3
9/11/2007	5.67	1.12	1.19	1.77	1.31	15.84	0.70	2.79
9/12/2007	5.33	1.33	1.24	1.67	1.27	14.00	0.68	2.68
9/13/2007	5.00	1.69	1.39	1.60	1.29	14.32	0.67	2.68
9/14/2007	5.49	1.70	1.50	1.67	1.36	14.78	0.66	2.76
9/15/2007	5.68	1.65	1.48	1.73	1.40	15.51	0.65	2.88
9/16/2007	6.00	1.76	1.54	1.76	1.39	15.72	0.58	2.93
9/17/2007	6.26	1.63	1.51	1.79	1.37	15.64	0.49	3.19
9/18/2007	6.24	1.70	1.52	1.79	1.40	18.63	0.44	3.17
9/19/2007	5.12	1.76	1.38	1.76	1.42	21.46	0.36	3.33
9/20/2007	4.83	1.76	1.59	1.75	1.42	21.69	0.29	2.78
AVG	5.56	1.61	1.43	1.73	1.36	16.76	0.55	2.92

USGS MEAN DAILY STREAMFLOW SUMMARY; 9/11-20/2007 - VALIDATION PERIOD

data downloaded from USGS NWIS 10/09 & 10/14 2009
data at:G:\WQP\WQ_Modeling\Yellowstone River\Nutrient Criteria\Physics\Hydrology\USGS streamflow data
data checked against original dv download by KFF 12-16-2009

VALIDATION - FORSYTH

9/11/2007	4,100	A
9/12/2007	4,160	A
9/13/2007	4,100	A
9/14/2007	4,100	A
9/15/2007	3,980	A
9/16/2007	4,000	A
9/17/2007	4,030	A
9/18/2007	4,000	A
9/19/2007	3,980	A
9/20/2007	4,070	A
AVG	4,052	

VALIDATION - MILES CITY

9/11/2007	4,390	A
9/12/2007	4,510	A
9/13/2007	4,500	A
9/14/2007	4,410	A
9/15/2007	4,360	A
9/16/2007	4,220	A
9/17/2007	4,240	A
9/18/2007	4,290	A
9/19/2007	4,210	A
9/20/2007	4,300	A
AVG	4,343	A

VALIDATION - GLENDIVE

9/11/2007	4,520	A
9/12/2007	4,780	A
9/13/2007	4,850	A
9/14/2007	4,850	A
9/15/2007	4,820	A
9/16/2007	4,840	A
9/17/2007	4,740	A
9/18/2007	4,750	A
9/19/2007	4,770	A
9/20/2007	4,710	A
AVG	4,763	

VALIDATION - TONGUE RIVER

9/11/2007	212	A
9/12/2007	243	A
9/13/2007	234	A
9/14/2007	212	A
9/15/2007	196	A
9/16/2007	195	A
9/17/2007	198	A
9/18/2007	201	A
9/19/2007	216	A
9/20/2007	227	A
AVG	213.4	

VALIDATION - POWDER RIVER

9/11/2007	94	A
9/12/2007	89	A
9/13/2007	85	A
9/14/2007	93	A
9/15/2007	101	A
9/16/2007	104	A
9/17/2007	104	A
9/18/2007	85	A
9/19/2007	85	A
9/20/2007	85	A
AVG	92.5	
ADJUSTMENT[1]	77.9	

[1]Adjusted to value at mouth - see suplement on adjustment method

APPENDIX C - MODEL INPUT AND OUTPUT FILES

Available upon request

or

available as of May 13, 2013 on the DEQ Website at:
http://test.deq.mt.gov/wqinfo/standards/NumericNutrientCriteria.mcpx

To access files in this folder please extract the zipped folder onto your computer.

APPENDIX D - PEER-REVIEW AND RESPONSE TO COMMENTS

October 5, 2012

Tina Laidlaw
USEPA Montana Office
10 West 15th Street, Suite 3200
Helena, MT 59626

Hi Tina:

Enclosed is a memo containing Montana Department of Environmental Quality's (DEQ's) responses to the Nutrient Scientific Technical Exchange Partnership & Support (NSTEPS) peer review for the Yellowstone River nutrient criteria model. We have done our best to address each comment (where appropriate) and will be revising the draft report accordingly. In an effort to make the subsequent pages easy to follow, we have shown the reviewer's comment in italics and our response in plain text. Please let us know if you need clarification, or additional information about any of the content.

Finally, we apologize about the lengthy turnover time in our response. This was largely a function of my academic commitments over the last year. In any regard, we look forward to discussing items as needed.

Best wishes,

Kyle F Flynn

Kyle Flynn, P.H.
Lead Hydrologist
Montana Department of Environmental Quality
Water Quality Modeling Program
1520 East 6th Ave.
Helena, MT 59620
Tel:(406) 444-5974
Fax:(406) 444-6836

SECTION 1.0 - Responses to Reviewer 1

"General Comments
This is a well written report on "Using a computer model to derive numeric nutrient criteria."
There are relatively few errors in the draft, which made reviewing clear. The use of multiple sources of information, including a computer model, is a very good idea for establishing nutrient criteria. The many concepts developed and employed in this effort are innovative, well founded, and sound. However, I disagree with the conclusions that model conditions warrant more credibility than other sources of information and that model results should be used to set nutrient criteria for the Yellowstone River.

In summary, my short responses to the questions are:

1. *The data used to run, calibrate, and validate the model were appropriate, but not sufficient.*
2. *Model calibration and validation were not good, because the fit of data to model runs was poor for a key endpoint variable, benthic algal biomass, and many results were biased.*
3. *The uncertainty of model predictions was problematic because: the model was not validated well for a key endpoint variable; the model was used to extrapolate to nutrient conditions outside the range for which it was calibrated and validated; and the model did not simulate extreme values well.*
4. *pH and algal biomass response endpoints should be used to establish nutrient criteria. The most sensitive response to a stressor (i.e. nutrients in this case) should be used to establish stressor criteria, even if different response endpoints are most sensitive in different types of habitats (in this case shallow and deep river habitats).*
5. *The appropriate methods were used to gather information about the development of nutrient criteria, but the results of the computer model were overstated and overweighted in a premature decision on nutrient criteria."*

1. Please evaluate the sufficiency and appropriateness of the data used to run the model.

"The data used to develop the model was appropriate, but not sufficient.

The computer model was designed to measure important response variables, such as benthic algal biomass, pH, and DO. These parameters respond either directly or indirectly to variation in nutrient concentrations and are used in either narrative or numeric water quality criteria in many states. These variables are highly appropriate from the perspective that we want to protect uses of waters. We know enough about nutrients to know the effects of nutrients instream and downstream. With proper research and synthesis of results, we should be able to set nutrient criteria above minimally disturbed conditions without threatening designated uses, such as drinking water, recreational uses and aesthetics, and support of biodiversity. Although we may not be protecting aquatic biodiversity of taxa that are highly sensitive to moderately increased nutrient concentrations in a habitat with nutrients above minimally disturbed condition, presumably those taxa are being protected in other habitats in which minimally disturbed condition is being protected (invoking tiered aquatic life uses). With the knowledge that biodiversity of some nutrient sensitive taxa will not be protected at nutrient concentrations that generate algal biomasses greater than 150 mg chl a m^{-2} and pH and DO standard violations, benthic algal biomass, DO, and pH can be appropriate endpoints for managing nutrients."

We disagree with the first portion of this comment (i.e., *"The data used to develop the model was appropriate, but not sufficient"*) and suggest that the DEQ effort meets/exceeds most steady-state modeling applications (see Mills et al. 1986; Barnwell et al. 2004; and reviewer 2's comments), including prior modeling studies in the literature (Paschal and Mueller, 1991; Park and Lee, 2002; Kannel et al. 2006; Turner et al. 2009). If anything, we feel it should be described as comprehensive.

"The right variables were modeled, measured, and calibrated in the field, but the sample size was low. Many of the key environmental variables were measured in the field, but they were measured at less than 10 locations. This limits the power of the comparison, much as a low sample size limits the statistical power in hypothesis testing. Was the fit or the lack of fit of the model to data due to chance or was it true?"

Sample size is just one of several factors that should be considered in modeling. According to Mills et al. (1986), other factors include site accessibility, historical locations, critical points of maximum or minimum concentration, and locations where water quality standards are expected to be violated. Because there are no hard and fast rules for sample size, an appropriate *n* is left up to the professional judgment of the modeler. Mills et al. (1986) suggest the sample size should be sufficient to describe the longitudinal profile of the river. So in the case of the Yellowstone, this was done. For example, we accommodated variability such as incoming tributaries, wastewater treatment plant discharges, critical downstream points of concentration, and spatial differences in temperature brought about by climatic gradients and hydrogeomorphology. So for the reviewer to suggest that random chance explained the structural differences in the data (e.g., larger diel oxygen swings in enriched areas, changes in algal biomass, increasing suspended solids, etc.), is simply not plausible. In this regard, we find the reviewer's comment speculative and without basis.

"The study should have been designed to have the calibration and validation datasets at the same time of year, perhaps sampling during summers of 2007 and 2008. The differences in temperature and light (day length and sun angle) between August and September could be substantial given they are within range that macroalgae like Cladophora are especially sensitive. August and September also have very different algal accumulation histories and processes regulating algal ecology probably differ as a result. Interannual variation in physical and chemical conditions in the Yellowstone River are relatively predictable, because of discharge regulation by snowpack melting, compared to rivers in parts of the country where unpredictable rain events have great effects on discharge and resulting physical and chemical conditions (e.g. light and nutrient concentrations)."

Similarity of environmental conditions (e.g., light, temperature, etc.) is not a necessity when considering mechanistic studies. Process-based models explicitly account for water temperature variation, solar radiation/time of year, biological rates, etc. thereby accommodating the differences pointed out by the reviewer. In fact, Chapra (2003) actually suggest that process-based models be calibrated and validated to substantially different conditions, such as flow, loadings, or climate. For example, a Level 2 model confirmation (i.e., the best) would require the model to be applied to cases with significantly different loadings and meteorology. While we did not meet this stringency, we did achieve a Level 1 confirmation which essentially means the model was applied to different meteorology and flows. That said, the accumulation history/autocorrelation of algae between August and September is a valid concern. We are currently investigating whether this is an important consideration or not.

"Another concern was having sufficient scientific foundation for model coefficients. Admittedly, some knowledge is better than none, but assuming that coefficients developed in lakes or other parts of the country and for different kinds of algae in one condition or another would apply to this location seems premature. Many of the parameters were developed in the 1970s or earlier, not that old is necessarily bad, but it is an indication that few new components were available or were found in the literature for use in the computer model. More field and laboratory research is needed to quantify the parameters being used in processed based models."

We did not directly apply coefficients from lakes or other parts of the country as suggested by the reviewer. Rather we made an initial assumption about such values (and associated ranges from the literature) and then calibrated those values to site-specific measurements (e.g., biomass, chemistry, water quality data, etc.). Such practice is common in water quality modeling and eliminates the need for direct parameter transfer as suggested by the reviewer. So the real issue seems to be kinetic parameterization of the model. We can only point to the fact that we used a combination of field/laboratory studies (e.g., light-dark bottle experiments, delta-method, SOD measurement, etc.) and field-calibrated state-variables (e.g., DO, pH, algal biomass, etc.) to provide the best (admittedly not perfect) model representation. Allowable ranges of coefficients were bounded by the literature and included quantification of both parameter sensitivity and uncertainty through first-order error and Monte-Carlo analysis techniques. While we agree that more data is always nice (note: we would love to do more field and laboratory research), at this time enough is known about site-specific biogeochemical processes (e.g., algal assimilation, hydrogeometric properties, chemical kinetics, etc) to provide reasonable assessment of the river's eutrophication response for regulatory purposes.

2. Please evaluate the model calibration and validation

"Model calibration and validation were not good, because the fit of data to model runs was poor for a key endpoint variable, benthic algal biomass, and many results were biased."

It is unclear to us what "not good" is, but root mean squared error (RMSE) of our simulation was 21.8 and 35.0 mg Chl*a* m^{-2} during August and September 2007 (n=77, excluding filamentous sites and a site with nitrogen fixers). Using a worse-case combination which includes filamentous algae and nitrogen fixers, RMSE was 55.5 mg Chla/m^2 (n=90), which approximates a seasonal average (i.e., average of August and September). While such errors are apparently large (according to the reviewer) they are no worse than routinely reported for empirical studies in the literature. For example, we compiled regression statistics via digitization of figures for about a half dozen of the more commonly cited nutrient-algal biomass papers and found that benthic algal biomass predictions, whether empirical or mechanistic, are quite similar (**Table-1**). In fact, the mechanistic model performed slightly better in nearly all instances than the studies considered. Plus it had the added benefit that other water quality state-variables such as DO, pH, etc. could also be simulated which cannot be done with a simple biomass model.

In consideration of **Table-1** though, it is important to keep in mind that the relative magnitude of RMSE is influenced by the range of biomasses evaluated, i.e., larger biomasses have the potential for greater prediction error than smaller biomasses and thus artificially weight the computed RMSE statistic. Thus some caution is needed in interpretation of results. Likewise, we suggest a more thorough review of both mechanistic and empirical models be completed before a definitive conclusion can be made about the predictive ability of each model type.

Finally, as pointed out by the reviewer, our model does contain bias. We have described it in Section 10.4.3.2 as under-prediction of high biomass and over-prediction of lower biomass (especially for filamentous algae). The prediction problems at the upper end reflect the inability of the model to simulate filamentous growth whereas those at the lower end are strictly applicable to diatom species. We clearly would like to remedy this deficiency, however, given the amount of filamentous algae in the lower Yellowstone River, further time and resource spent on model development is not warranted. We will address the filamentous concerns in the future, when both algal communities are present and necessitate the development of a model with better prediction capability.

Table 1. Comparative error analysis of commonly cited literature studies.

Study	Location	*RMSE* (mgChla/m^2)	*n*
Lohman et al. (1992)	12 streams and 22 sites in northern Ozarks, Missouri (annual mean of TN)	27.4	44
This study	90 algal sites Yellowstone River, Montana (instantaneous measurements during growing season)	29.6 [1] 55.5 [2]	77 90
Dodds et al. (1997)	205 streams or sites worldwide (seasonal mean of TN)	49.5	146
Suplee et al. (2012)	8 sites Clark Fork River, Montana (seasonal mean TP)	73	84
Chételat et al. (1999)	13 rivers in southern Ontario/western Quebec (TP)	85.4	33
Biggs (2000)	25 runoff fed rivers in New Zealand (SIN)	326.5	30
Welch et al. (1992)	26 sites in 7 New Zealand streams; mechanistic model	723	26

[1] Excluding sites where filamentous biomass or nitrogen fixers were present.
[2] All sites.

"Not much change was needed in many model parameters to calibrate the model, but many parameters for benthic algal growth were substantially different between the initial estimate and calibrated value (Tables 9-5, 9-6, and 9-7). Almost no discussion followed on the magnitude of these changes and if they were reasonable."

Initial parameter estimates are based on previous recommendations or initial data evaluations which must be adjusted on a per-system basis through model calibration (as described previously). Thus the magnitude of change from the initial parameter estimate is not a factor of whether a calibration is suitable or not (the fit between the observed and simulated data is!). In retrospect, we could have probably done a better job describing this in the text though. We did provide details on where estimates originated from in Section 8 (e.g., C:N:P ratios, subsistence quotas, nutrient uptake estimates, etc.) and we will be sure to add this reference to Section 9. Finally, we will add text describing the fact that values must be calibrated (i.e., an initial estimate is just that, and deviation from that does is not a significant concern provided the calibrated value is within the range of the literature).

"At least one set of the changes in parameters was relatively easy to evaluate and determine if they were reasonable. The mass ratio of N:P in algal cells is assumed to be 7:1, and in the Yellowstone River was often lower because of the relatively low supply of N versus P in the river. The initial mg N and P per mg algae (subsistence quotas for N and P) for benthic algae were assumed to be 0.7 and 0.1, respectively (Table 9-6).

- *The real issue is the relatively large change in one value during calibration and the unrealistic ratio for parameter values resulting from that calibration. The resulting calibration values of parameters for subsistence quotas for N and P were 3.20 mg N and 0.13 mg P, respectively. Even though each of these parameters independently fit within the range of possible values reported in the literature (remembering that one outlier in the literature has great effects on this range), the ratio seems very high for conditions within the Yellowstone River. The resulting mass ratio of subsistence levels of N and P was 3.20:0.13, which is more than 3 times the expected 7:1 ratio and 6 times the 4:1 ratios observed in low N habitats like the Yellowstone."*

It is commonly misconceived that subsistence quotas scale at Redfield ratio (7.2:1 by mass). However, Shuter (1978) provides a compilation of minimum cell quota data for N and P vs. biovolume (for phytoplankton) that seem to disprove this. From data on more than 25 algal species it is shown that N to P ratios deviate substantially from Redfield near the minimum cell quota. Recent work by Klausmeier et al. (2004) supports this assertion. They suggest resource acquisition machinery (i.e., nutrient-uptake proteins and chloroplasts) are P-poor, making the N:P ratio higher (ca. 20-30:1 by mass) nearer to the cell quota. Conversely, under nutrient replete conditions (more like Redfield) P-rich ribosome assembly machinery for exponential growth is more prevalent leading to lower N:P ratios. All of these findings are consistent with the classic work by Goldman et al. (1979) where it is shown that algal cellular N:P ratios are strongly influenced by the alga's growth rate. At very low growth rates (i.e., those approaching the minimum cell quota) cellular N:P ratios increase greatly to 45:1 (by mass). Hence we feel the ratio we have in the model is justified.

- *"Although internal N and P half-saturation constants are substantially different types of parameters than subsistence quotas, both are involved with algal growth, both were changed substantially during calibration, and ratios for both were unusually high."*

Very little data exists on internal N and P half saturation constants so we assume that this comment is pertaining to the external values. As mentioned previously, deviation from the initial estimates is not a problem (referring back to our previous response to this same question). However, we <u>do</u> agree the values required for calibration seem high in comparison to other work (e.g., Bothwell; 1985, Borchardt, 1996; Rier and Stevenson, 2006). That said Bothwell (1989) shows that low saturating levels are probably only valid during the cellular growth, at a time when nutrient supply is high and is not impeded by diffusion through the algal mat. Thus when algal biomasses are higher (or detrital accumulation is significant), it is possible that nutrient gradient/diffusion limits nutrient supply which may explain why higher values are needed to calibrate the model to a natural river. It is important to also realize that the Droop (1974) internal stores model is being used and thus to frame the overall response as a Michaelis-Menton or Monod saturation model, output biomass and soluble nutrient levels must be considered. By doing this we found that peak biomass saturated at around 152 μg/L soluble inorganic nitrogen (SIN) and 48 μg/L SRP (when not limited by other factors). Values such as these are not that different than suggested by the literature thereby providing additional confidence in the model's predictions.

Note: If the comment was specifically about internal half-saturation constants (the capacity for nutrient uptake based internal cellular stores), we acknowledge these values are poorly understood. Our best understanding is that they can be scaled in accordance with subsistence quotas at a ratio of around 1.0 for N and 0.5 for P (Di Toro, 1980; Droop, 1974; Rhee, 1973; Rhee, 1978). Given the uncertainty in their value, they were calibrated.

- *"The same kinds of problems were noted for the phytoplankton (Table 9-7)".*

Again, initial phytoplankton coefficients are estimates only, and must be calibrated. We will add a discussion regarding deviation from the initial estimates and what this means.

- *"A confusing issue initial parameter values (e.g. 0.7 mg N or 0.1 mg P per mg algae) indicate 70 and 10% of the algae were composed of N and P. Most of algal mass is carbon, not N or P. Presumably the units or my understanding of what these parameters mean were wrong."*

The reviewer is correct that the units could be easily confused. The values referenced are the initial estimates of minimum cell quota, or minimum level of nutrient deficiency normalized to Chl*a* [i.e., before our review of the Shuter (1978) or Klausmeier et al. (2004)]. As suggested by the reviewer, the actual makeup of algal cells is much different at a stoichiometric ratio of 40 mgC to 7.2 mgN to 1 mgP (i.e. Redfield).

"Fit of the model, similarity between predicted and observed conditions, was better for physical than chemical parameters, and better for chemical than biological parameters. QAPP criteria were not met for 1 out of 5 of the parameters assessed (Table 10-1). The variable with poor fit based on RMSE and RE was benthic algal biomass, either by using the Q2K or AT2K model. Since benthic algal biomass was a key response endpoint, and an endpoint for which nutrient criteria were eventually going to be made, it was important that the model predict benthic algal biomass well."

This is correct, the poorest part of the simulation was the biological component. However, the algal simulation error was quantified and was no worse than if we were to use other methods [referring to the previous discussion about Lohman et al. (1992), Dodds et al. (1997), Chetelat et al. (1999), Biggs (2000), etc.]. So if past efforts were acceptable (some of which were used in criteria determination), why would this effort be any different?

"As suggested on page 10-21, I agree that the AT2K model "allows us the ability to gain better information about spatial relationship of biomasses across a river transect," but I don't agree that AT2K model predictions were sufficiently accurate for the purposes intended for the modeling effort. High benthic algal biomasses were consistently under-predicted."

As indicated previously, the model's accuracy is comparable with past studies which means it should be suitable for its intended purpose (i.e., nutrient regulation on large river during the growing season where a vast majority of algal growth is closely attached to the bottom). That said, long isolated streamers of filamentous algae such as *Cladophora* present a problem. Computed biomass is greatly underestimated in these instances and we attribute this to the fact that the model simulates benthic growth in one-dimension vertically (i.e. thickening of an algal mat). In contrast, long *Cladophora* streamers grow up into the water column in 3-dimensionally which results in considerably higher biomasses for a given nutrient level and spatial area. Fortunately about 97% of all algal samples were diatom-like, so we do not see the underprediction of these isolated instances an issue (note: species shifts from diatoms to filamentous are a valid concern and we will evaluate this consideration <u>if</u> the river moves closer to the established criteria).

"During review of figures, I became concerned that deviations between observed conditions and conditions predicted by the model are more serious if they are biased than if they are randomly distributed above and below model predictions. This bias would not be captured in the RMSE and RE statistics for goodness of fit. For example, even though the RE is only 7.3% for TN calibration and 1.38% for validation (Figure 10-7, the model overestimates TN concentrations). The bias in predictions (residual error) is common in many of the nutrient and biological parameters. In most cases, bias was either high or low along the river, but in some cases it systematically switched from high to low, which you could imagine was the case for the August 2000 phytoplankton validation

(Figure 11-9). Systematic bias along the river is a concern because habitat conditions change systematically along the river."

We agree with the reviewer that model bias is undesirable. However, the level of bias suggested (ca. 10%), is hardly of concern (see Moriasi et al. 2007). Errors of this magnitude are considered "good" in the modeling literature. More importantly we feel the reviewer is mistaken in characterization of error calculation. RE is in fact a direct measure of bias, e.g., it sums the residual errors (predicted-observed) and divides those by the observations. So for the figure of concern (i.e., Figure 10-7), approximately 50 µg/L of bias occurs. While such an error is not conservative (i.e., does not side with the resource) this is not a great concern given the overall magnitude of nutrient levels in the river. Also, from review of the summary statistics in Table 10-1, it should be noted that several state-variables have larger bias. These are detailed in subsequent comments. Finally, with systematic bias, we would suggest this has more to do with data variability than systematic model error. While systematic habitat changes do occur in the river (e.g., shallowing near Miles City, increased turbidity below the Powder River, water temperature changes, etc.), we have characterized these features well and do not see how systematic artifacts could occur so rapidly in the longitudinal profile (referring to the reviewer's contention about the August 2000 phytoplankton data).

The model did not capture extreme conditions well, especially for benthic algae. If there was little variation, the model tended to fit much better than if a parameter varied greatly over the range of nutrient and habitat conditions in the river. For example, diurnal variation in dissolved oxygen and discharge were simulated well by the model, but pH and benthic algal biomass which varied much more than DO and discharge were not simulated well by the model.

The model may not have been able to simulate the high algal biomasses that accumulate in the river. For example in Figure 10-15, the model never predicted algal biomass to be greater than about 70 mg chl a m^{-2}. However, several observations of higher chlorophyll were observed. In addition, most of the observed levels of chlorophyll a were less than 50 mg chl a m^{-2} and fell within a confidence envelop that probably had a width of 40 mg chl a m^{-2}. So it would have been difficult for the model to be wrong when benthic algal biomass was less than 50 mg chl a m^{-2}. When benthic algal biomass was predicted or observed to be greater than 50 mg chl a m^{-2}, only 1 of the 10 prediction/observation points were within the RMSE confidence envelop."

In regard to the benthic algae simulation (and the inability to simulate high biomasses), the reviewer is correct that the cumulative frequency plot in Figure 10-15 shows under-prediction of higher biomasses which is a concern to us as well. We have been forthcoming about this in our discussion, and did additional analysis to make certain that the model would generate anticipated biomass levels under eutrophied conditions. This is described in Section 8 and Figure 8-5 and we show that maximum expected biomasses under nutrient and light replete conditions (with assumed losses of 50% from respiration and scour/grazing) would be around 300-400 mgCh*la* m^2, similar to that suggested by Stevenson, et al. (1996) for diatom communities. So while the model did consistently underestimate some field measurements (mostly filamentous algae), it will achieve maximum expected diatom community biomass under nutrient enriched conditions. Finally the reviewer is technically correct that the RMSE envelope covers nearly the entire simulated range (i.e., in their comment "*it would be difficult for the model to be wrong*"). However, this comment is somewhat misleading as nearly all of the data falls along the 1:1 line (in a structured fashion) and is certainly not random as inferred by the reviewer.

"Another issue with this model fit analysis is also the skewness of the distribution of observed and predicted values, with most points within 1/6th of the range of potential values (<50 mg chl a m^{-2} with a range of 0-300 mg chl a m^{-2}). Basically, it seems the model was not tested in the range of conditions in which it is intended to be applied."

We have no control over the skewness of the data as it is simply a function of field conditions and data collection methodology. The reality is that given the nutrient and light limitation of the river biomasses are low (<70 mgChl*a*/m^2), with exception of a few anomalous filamentous algal point measurements. With this understanding, it is surprising to us that the reviewer suggests we failed to test the model over the range of appropriate conditions. The immediate question that comes to mind is: (1) would we need a model if such conditions were already occurring and (2) could river-wide conditions for everything else (DO, pH, nutrients, etc.) be reasonably determined using any other approach (e.g., such as experimental troughs)? The obvious answer to both is no. Hence the primary purpose of the model is to help understand the response to a given set of enriched conditions while at the same time maintaining the fundamental/theoretical constructs of the eutrophication process. Finally, the reviewer is incorrect when implying that empirical restrictions be placed on process-based models. It is well-known that mechanistic models are a useful for predicting conditions outside of the environmental conditions they were developed (EPA, 2001; Canham et al., 2003).

3. Please comment on the uncertainty in the model predictions

"The uncertainty of model predictions was problematic because: the model was not validated well for a key endpoint variable; the model was used to make predictions for nutrient conditions outside the range for which the model was calibrated and validated; and the model did not simulate extreme values well. In particular, the inability of the computer model to simulate extreme values in benthic algal biomass was a concern."

We tend to disagree with this blanket statement and have described why in previous responses. To reiterate: (1) we did show that the algal simulation was no worse (in fact better) than many of the literature suggested approaches, (2) contrary to what the reviewer has indicated, it is OK to apply a mechanistic model beyond conditions which it was calibrated/validated (provided assumptions used in development of the model are valid), and (3) simulating extreme values (i.e., isolated cases where filamentous algae occur) is not an important consideration in this study.

"The poor prediction of algal biomass and inability to really evaluate model prediction of pH and other important response variables was discussed above."

The reviewer has not anywhere demonstrated a deficiency to evaluate pH or other important response variables (such as DO, nutrients, etc.). The fact is, short of benthic algae (which seems to be the reviewer's main focus), nearly all simulated state-variables achieved QAPP project requirements (and even algae did in one instance).

"A basic tenet of modeling, either statistical or highly calibrated computer models, is limiting extrapolation of results outside the range of conditions in which the model was developed. This model was employed outside the range of conditions for which it was calibrated. Since the computer model performed much worse when applied to September than August conditions, due to likely seasonal effects, wouldn't we also expect the same issues with performance outside the range of nutrient concentrations in which the model was calibrated?"

The reviewer's statement regarding extrapolation of modeling results conflicts with EPA guidance. In fact, EPA (2001) clearly articulates in Chapter 9, Use of Models in Nutrient Criteria Development that, *"Considerably more space is devoted to mathematical models, because they are capable of addressing many more details of underlying processes when properly calibrated and validated. They also tend to be more useful forecasting (extrapolation) tools than simpler models* (referring to empirical models), *because they tend to include a greater representation of the physics, chemistry, and biology of the physical system being modeled (NRC 2000)"*. We therefore do not understand the reviewer's concern, especially since process-based models have a long and successful history in waste-load allocations and effluent loading studies (Thomann, 1998; Chapra, 2011).

With respect to the seasonal issue (September vs. August), there is no reason to make the linkage suggested by the reviewer. We in fact provided a very satisfactory explanation for the deficiency between August and September 2007 and also completed a second validation for August 2000 which confirms the model performs well during peak growth conditions (i.e. August). Additionally, the calibration and confirmation were collectively completed over a range of different soluble nutrient conditions including nitrogen levels ranging from 5-105 μg/L and phosphorus concentrations from 3-17 μg/L (across the longitudinal profile). As such, soluble nutrients spanned almost the entire range evaluated for criteria determination, with the caveat that nutrient supply was elevated over only a small spatial extent usually in the vicinity of the wastewater treatment plants. Thus to question the model performance over a period which in essence has already been validated for varying nutrient conditions (i.e., August) is unjustified.

"Process based models (i.e. computer models) are theoretically better than statistical models for predicting outside the range of original conditions in which they were calibrated. However, the extent and magnitude of calibration from an initial values used in model is a key issue for using process based models to predict outside the range of calibration. Prediction outside the range of conditions for which either the statistical or process based model was calibrated requires that we know enough about the system and the behavior of the system in the two ranges of conditions (e.g. August versus September, or low and high nutrient concentrations) that we are confident that the models accurately describe behavior of the system. The less that you have to calibrate a model to new conditions to get a good fit, the more confident you can be that the model will perform well in a new set of conditions. The more fundamental the processes are that are simulated in the model and the fewer number of assumptions made for use of the model, the more certain you can be that the model will predict responses well in a set of conditions for which it was not calibrated.

Since there is little evidence that the model did perform well, either calibrating for key endpoints or predicting responses during validation, we should have concerns about accuracy of predictions by the model for ecological responses in higher nutrient concentrations for which the model was tested. In addition, many key parameters in the model were changed greatly during calibration from what were initially thought to be appropriate. So based on model performance, we cannot be certain that it will perform well outside the range of conditions in which it was calibrated, or even within that calibration range for some key parameters."

We agree that process-based models are better than statistical models for predicting conditions outside the range which they were developed (i.e., that is their primary utility), but disagree that *"there is little evidence that the model did perform well"*. If fact, we have clearly articulated the model's predication capability throughout the draft report as well as in many of our responses. One further clarification is necessary though. The reviewer describes August and September as "*low and* *high nutrient concentrations*". However, this is not the case. Rather nutrient supply was the same both periods (i.e. loadings were similar), but uptake during each period was significantly different. Finally, with respect to the certainty of model predictions, the entire premise of the model is to represent fundamental biogeochemical processes. These were shown to be adequate for August low-flow conditions (based on two different years of data, i.e., 2001 and 2007) and over a large longitudinal extent. Thus it is reasonable to conclude that the model is suitable for making regulatory predictions over this time-frame, especially since as noted previously, nutrient supply was sufficiently variable in both years.

"Many assumptions needed for the model also seemed to reduce credibility of its results. Some assumptions were probably met as well in the Yellowstone River as anywhere. For example, the assumption about the model simulating a steady state equilibrium is certainly more appropriate for rivers like the Yellowstone with snow-melt dominated and relatively predictable hydroperiods versus many other rivers where storm events have dramatic and unpredictable effects on hydroperiod."

Violation of model assumptions by the ecosystem may also explain why the model simulated the ecosystem poorly. Of course assumptions are necessary, but some violations of assumptions or combinations of violations may accumulate explain the unsatisfactory behavior in the model. Here are a few examples:

- *The assumption that velocity and channel substratum are "sufficiently well mixed vertically and laterally" (pg 5-8, lines 3-4) may explain why the high algal biomasses were not simulated. If average, versus optimal velocity and substratum were used that would underestimate the high algal accrual possible in optimal velocity and substratum conditions."*

  We disagree that the model simulated the ecosystem poorly (for all of the reasons stated previously) but do agree that spatial variability of substratum and velocity may be an important consideration in algal growth. We are working on improving modeling techniques to better represent these physical processes in riverine settings. The assumption of vertical and lateral mixing referenced by the reviewer holds only for the water column (i.e., turbidity, nutrient concentrations, phytoplankton, etc) and we will revise the text to make this clearer.

- *"Why assume dynamic equilibrium between particle re-suspension (drift) and deposition (settling)(pg. 8-20, lines 24-25)?"*

  We will rewrite this sentence to clarify. Dynamic equilibrium between particle resuspension and settling was based on conclusions of Whiting, et al. (2005) which was based on longitudinal sediment analysis of the Yellowstone River. For the model we applied our calculated Stokes settling velocity of 0.012 m d^{-1} for sediment and 0.086 m day^{-1} for phytoplankton (calibrated down to 0.05 m day^{-1}) reflecting a net loss in the mass balance for each term.

- *"Why assume the typical meterological year during a ten year period. For example, to understand the conditions under which problems would arise 1 in 10 years, aren't regional weather patterns a likely cause of those problems. Rather than running a typical meteorological year, shouldn't the 10-year extremes be boundary conditions for a run to understand the effects of less common conditions?"*

The use of a typical meteorological year stems from the desire to not alter the underlying frequency of occurrence. For example, if a 1 in 10 year low-flow condition were simulated with a 1 in 10 year climate (both of which have independent probabilities), the underlying design condition would be a 100 year event (probability of occurrence of 0.1 x 0.1 = 0.01). Such an infrequent event is not appropriate for nutrient regulatory management. As indicated by the reviewer, however, an equally viable approach would be to use a 1 in 10 year climate, with a 1-year flow condition although in this instance the latter reflects a much larger system volume (from the increase in flow) which would likely outweigh any extreme climatic effects.

Note: We have modified the design flow to a 14Q5 (1 in 5 year low-flow condition) to better align with EPA recommendations on allowable frequency of exceedance of standards (which were originally based on a biologically 4-day average flow once every 3 years, i.e., 4B3). The 4B3 is often used as a basis for U.S. EPA chronic aquatic life criteria.

"In addition to violation of the assumptions in the model, there may be issues with the analytical foundation of the model to accurately represent ecosystem processes; but I am not sufficiently familiar with the model to make that judgment. For example:

- *Were growth patterns and differing spatial resource limitation (density dependence) for macroalgae and microalgae or algal taxa included in the model?"*

  It would have been helpful for the reviewer to familiarize themselves with the model prior to doing a critique of its analytical foundation, but in general we will try to answer each question straightforwardly. Relative to different growth patterns/state-variables for each algal taxa, Q2K models only a single algal species therefore any difference between macro and micro-algae species is only accommodated through parameter lumping. We recognize this as a model deficiency (especially if applied in an area where both macro and micro-algae were in competition), however, a majority of the river sampling sites (~97%) were dominated by a mixed assemblage of diatom species which at least reduces the concern of macro- and micro-algal dynamics. Thus it was not a concern in the modeling endeavor.

- *"Space limitation in the model, if I understand it correctly, is not the correct conceptualization of the process that regulates density dependent growth of benthic algae. Developing a more realistic characterization of the processes regulating benthic algal accumulation and density-dependent depletion of nutrients within mats would be very interesting and perhaps improve model predictions. Effects of mixing and diffusion vary greatly between different types of algae that grow in differing nutrient and temperature ranges, such as macroalge (Cladophora) and microalgae (diatoms)."*

  While in one section of the report we use a logistic function/space limitation to illustrate biomass accumulation for the purpose of estimating zero-order growth rates (under optimal nutrient and light conditions), such a formation is not actually used in the Yellowstone River model. Instead the governing differential equation for the mass balance of algal biomass is based on Chapra et al. (2008) where biomass increases due to photosynthesis and is moderated by a number of loss terms including respiration, excretion, and death (inclusive of grazing and scour etc.). This would have been clear if the reviewer would have taken the time to review the Q2K model which can readily be found on the EPA website http://epa.gov/athens/wwqtsc/html/qual2k.html. The model is based on the work of McIntire (1973), Horner et al. (1983), Uehlinger, (1996), and Rutherford et al. (2000), includes Droop (1974) nutrient limitation (i.e., the internal stores model), saturation light limitation (Baly, 1935; Smith, 1936; Steele, 1962; light), uptake dependent on internal and external nutrients (Rhee, 1973), and many other physiology-based processes. In this regard, the effects of nutrient diffusion into the algal mat are not explicitly considered, but are implicit in calibration of the external half-saturation constants for nutrient uptake.

- *"Was N-fixation included in the model and the potential for N transfer between epiphytic diatoms with cyanobacterial endosymbionts on Cladophora? Is it possible that Cladophora cells close to the substratum take up nutrients and transfer them to younger, actively growing cells in the ends of the filaments suspended in the water column. Only the cells at the tips of Cladophora filaments reproduce, so they are younger and have fewer epiphytes than cells at the base of filaments. Cladophora cells that are closer to the substratum, having more epiphytes, bacteria, and entrained detritus as well as slower currents, have greater potential for uptake of recycled nutrients in the epiphytic assemblages around them than younger cells in the water column. Cladophora does not have complete cross walls between cells, so fluid in cells can theoretically mix between cells, which would be facilitated by the movement and bending of filaments in currents. Thus, nutrient concentrations in the water column may be poor estimators of nutrient availability to Cladophora, as well as other benthic algae, because of nutrient entrainment and recycling in the mats."*

  N-fixation is <u>not</u> included in the model and its importance (at one site) was identified only after finding discrepancies between simulated and observed data. Similarly, nutrient exchange from epiphytic diatoms with cyanobacterial endosymbionts to *Cladophora* is <u>not</u> represented. Both are far too detailed processes for a general purpose water quality model. Finally, while the *Cladophora* mat self-sustainment process described by the reviewer is interesting and may occur, the concept seems in conflict with the common observation in Montana and elsewhere that dense stands of long streamers of *Cladophora* most frequently colonize the riffle regions of streams and rivers; this was reported as long ago as 1906 (Fritsch, 1906). Increased turbulence and advection in riffles clearly creates preferred habitat, in part because it induces more nutrients from the water column to go deeper into the mat, allowing for continued photosynthesis (Dodds, 1991). If the mat nutrient-recycling process described previously is important to mat maintenance, there is still the obvious question of what stimulates *Cladophora* mats and long streamers to develop in the first place? The scientific literature is replete with works dating back to at least the 1950s indicating that *Cladophora* blooms are associated with elevated nitrogen and phosphorus concentrations in the <u>water</u> of rivers and streams (see Whitton, 1971 and Hynes, 1966 for starters). As such, we believe the scientific literature generally supports the idea that nutrient concentrations in flowing waters are correlated with the development of algal mats.

"Another reason for questioning model predictions could be the high nitrogen and phosphorus concentrations that are predicted to generate nuisance blooms of benthic algae: 700 µg TN L^{-1} and 90 µg TP L^{-1} in Unit 3 to prevent pH violations and 1,000 µg TN L^{-1} and 140 µg TP L^{-1} in Unit 4 to prevent nuisance benthic algal problems. Although we know relatively little about nutrient concentrations affecting pH in river, these phosphorus concentrations are many times higher than phosphorus concentrations thought to cause nuisance levels of benthic algal biomass, e.g. greater than 150 mg chl a m^{-2}. Admittedly, there's a great range limiting and saturating nutrient concentrations in the literature, but a 30 µg TP/L benchmark was proposed in the Clark Fork, which

is upstream from this location. Why have higher numbers in the larger mainstem of the Yellowstone River? If we assume Leibig's law of the minimum, and nitrogen and light are sufficiently great to allow algae to grow, why wouldn't the marginal habitats of the Yellowstone River generate nuisance algal biomasses at 30 µg TP/L? At least one reason could explain that discrepancy. The reactive portion of the TP may be lower in the Yellowstone River than in smaller streams where nuisance blooms of benthic algae commonly occur at TP concentrations around 30 µg TP/L. The soluble fractions of total nutrient concentrations, assumed to be the most readily available fractions, were very low in the Yellowstone River during low flow conditions (Table 6-6). However, caution should be exercised when assuming only the soluble fraction of TP is bioavailable; mounting evidence indicates that entrained particulate P and N are recycled in benthic algal mats."

Higher criteria occur in the Yellowstone River for two reasons. First, the response to nutrients is integrated over the wadeable region (<1 m depth), which as Hynes (1969) points out, means that only a portion of the river bottom will be conducive to algal colonization and growth. The second is river turbidity which is considerably higher than western Montana wadeable streams. Hence the comparison between the Yellowstone River and the Clark Fork River by the reviewer is not valid. They are in fact different ecoregions, the lower Yellowstone is significantly more turbid and deeper than the Clark Fork River, and finally the former drains to the Missouri River and the latter to the Columbia River. The reviewer is correct though in one regard, that the Yellowstone River should still grow algal biomasses on the margin of the river at lower nutrient levels; this is the very reason we developed the AT2K model, i.e., to integrate the effect over the entire management area.

With this in mind, the manner in which management endpoints are computed strongly affect the criteria. For example, we used the average benthic algal biomass that develops in the wadeable zone (defined as depths of ≤1 m) as our regulatory endpoint. By doing this, it means that algae in the deeper regions of this zone are significantly light limited, and thus the areal-average response is lower. If we managed the river so that no stone were to exceed 150 mg Chl*a*/m^2, the criteria would be different and would be nearer the levels suggested by the reviewer [around 35 µg/L SIN and 10 µg/L SRP which if applied to the soluble regressions of Biggs, (2000) and Dodds et al. (1997) yield biomasses that are less than, or very close to nuisance levels]. However, regulation of a single stone (i.e., the single highest algae level) would not be consistent with the way the algal biomass threshold was derived. For example, the basis of Suplee et al. (2009) was that participants were shown photos of entire river reaches and were asked their impressions (acceptable/non-acceptable) of the entire scene. Since the impressions would be based on the overall appearance of the algae levels (not a single point), and, correspondingly, the algae biomass values provided were the reach averages (of n=10 to 20 replicates), we must regulate biomass for the average of the wadeable region, not the single highest Chl*a* value recorded (i.e., the single most-green stone).

"The model prediction that low DO is not likely in the Yellowstone River seems reasonable. The Yellowstone River is relatively hydrologically stable, so it is probably not prone to types of extreme low flow events that allow development of low DO with resulting fish kills. Rivers and streams are probably much more susceptible to high pH and fluctuating pH conditions than to low DO; but both phenomena have not been studied sufficiently to understand thoroughly."

We concur with this statement, and also point out that choosing a process based model allowed us to understand both DO and pH dynamics, something that cannot be determined through statistical methods. Thus there is merit to the mechanistic approaches beyond what could be determined using empirical analysis.

4. Please comment on the appropriateness of using response variables, such as chl-a and pH, as model endpoints for numeric criteria derivation, and thus protection of water quality from nutrient pollution. Please comment on the spatial application of different response variables for deriving numeric nutrient criteria (pH was used for the upstream segment while benthic algal biomass was used in the downstream segment).

"pH and algal biomass response are appropriate endpoints for justification of nutrient criteria. pH is more directly linked to negative effects on aquatic fauna than nutrient concentrations, so pH is a more proximate threat to a valued ecological attribute. High algal biomass is known be an aesthetic problem in rivers, as established in the great study by Suplee et al. As described above, nutrient criteria above minimally disturbed conditions that prevent nuisance algal accumulations and violation of pH and DO standards may not protect biodiversity of some nutrient-sensitive taxa; however chl a and pH, as well as DO, are appropriate endpoints for protecting designated uses.

The most sensitive response (e.g. chl a, pH, or DO) to a stressor (i.e. nutrients in this case) should be used to establish stressor criteria, even if different response endpoints are the most sensitive in different types of habitats (in this case shallow and deep river habitats). An important goal of environmental management should be protection of ecosystem services. Of course all ecosystem services should not have to be protected in all waters, but appropriate protection is warranted. Montana DEQ and presumably a majority of the people of Montana have supported water quality criteria related to pH and benthic algae. So nutrient concentrations should not be allowed that would generate unacceptable risk of violating the pH and nuisance algal biomass criteria.

The focus on shoreline algal biomass was also appropriate because that is where people most commonly observe the water as they use the resource for recreational purposes."

We agree with this comment.

5. What other analytical methods would you suggest for deriving numeric nutrient criteria for the mainstem Yellowstone River?

"The appropriate methods were used to gather information about the development of nutrient criteria, but the results of the computer model were overstated and overweighted in a premature decision on nutrient criteria.

Processed based (computer) models are very informative and valuable, but they are just one line of information. Three basic research approaches can be used to develop numeric nutrient criteria: observing patterns in nature and quantifying relationships between nutrients and key endpoint variables with by statistical models (e.g. regression models); simulating patterns in nature using process-based models; and experiments in controlled environments in which environmental conditions are purposefully manipulated. Each of these methods complement each other. When they all do not agree, then conclusions are suspect. In this case, the predictions of the computer model do not match results of other research based on statistical models and experiments. Even though there are plausible reasons for those discrepancies, there is little reason that the computer model is accurate."

We do <u>not</u> believe our results were *"overstated and overweighted"* and defer to reviewer 2's comment in support of this. Also, we factually disagree with the reviewer that it is appropriate to draw direct parallels between the computer simulation and other methods suggested (e.g., statistical and experimental). They reflect distinctly different processes, meaning we shouldn't try to force a large river into a wadeable streams approach! A large river (e.g., the Yellowstone) has great spatial variability in light (as described throughout the draft report) whereas wadeable streams are shallow and homogenous. While similar methods could be used to develop statistical or experimental procedures for large rivers, the reviewer misses a very important part of water quality management, that is algal biomass is just one endpoint of consideration. What about pH, DO, or any other important water quality indicators? How would we evaluate their response if just a statistical model of biomass was used? Even if we had such a model, could the model be extended suitably to ascertain criteria? Finally, if streamside mesocosm experiments were completed, would these be comparable to a large river which is primarily deep and turbid and has large underwater areas unsuitable for significant algae colonization? These are questions we asked ourselves prior to initiating the project and simply couldn't answer (even if we combined the statistical and experimental methods). Thus in our opinion modeling is the best line of evidence for criteria determination. Other methods were considered (reference sites, the literature, and experiments) but these were not used due to their inherent limitations in representing the large river response.

"Despite that lack of fit between computer model predictions and measured conditions in the river, during both calibration and validation, the computer model was used. In a simple comparison of accuracy of the computer model predictions of high algal biomass as a result of higher nutrient concentrations (Figure 10-5) and the regression model characterizations between algal biomass and either TN or TP (Figure 15-2), show the regression model warranted more credibility. For the computer model, there was no relationship between algal biomass predicted and the algal biomass observed at stations (Figure 10-5). Plotting these abundances in Figure 10-5 on a log-log scale may have improved the apparent fit, but lack of fit at higher biomasses is likely. Remember the discussion above about lack of data points above 50 mg chl a m^{-2} and poor range of observed conditions. For the regression models, the results were variable but plausible (Figure 15-2). If N:P ratios are low and N limits algal growth, then we'd expect a relationship between algal biomass and TN and not between algal biomass and TP concentration. The range of TP concentrations (and bioavailable P indicated by those concentrations) may have been above the TP concentration considered to have strong effects on benthic algal growth (e.g. 30 µg TP/L). The range of TN concentrations may have crossed the sensitive range and below the limiting nutrient concentration for TN; therefore TN may have been the primary limiting nutrient in the Yellowstone River. Thus, the Montana DEQ got a relationship between TN concentrations and benthic algal biomass, but not TP concentrations and benthic algal biomass. I disagree with the interpretation by Montana DEQ about these relationships. These relationships do show that TN concentrations below 505 µg TN/L should constrain average algal biomass to less than 150 mg chl a m^{-2}, but the lack of significance in the TP algal biomass relationship indicates it should not be used to set a TP criterion. This relationship between TN and algal biomass is really the only evidence in the report for nutrient regulation of benthic algal biomass."

The reviewer suggests that a *"lack of fit"* between the observed and predicted plot (Figure 10-5) makes the mechanistic model unreliable and less suitable than the algal biomass regression in Figure 15-2. However, no evidence is provided supporting this statement. In fact, we have shown previously through analysis of RMSE that the errors are comparable to refereed literature (which is frequently relied on for nutrient criteria). Similarly the use of loose statistical dependence as shown in Figure 15-2 (r^2=0.34, which the reviewer forgot to point out is also log-scale) would be careless given the way the data is collected (oriented toward the shallow regions) and simply not the best available information (which we have compiled via the data collection and modeling). Finally, even if the regressions mentioned by the reviewer were suitable, they cannot be extrapolated beyond the observed data, which is a problem give that the concentrations in the river are well below nuisance levels. All of that said, we do agree on one thing, that the TP regression in Figure 15-2 is not useful and we will revise the report indicating this.

"If benthic algal biomass is not simulated accurately by the computer model, can we trust predictions of pH and DO that respond to changes in algal biomass? pH and DO predictions of the computer model were also not validated well because of low sample sizes and ranges of conditions in which the model was calibrated."

We feel this comment is misleading and we have shown that both DO and pH were reliably simulated in two separate August low flow conditions in 2000 and 2007 including both spatial averages and associated diurnal variability. We also stress that the DO and pH response are implicitly a function of the photosynthetic response, which in the case of the Yellowstone River was directly driven by benthic algae. So even if our biomass point measurements did not match perfectly, the community response was correct (as substantiated by reviewer 2's comments). Also, we also point out that the "*low*" sample size mentioned by the reviewer was on par with any academic modeling study (nationally and internationally) and that conditions in the study were sufficiently variable for the intended analysis.

"Another question develops about whether TP concentrations need to be kept below a TP criterion that would constrain algal biomass, if TN concentrations are below that 505 µg/L; but that question is a policy deeper policy question. If TN is kept below 505 µg/L, then presumably there would not be a response of benthic algae to TP if N is the primary limiting nutrient. However, the 505 TN and 30-60 TP range seem close to what I would expect to be saturating nutrient concentrations. So, a combination of TN and TP criteria would provide double protection against risk of high algal biomass."

We agree that criteria levels for both TN and TP are protective and should accommodate future shifts in nutrient availability. That said, water quality managers must use common sense when determining nutrient control strategies and permitted load limits. According to Liebig's law of the minimum, a single available resource (e.g., soluble N or P) will limit yields at a given time which implies that only a single nutrient should be considered in management (unless they are both close to limiting, e.g., co-limiting). Soluble concentrations are difficult to quantify however (Dodds, 2003), and thus we have used the rate of uptake/recycle and associated transport in the model to determine how total nutrients at one point relate to conditions at another (note: these points are different longitudinally because of advection). Given that minimum acceptable nutrient criteria outlined by U.S. EPA were total nutrients, and the fact that totals better lend themselves to ambient nutrient monitoring, permit compliance, and monitoring, we thought this was the most reasonable approach toward criteria development.

"Good calibration of models, computer or regression, should not be expected in a river without a good range of nutrient that result in algal problems at some place across the range of nutrient conditions. In habitats in which no algal problems are observed, it is possible that sediments and low light constrain algal accumulation such that nutrients have no effect on instream algal related conditions. In this case, downstream effects should be the concern/endpoints of criteria. Alternatively, it is possible that most that we know about the asymptotic relationship between nutrient concentrations and algal biomass is not true; or for some other reason, TP concentrations

above 50-100 µg TP/L do regulate benthic algal biomass. Then the high nutrient concentrations as those proposed (700 µg TN L^{-1} and 90 µg TP L^{-1} in Unit 3 to prevent pH violations and 1,000 µg TN L^{-1} and 140 µg TP L^{-1} in Unit 4 to prevent nuisance benthic algal problems) would be appropriate in the Yellowstone River."

We calibrated the models to a range of nutrient conditions so we are not entirely sure where the reviewer is coming from by suggesting the calibration was insufficient (recall soluble N varied longitudinally from 3 to >100 µg/L and variants in soluble P were from ≈3-20 µg/L). Additionally, as mentioned previously, we have shown site-specific environmental considerations (e.g., light) do in fact play a significant role in the productivity of the Yellowstone River (see Figure 2-2 and 2-3 and associated discussion). Lastly we have in fact defined the asymptotic relationship between ambient nutrient levels and biomass response (among other variables) through the model. The response of the Yellowstone River is different than saturating responses of other methods because of, as stated previously, gradients in light. So in fact we are <u>not</u> suggesting, "…*that most that we know about the asymptotic relationship between nutrient concentrations and algal biomass is not true…*" rather that the conditions of previous studies are far different than our application, which is why we chose a modeling approach in the first place. With that in mind, the levels determined in the study are not surprising. In fact, they are very comparable to concentrations suggested for other light-limited wadeable streams in eastern Montana, e.g. ≈1,400 µg/L TN and ≈140 µg/L TP for the Northwestern Great Plains (Suplee and Watson, 2012).

"Continued research in the form of monitoring of the Yellowstone River, surveys of other large rivers, experimental research, and computer modeling will be needed to develop nutrient criteria that protect ecosystem services of large rivers without overprotection. Continued monitoring in the Yellowstone River will enable assessment of whether nutrient concentrations are increasing and nuisance algal biomasses and high pH are becoming more frequent. This will forewarn managers that nutrient related problems are developing and will provide the additional information needed for better computer and regression models used to establish nutrient criteria. In the report, Montana DEQ did propose continued monitoring and data analysis with one goal being learning more about nutrient effects in the river for potential revision of the proposed nutrient criteria. But will reducing the nutrient criteria, based on new science, be practical politically. Why will the public believe the new science if the old science was not sufficient? Why hurry to have nutrient criteria if there are no known problems? Was this the wrong place to try to develop nutrient criteria for large rivers?

A concerted national effort should be developed and maintained to gather the kind of information needed for developing nutrient criteria in large rivers. Monitoring data as well as experimental results should be gathered and evaluated with statistical models and integrated in processed based models to provide sufficient information for development of nutrient criteria in large rivers. Great similarities exist among the large rivers of the world, such that information learned in multiple rivers should be able to be synthesized and related to other large rivers. Until this information is gathered and analyzed, perhaps the most prudent nutrient management strategy is to try to maintain current conditions if there are no existing problems."

We agree with the reviewer that additional computer modeling and further surveying/sampling of the Yellowstone River is important going forward; we point out that Montana has been one of the most active states when it comes to lotic nutrient standards development. Relative to the need for additional study, the reviewer assumes that continued research would result in more stringent criteria for the Yellowstone River, when in fact the standards could go either way. As a matter of point, the standards would also need to be changed if the beneficial uses of the river currently in law were to be changed by the public. The political reality of water quality standards is they are updated constantly which is why the Clean Water Act requires states to review them every 3 years. Sometimes standards are made more stringent, sometimes relaxed. Our experience in this matter has been that the public accepts improving engineering/science, and that these advances can result in changed water quality standards.

We disagree with the reviewer's suggestion that this may be the wrong river to study at the wrong time. Water quality standards are not just for polluted rivers they are also to protect those that are still healthy; that's why all states have non-degradation policies as part of their water quality standards. The Yellowstone is one of the fastest growing regions of the state (e.g., Billings population increased about 15% from 2000-2010) and nutrient-laden discharges from urban areas will only steadily increase. We selected this river segment because it was un-impounded (which simplifies modeling and interpretation of applicable water quality laws), it is well gaged, and reasonably reported on in both the open- and grey-literature. This means that there was a good chance of successfully developing nutrient criteria for the river.

Finally, we agree with the reviewer that a concerted national effort to gather data on large rivers including the use of modeling and experimental research would be valuable. We hope that the academic community will undertake such work. However, our finding has been that national efforts to develop numeric nutrient standards for large rivers by anyone, academic, governmental, or private, has been slim to none. This has occurred is in spite of the fact that former Vice President Gore's Clean Water Action Plan, which called on states to develop numeric nutrient criteria for waterbodies, was published in the Federal Register in 1998, fourteen years ago! Work on large river nutrient standards needs to be started by someone, somewhere, and we feel our study was an excellent start. We believe the use of existing water quality models (and development of new models such as the one described by DEQ) will help advance criteria-development methods nationally. Note that the Water Environment Research Foundation (WERF) is currently researching the use of such models for site-specific criteria determination.

"A couple editorial changes worthy of note:
Figure 9-1 makes much more sense to me if Table 8-1 were changed to Table 9-1."

Thank you. The section numbering changed several times and we did not get corrected in Figure 9-1. We will make this change.

"Figures 13-4 and 15-2 were hard to understand because the independent variable (nutrient concentration) was not on the X axis."

We have received this comment from reviewer 2 and will make this change. We had initially plotted the state-variable of interest on the abscissa and the criteria on the ordinate as nutrient criteria are really the dependent variable. However, since this apparently has been confusing to a number of people, we will make the change.

SECTION 2.0 - Responses to Reviewer 2

"Using a Computer Water Quality Model to Derive Numeric Nutrient Criteria, Lower Yellowstone River, MT (Montana DEQ, 2011) provides a comprehensive discussion of Montana Department of Environmental Quality (DEQ) efforts to develop nitrogen and phosphorus criteria for the lower Yellowstone. This is done through the development of a site-specific mechanistic nutrient response model that links nutrient loads to measurable endpoints associated with the support of designated uses in the river. The approach is consistent with EPA guidance on establishing TMDLs to address narrative nutrient criteria, which also results in site-specific objectives.

The result of the study is recommendations on site-specific nutrient criteria for the Lower Yellowstone. The results are truly site-specific as they depend on the conditions present in the Lower Yellowstone and it is not clear that they would be applicable to other, similar waterbodies. The results could serve as a template for the derivation of site-specific criteria for other large rivers; however, the evidently high level of effort required to complete this study may preclude wide application.

In general, the modeling and analysis presented here is well done and adequately documented. There are, however, some specific questions that should be resolved before finalizing the analysis. These are described below.

The site-specific nutrient response approach is attractive for several reasons. As noted by DEQ, there is a lack of reference watersheds for large rivers, and methods appropriate to wadeable streams are not transferable to large rivers. In addition, nutrients themselves (except at extreme concentrations) generally do not directly impair designated uses; instead, it is the secondary effects of elevated nutrients, generally involving algal growth, that lead to use impairment. These secondary effects differ according to site characteristics, such as light availability, residence time, and scour regime, which means that the assimilative capacity of a waterbody for nutrients is inherently site-specific and determined by a variety of co-factors; thus the most economically efficient nutrient criteria should also be site-specific.

DEQ has developed site-specific criteria for the lower Yellowstone that reflect specific characteristics of the basin. Notably, the river is deep and turbid, both of which characteristics reduce light availability and thus also reduce the expression of nutrient impacts through algal growth. In other words, these characteristics of the Yellowstone River serve to increase its assimilative capacity for nutrients.

It is clearly appropriate to consider the hydrologic characteristics of the Yellowstone in developing site specific criteria. In particular, the amount of flow and depth of the river, which reduce the area in which benthic algae can grow, is a largely natural condition. The case for turbidity is a little less clear. The tributaries of the Yellowstone, especially the Powder River, are believed to be naturally turbid. However, the present day turbidity is also affected by land use practices (silviculture, agriculture, grazing, mineral extraction, etc.). If turbidity is greatly elevated by anthropogenic sources then it would appear inappropriate to count the full effect of high turbidity on reducing algal growth as a "credit" that allows for higher nutrient concentrations.

The report (p. 4-8) says, regarding sediment loads in the Powder River, "Much of its contribution may be natural. A number of other anthropogenic non-point sources are believed to occur…" There are turbidity standards for the lower Yellowstone. These allow a maximum increase of 10 NTU relative to natural conditions (Table 4-3). The lower Yellowstone has not been assessed as impaired by turbidity, but it is not clear if an analysis of natural turbidity levels in the system has been performed. It would appear most appropriate to evaluate nutrient criteria with turbidity constrained to meet standards – i.e., *the natural turbidity regime plus 10 NTU. At a minimum, the report should discuss these issues and make a case for the selected approach*."

We had debated the reviewer's consideration prior to the publishing of our draft report and came to the conclusion that a large percentage of the sediment load in the river was natural. We did not state why however. Our justification is as follows. First, a fairly large increase in turbidity occurred downstream of the Powder River when in fact there was no flow contribution to account for such changes. Peterson and Porter (2002) note similar findings writing, *"Water turbidity increased two-fold between Y7 (Forsyth) and Y8 (Miles City), downstream from the Bighorn and Tongue River tributary confluences, then increased from 12 NTU at Y9 (Terry) to 24 NTU at Y11 (Glendive), downstream from the Powder River confluence. However, the Powder River was dry prior to and during the time of sampling in late August 2000"*. Given that both studies found similar changes at similar times (i.e., when the Powder River had very little flow), we concluded that a large unaccounted for autochthonous source exists in the lower river. Most likely it is previously deposited sediment from the Powder River. Still it is unclear whether this load is natural or anthropogenic.

The historical description below provides a persuasive argument clarifying DEQ's argument for natural. Vance et al. (2006) indicate that Francois Antoine Laroque, passed through the lower Yellowstone in the early 1800's (prior to Lewis and Clark). He describes, "*The Powder River is here about ¾ acre in breadth, its water middling deep, but it appears to have risen lately as a quantity of leaves and wood was drifting on it…It is amazing how very barren the ground is between this and the less Missouri, nothing can hardly be seen but those Corne de Racquettes (prickly pear cactus). Our horses are nearly starved. There is grass in the woods but none in the plains…The current of the river is very strong and the water so muddy that it is hardly drinkable. The savages say that it is always thus and that is the reason that they call it Powder River; from the quantity of drifting fine sand set in motion by the coast wind which blinds people and dirtys the water.*"

Similarly, on Friday July 30th, 1806, William Clark of the Lewis and Clark expedition noted, "*Here is the first appearance of Birnt hills which I have Seen on this river they are at a distance from the river on the Lard Side…after the rain and wind passed over I proceeded on at 7 Miles passed the enterance of a river the water of which is 100 yds wide, the bead of this river nearly ¼ of a mile this river is Shallow and the water very muddy and of the Colour of the banks a darkish brown. I observe great quantities of red Stone thrown out of this river that from the appearance of the hills at a distance on its lower Side induced me to call this red Stone river. [NB: By a coincidence I found the Indian name Wa ha Sah] as the water was disagreeably muddy I could not Camp on that Side below its mouth.*"

The previous descriptions in our opinion provide convincing evidence that much of the sediment load from the Powder River is natural. After all, it is hard to imagine anthropogenic sources could elevate turbidity above pre-settlement levels by any meaningful amount. To put the magnitude of the load into perspective, NRCS (2009) estimates the current sediment load of the Yellowstone River at Forsyth, MT as being 3,769 ac-ft/yr whereas the Powder River itself has a load of 3,400 ac-ft/yr (nearly the same amount as the entire upper Yellowstone drainage area). So while no formal sediment source assessment exists to quantify the natural and anthropogenic fractions, we feel it is reasonable to conclude that there has always has been a very large natural sediment loading originating from this region and any turbidity that exists during low-flow conditions is likely natural.

"One additional caution regarding the study in general is that the authors take some liberties in reinterpreting numeric criteria from the Administrative Rules of Montana into "more appropriate" forms.

- *Total dissolved gas levels must be ≤ 110 percent of saturation: The Montana administrative code seems to establish a clear limit of 110 percent of saturation. The authors argue (p. 13-15) "the standard is mainly intended to control super-saturation of atmospheric gas below dam spillways… A thorough literature review… shows that fish are tolerant of much higher total gas levels than the state's standard when the gas pressure is driven by oxygen. For example, fish have been found to tolerate DO saturation levels to 300% DO without manifesting [gas bubble] disease… DO supersaturation levels observed in our model runs were never greater than 175% of saturation and were therefore not an endpoint of consideration with respect to gas bubble disease…" In my opinion, this argument is sensible; however, it is not what the rule says. Presumably, a modification to the criterion should be needed to eliminate consideration of meeting the dissolved gas target from the analysis.*

- *Induced variation in pH must be less than 0.5 pH units within the range of 6.5 to 9.0 or without change outside this range: This requirement is also established in the Montana code. The authors (p. 13-12) contend that this is mistaken and should reflect a two-part test (greater than 9 units and induced variation of 0.5) "as pH in the range of 6.5-9.0 is considered harmless to fish and diurnal changes (delta) greater than 0.5 are only unacceptable when they push the pH outside the 6.5-9.0 range." As with total dissolved gas, this argument makes some sense, but appears to be at odds with existing regulations."*

The reviewer is correct that we did not adhere strictly to the letter of Montana law in identifying and applying water quality endpoints in the model. Rather, we applied current science relative to the effects of TDG and pH. In doing so we understand that we may expose ourselves to some criticism. However, we felt (especially the author who is in DEQ Water Quality Standards) that scientifically-based model endpoints are more important than upholding an antiquated standard given the real intent of water quality criteria is to protect the uses. That said, water quality criteria are often updated/changed to reflect the current state of the science with the underlying intent always remaining unchanged; that is the protection of fish and aquatic life. The relative shortcomings of the two currently-adopted criteria in question (e.g., the fact that aquatic life tolerate higher TDG if it is DO driven, and the two-part pH test) will likely be addressed by DEQ in a future triennial review. Thus, it was better to use appropriate TDG and pH endpoints with the anticipation that current criteria are likely to be updated anyway.

1. Please evaluate the sufficiency and appropriateness of the data used to run the model.

"An extensive data collection effort was undertaken to support the modeling. This effort was specifically designed to support QUAL2K application. The data included two 10-day synoptic surveys (August and September 2007) at multiple sites, along with YSI sonde deployment at 20 or so mainstem and tributary sites throughout the summer. All water quality data were collected in accordance with a QAPP. In addition, a variety of historical data were also located and documented, including a synoptic USGS data set from August 2000. Data were also collected via algal growth rate experiments.

Three good synoptic data sets should be sufficient to test, calibrate, and validate and steady-state model such as QUAL2K. Additional inputs, such as climate forcing, are well documented

Estimates of reaeration and SOD are key inputs to QUAL2K modeling and often difficult to disentangle. DEQ used the approximate delta method of McBride and Chapra to estimate reaeration rates from continuous sonde data. The resulting estimates of k_a have 95% confidence limits on the order of about 1-1.5 day^{-1} on mean values from 2-7 day^{-1}. An attempt was made to estimate SOD with in situ chambers, which is the preferred method, but these failed due to the coarse nature of the river substrate that prevented a good seal. Therefore, estimates were instead estimated from incubated cores, resulting in values that are consistent with literature values for sand bottoms (around 0.5 g/m2/d). However, the authors then state that "percent SOD coverage was visually estimated at each field transect", resulting in values of either zero or 5 percent "cover by SOD" by reach. This percent cover operates as a scaling factor on SOD; thus the authors have effectively reduced SOD in the model to near zero. How it is possible to determine SOD cover visually is not explained, as the levels cited are typical of sands, not mucks. It further seems unreasonable that reaches can have 80 – 100 percent cover by algae but zero "cover" by SOD. Thus, SOD may be underestimated in the model. This in turn may introduce some bias into the benthic algae and diurnal DO calibration."

The reviewer is correct that SOD was low in the river but these values originated from actual observations (even though they were cores) and deviation from them would simply not be justified. In characterizing the percentage of the river that was SOD generating, we relied primarily on the substrate characterization in the field. We observed sediment at 11 locations within each sampling transect and used particle size (i.e., fine grained) as a surrogate for SOD generating material (which were characteristic of our core measurements). In all cases, <5% of the channel substrate would qualify as SOD responsive, which is shown in Table 8-7. In many instances none of the sampling locations contained fines. Admittedly, our *n* was small, but observations did generally fit our conceptual understanding of the river, that is it comprises a well-armored cobble/gravel bed with high flow velocities devoid of organics or other SOD generating material. This does not mean that depositional areas/shallow-water environments where higher SOD (mucks) are not present. Review of aerial photography indicates that such areas do exist, primarily behind the Cartersville diversion dam near Forsyth and in braids and oxbows. The overall spatial extent of these areas is small relative to the channel however. Finally, as the reviewer is aware of, SOD is a direct scaling factor on the oxygen mass balance. Assuming respiration, reaeration, and nitrification are reasonably known (which they were), the leftover deficit (which was small) would have to be attributed to SOD.

In regard to the algal cover percentage (80-100%), again we relied on field data. While percent cover is again a subjective measurement, we find no reason to deviate from our original observations. Admittedly, the water was too deep to make a visual assessment in several instances (noted as not visible on the field form), but the presence Chl*a* was verified analytically at nearly all transect sites (even on sands/clays). Lastly, the percentages applied in the model are consistent with diurnal oxygen (DO) profiles of the river. For example, in order to meet the productivity response of the river, an areal coverage of 100% was required.

"Another potential area where data are somewhat weak is in the estimates of groundwater quality. This input is based on wells less than 200 feet deep and within 5 km of the river. The problem is the assumption that well measurements are equivalent to the quality of water that discharges from groundwater to the river. Typically there can be significant amounts of nutrient uptake by sediment bacteria during the seepage process. This, however, appears to constitute only a very small portion of the total nutrient mass balance and so is not a significant cause for concern."

We concur that the groundwater contribution is hard to estimate due to the reasons mentioned by the reviewer. We also suggest though that this is not a major concern due to the following reasons: (1) flow at the upstream boundary encompasses nearly 70% of the inflow to the study reach and groundwater flux comprised less than half of the remaining 30%, (2) estimated groundwater loads (Figure 17-5) were only 4.6% and 1.8% of the soluble N and P supply to the river, and (3) groundwater nutrient concentrations were not considerably high relative to polluted aquifers. So while the values used in the model could be in error due to the reasons mentioned by the reviewer, we feel this would likely result in only a small loading error.

2. Please evaluate the model calibration and validation

"Calibration was performed on the August 2007 dataset with validation on the September 2007 dataset. An additional validation test was undertaken with 2000 USGS data. The calibration was carried out in accordance with a plan and criteria pre-specified in the QAPP for temperature, DO, phytoplankton chlorophyll a, and bottom algae chlorophyll a. The authors are commended for using the approach of pre-specifying criteria, which is consistent with EPA QA recommendations, but often not done in modeling studies. One concern with the approach is that the QAPP criteria are not based on an analysis of the level of precision needed to meet decision needs under a systematic planning approach but rather seem to be mostly derived from literature recommendations. (The QAPP does not actually state the basis for the selection of the criteria). The specified criteria for Relative Error and Root Mean Squared Error are aggressive but feasible for temperature (±5% or 1° C) and dissolved oxygen (±10% or 0.5 mg/L). The targets for chlorophyll a (±10% for phytoplankton and ±20% for bottom algae) are, in my experience, more stringent than is likely to be attainable for models of this type – particularly for bottom algae chlorophyll a, as this is affected by a variety of processes, including grazing, scour, and variability in the carbon:chlorophyll a ratio, that make precise prediction difficult. The QAPP did not specify acceptance criteria for the pH calibration, as pH was not identified as an important decision variable until after development of the model. It would also have been desirable to specify acceptance criteria for the nutrient simulation (e.g., ±25%), but it would not be appropriate to add acceptance criteria after the fact."

The reviewer is correct that we probably did not do enough up-front consideration of model acceptance criteria (i.e. based on the level of precision needed to meet decision needs) but rather relied on the literature. However, the primary reason was that prior precedent does not exist for making these decisions. For example it was unclear (at least to us) what level of precision may be needed to make acceptable decisions (e.g., would the system be very sensitive to nutrient additions, how does the pH respond, etc.). We would have for the most part been relying on professional judgment. We are also in agreement with the reviewer that our state-variable targets were probably too aggressive. In hindsight, it would have been nice to have provided more flexibility in these values, as well as specifying pH and total nutrient targets *a priori*. In this regard, we will now have to work through these considerations in development of the criteria using knowledge about uncertainty and past criteria development efforts.

"Model parameters and rate coefficients adjusted during calibration are clearly documented and compared to literature values – in most cases. For some reason, the literature ranges for algal stoichiometry and various Arrhenius temperature coefficients are cited as "n/a", although citations are available; however, none of these values look to be unreasonable."

The reviewer is correct. Stoichiometry values can be found in Bowie et al. (1985) and have also been recommended by Chapra et al. (2008). We will revise the table to include suggested ranges. In regard to the temperature coefficients, we did not calibrate these values and we will note that in the footnote of the table.

"Results of model calibration and validation (both September 2007 and August 2000) are summarized in Table 10-1, where it is stated that the QAPP criteria are met except for benthic algae. This is not quite correct, as the Relative Error for DO in the 2nd validation is 18.5%, greater than the criterion of ±10%."

Thank you for finding this mistake. We will revise the table and text.

"Most aspects of the model fit appear quite good. One problem area is the nitrogen simulation. While total N is fit well, there are large relative errors in the nitrate and ammonium simulations. The model consistently underestimates observed NH_4-N concentrations, while overestimating NO_2+NO_3-N during the calibration and underestimating it during the validation. The authors suggest that this is mostly due to changes in trophic condition between August and September, but it looks as though there is something else occurring, probably associated with estimated boundary conditions for incremental inflows."

We agree with the first part of this comment and will investigate how minor recalibration to reduce the nitrification rate will influence the simulation (thereby increasing NH_4-N and decreasing NO_2+NO_3-N). We expect that such a change will probably have a greater effect on ammonium than nitrate/nitrite given their comparative concentrations. Relative to the change in trophic condition, we still contend that shift in river photosynthetic response is the most valid hypothesis, more so than the shift in incremental flows and associated boundary conditions as suggested by the reviewer. For example, we made it a point to evaluate different flow and concentration conditions for each period (August and September) for both tributaries and irrigation return flows as described in Section 7. While some of this data was regressed/estimated, it was reasonably similar both months. Likewise, the relative contribution of these sources was small in comparison to the overall headwater boundary condition soluble nutrient load (as previously noted, referring to the fact that the headwater constituted 70% of the available nutrient load to the reach). In our opinion then, the magnitude of such errors would not be sufficient to cause the large difference observed between the two periods. Autotrophic response just simply slowed (e.g., nutrient uptake, diurnal variation in DO and pH, etc.) which combined with other indicators (i.e., algal physiology evaluations, water temperature, daylength, etc.) make us believe the change in photosynthetic response and resulting nutrient uptake was driven by algal senescence.

"In addition to the base QUAL2K model, the authors made use of several related tools. First, they worked cooperatively with Tufts University to develop a new model, AlgaeTransect2K (AT2K) that relates longitudinal QUAL2K model output to lateral benthic algal density. This tool was designed to account for lateral heterogeneity in areas where only the wadeable, nearshore areas have sufficient light to support significant bottom algae growth. It is not entirely clear how well AT2K works when applied essentially as a post-processor to QUAL2K. That is, the QUAL2K model calibration relies on laterally averaged conditions – including the effects of benthic algal growth calculated based on mean depth. As the relationship between depth and light attenuation is not linear it would not seem appropriate to apply AT2K as a post-processor to QUAL2K results; rather the laterally averaged

bottom algae density from AT2K would seem to need to be re-input to QUAL2K in an iterative process until convergence was obtained."

We do not know a good way to characterize the utility of AT2K when applied as a post-processor to QUAL2K other than to suggest the following: (1) simulated areal biomasses when laterally averaged are nearly identical to the lateral average in QUAL2K (meaning both models converge on the same areal biomass) and (2) calibration of both models was done with only a single set of rate coefficients so that the kinetics in each model are identical despite their difference in conceptual representation. That said, the problem described by the reviewer is plausible and illustrates at least one potential concern when dealing with multi-dimensional water quality problems. Transect station-specific computations from AT2K could in fact be theoretically differ from laterally averaged computations in Q2K, especially with regard to spatial differences in river productivity. These differences would be most likely to affect the oxygen and pH mass balances but it seems like the spatial errors cancel otherwise depth- and width- averaged results from the longitudinal model would not be correct. Thus the calibration method employed by DEQ (i.e., adjustment of rates in both models until acceptable agreement in both models was achieved) seems like the most reasonable method and is valid for discerning the spatial detail of periphyton at a given river transect (instead of transfer of forcing or biomass data as suggested by the reviewer).

"The apparently weak fit to observed benthic algae chlorophyll a is of less concern, as this measure is typically highly variable both in space and time. The fact that both the longitudinal and diurnal profiles of DO and pH are well simulated suggests that the algal simulation is acceptable."

We wholeheartedly agree and have made it a point to stress this as part of our response to reviewer 1 (who has a different opinion). Diurnal DO and pH give the true integrated effect of algal community processes which are equally, or perhaps more important, than noisy point algal measurements.

"Several additional minor criticisms regarding the calibration are:

- *The groundwater contribution was treated as the only unknown in the flow balance (p. 7-9). In fact, irrigation lateral return flows are entirely estimated, although a regression relationship is cited. This uncertainty in the estimate of groundwater accrual should be noted."*

We will revise the text on 7-9 to make this more apparent. We had put some text on page 7-8 describing this, and had a footnote on page 7-33, but we will revise the groundwater discussion on 7-9 so it isn't perceived as misleading.

- *"Evaporation losses from the river are modeled as diffuse abstractions, which remove constituent mass as well. DEQ recognized this as an issue, but the model has not yet been modified to allow removal of water only."*

A beta version of Q2K now has this functionality, but at this point it is not practical to apply the new version of the model given the significant effort to reconfigure the report and associated modeling results (even though very little change is expected). Given that evaporation is a very small portion of the water balance (see page 7-9), we feel it is OK to proceed as currently proposed.

3. Please evaluate the model calibration and validation

"Uncertainty in model predictions, as shown by the calibration and validation exercises, is fully acknowledged and discussed in some detail in the text. In addition, Chapter 14 presents an error propagation analysis in which the effect of uncertainty in boundary conditions, model parameters, and rate coefficients on model predictions is examined. This was accomplished through Monte Carlo analysis using QUAL2K-UNCAS, a re-write of the original QUAL2E-UNCAS uncertainty analysis. (This model version does not appear to be publicly available.). Headwater boundary conditions appear to be the most sensitive parameter controlling pH (which is significant, as pH becomes the decision criterion for the upper reach). However, this conclusion would be better supported if sensitivity to irrigation return flows was also evaluated."

The reviewer is correct that UNCAS for QUAL2K is not in the public domain and awaits publication. Contrary to as suggested by the reviewer though, we <u>did</u> evaluate irrigation return flows as part of the UNCAS work in Section 8.0. Confusion about this may result from the fact that the nomenclature of the analysis was not clear. Large irrigation canals were included in the "point source" evaluation whereas lateral return flows were included in the "diffuse source" component. Another thing that may have added to the confusion is that NSC values for these boundary conditions were not in Table 8-1 and 8-2 (because DO, pH, benthic algae, and TN/TP were highly insensitive to their changes). We will add some text in both Sections 8 and Section 14 clarifying this.

"The major problem with the uncertainty analysis is the interpretation of results. These focus on the variance in output for TN and TP as a function of input uncertainty (excluding nutrient loads), which are used to suggest that the confidence limits on the proposed criteria are small. This approach is incomplete. Instead of TN and TP, the authors should be examining the effect of error propagation on response variables used to derive the criteria. For example, if the error propagation analysis resulted in large confidence limits in predicted benthic algal density it would be appropriate to set lower nutrient criteria to account for this uncertainty."

We think this is a perceptive comment and an oversight on our part. We will re-examine the perturbation variance of ecological responses (i.e., by including pH and benthic algae) as part of the final report. We will then use this information to draw better conclusions, if necessary.

4. Please comment on the appropriateness of using response variables, such as chl-a and pH, as model endpoints for numeric criteria derivation, and thus protection of water quality from nutrient pollution. Please comment on the spatial application of different response variables for deriving numeric nutrient criteria (pH was used for the upstream segment while benthic algal biomass was used in the downstream segment).

"The approach of using response variables is wholly appropriate for establishing site-specific nutrient criteria. The response variable analysis (if comprehensive) ensures that factors that actually impair designated uses are controlled to acceptable levels as a result of nutrient limits while protecting against the economic impacts of unnecessarily stringent limits based on generic nutrient concentration objectives. It is important, however, to ensure that all secondary impacts of nutrient concentrations that have a potential to impair uses are considered in this type of approach.

The response variable approach appropriately relies on the most limiting response in each reach. That is, each response variable must be controlled within criterion concentrations and other *appropriate limits. pH is the most limiting response in the upstream segment and benthic algal biomass the most limiting response in the downstream segment; however, the proposed criteria will protect both pH and benthic algal biomass in all analyzed segments of the river. Thus, the approach is appropriate.*

Application of the model was conducted using 14Q10 flows, typical August meteorology, and low-flow tributary boundary conditions. Selection of these conditions is well supported and documented in Chapter 12.

The model predicts that there is additional assimilative capacity for nutrients under current conditions. Therefore, the model was used to evaluate nutrient criteria by simulating nutrient additions of NO3 or soluble reactive P (SRP) that achieve new concentration levels in stream – requiring an iterative procedure. Ten levels of NO3 (with SRP at non-limiting levels) and ten levels of SRP (with NO3 at nonlimiting levels) were tested. Resulting TN and TP concentrations were calculated by the model. Output from each test was compared to nutrient-related criteria or recommendations for DO, pH, benthic algal biomass, total dissolved gas, and TOC. Of these, the benthic algal biomass and TOC targets are recommendations, not standards.

The benthic algal biomass target of 150 mg/m2 chlorophyll a (as an average for the wadeable region) is DEQ's recommendation to protect recreational uses. This is certainly relevant to use support; however, some justification should be provided as to whether 150 mg/m2 as a wadeable zone average is adequate to support aquatic life uses as well as recreational uses – especially in light of recommendations for the Clark Fork of 100 mg/m2 as an average and 150 mg/m2 as a maximum density."

A lower benthic algae standard for the Clark Fork River (100 mg Chla/m^2 as a summer average) was recommended along with a 150 mg Chla/m^2 maximum in the 1990s as part of the Voluntary Nutrient Reduction Program (VNRP). However, estimates at this time were based on limited academic literature, which did not include evaluation of the public's opinion on the matter. Subsequently, Suplee et al. (2009) show that the public majority in the Clark Fork basin (i.e., Missoula) are accepting of average algae levels up to 150 mg Chl*a*/m^2 (but no higher). Thus, we believe that the 150 mg Chla/m^2 benchmark is, on average, appropriate. In regard to aquatic life uses, nutrient criteria are determined according to the most sensitive use. So if aquatic life standards were exceeded according to the model (e.g., pH or DO) they were used in establishing the criteria. We do <u>not</u> think that 150 mg Chla/m^2 impairs aquatic life uses in large rivers whereas it <u>does</u> in wadeable streams due to accrual of decomposing algae in pools (resulting DO minima <5 mg/L).

"TOC was compared to EPA recommendations for treatment thresholds to minimize harmful disinfection byproducts, and a footnote states "primarily we are concerned with whether or not any scenario would push the river over a required treatment threshold…", thus requiring a higher level of TOC removal. While this is related to drinking water uses, it appears to be more of an economic than a use-protection argument. The issue is moot, however, as TOC was not a limiting factor in the determination of assimilative capacity.

As mentioned in my introductory remarks, there are some issues with how the authors have interpreted (or re-interpreted) existing Montana water quality standards for pH and dissolved gas. The dissolved gas criterion would exceed the 110 percent threshold defined in the rule, if it was deemed applicable, and might thus require more stringent nutrient limits; however, the authors argue that this is not appropriate. It is stated (p.13-16) that the nutrient addition runs resulted in dissolved gas concentrations up to 175 percent of saturation; however, full details are not provided.

Regarding pH, this becomes a limiting factor for nutrients primarily because the natural pH of the system seems to be high (> 8.5 at the headwater reach for this analysis); thus only a small increment is needed to push it over the level of 9 standard units. The authors should likely discuss whether there are other anthropogenic causes contributing to elevated pH in the system."

We have already addressed both the total dissolved gas and pH standard interpretation issue earlier in our response (in the introductory remarks). With regard to human-caused factors that may have already elevated the pH of the Yellowstone River, our understanding is that a pH of 8.5 at Forsyth is natural or close to a natural. For example, multi-year monitoring studies show a longitudinal change in pH along the Yellowstone River, from just outside of Yellowstone National Park (median: 7.95) to Livingston (median: 8.0) to Billings (median: 8.2) to Forsyth (median: 8.4) (USGS, 2004). As the reviewer is aware, pH in freshwaters is largely controlled by the carbonate-bicarbonate buffer system (Morel and Hering, 1993) and surface waters in Montana are very often alkaline. Downstream of Billings cretaceous sedimentary rocks underlay the river and contribute to increasing calcium carbonate concentrations that elevate pH (USGS, 2004). In fact, according to the 25^{th} percentile bicarbonate concentration at Forsyth (90 mg/L; USGS, 2004) and open carbonate equilibrium theory (i.e., $H_2CO_3*=10^{-5}$ molar and $pK_{a1}=6.35$), pH should naturally be approximately 8.5 assuming all bicarbonate is geochemically derived (which seems reasonable using the 25^{th} percentile). Finally, the Big Horn River (upstream of the modeled reach) contributes a large proportion of flow to the Yellowstone River and has a median alkalinity of 188 mg/L as $CaCO_3$ (much higher than the Yellowstone River at Livingston, where median alkalinity is 54 mg/L as $CaCO_3$). The Bighorn basin is dominated by rangeland land uses which for the most part are natural. Thus while we cannot say with 100% absolute certainty that pH in our modeled reach is natural, the suggested values are fairly typical for larger rivers and streams in the Yellowstone River basin (median range: 8.1 to 8.5) (Lambing and Cleasby, 2006) and reasonably approximate natural.

5. **What other analytical methods would you suggest for deriving numeric nutrient criteria for the mainstem Yellowstone River?**

"In my opinion, the approach used is the appropriate one for the lower Yellowstone River as it provides a fairly comprehensive evaluation of stressor-response relationships specific to the site. A variety of other methods could also have been attempted. Most of these are summarized in Chapter 15 and would generally result in lower criteria. This is expected because (except for the continuous modeling option) they do not fully account for (or wholly ignore) the site-specific characteristics of the Yellowstone.
Briefly:

- *Literature provides a wide range of potential nutrient criteria values, some lower and some higher than the proposed lower Yellowstone criteria. None of the identified literature sources is fully applicable to a deep, turbid river in the High Plains. General recommendations (such as Dodds, 1997, guidance of 350 µg/L TN and 30 µg/L TP to keep benthic biomass below 150 mg/m2 chlorophyll a can be regarded as a lower bound that might apply if other mitigating factors (turbidity, depth) were not present.*

We agree with this comment and suggest it be referenced to counter reviewer 1's critical review.

- *Reference site approaches are in theory applicable; however, an appropriate unimpacted reference for the Yellowstone does not seem to be available. Setting criteria to an unimpacted reference* *condition would also tend to establish a lower bound level of no anthropogenic effect and not a site-specific estimate of assimilative capacity.*

We initially considered a reference site approach (see the QAPP for further detail) but found that the least impacted location was well upstream of the study reach almost entirely in the Middle Rockies ecoregion. Due to the fact that the site had significantly different character than the reach in question (predominantly because of natural reasons), use of the site was omitted.

- *Level III Ecoregional Criteria recommendations are, in essence, a formal summary of available reference site data. These recommendations are most applicable to wadeable streams and do not take conditions specific to the Yellowstone into account.*

- *Regression analysis is presented by DEQ relating benthic algal chlorophyll a to TN and TP in the Yellowstone. This implicitly takes into account some of the site-specific conditions present in the river. These regressions could be used to predict concentrations at which nuisance levels are exceeded; however, the coefficients of determination are quite low, indicating weak predictive ability. Thus the approach of using a calibrated, mechanistic model is preferable. I do suggest that the authors present a multiple regression analysis of benthic algae as a function of both TN and TP, similar to the equations developed by Dodds on the Clark Fork."*

  As suggested by the reviewer, we will include a multiple regression analysis (with adjusted r^2) in our final report.

- *"Continuous simulation modeling could also be used to provide a more detailed analysis of nutrient and algal dynamics over time in the Yellowstone. This would primarily be of academic interest, as the identification and simulation of critical conditions using the steady state QUAL2K model appears adequate for the purposes of establishing criteria."*

  We agree that a time-variable analysis might be of interest but we will not be pursuing such work given the limited benefit and added complexity. It should be noted that Washington Department of Ecology has just released a beta version of QUAL2Kw with dynamic capability (code from WASP) so this may be a consideration in the future (or for retrospective analysis of the Yellowstone River). Other researchers, i.e., the Water Environment Research Foundation (WERF) are also developing a numeric nutrient criteria toolbox as part of the Link1T11 research proposal (Limnotech, Tufts University, Brown and Caldwell, and others) which will further shed light on such approaches.

SECTION 3.0 - References

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Response to Yellowstone River Peer Review Questions

General Comments

This is a well written report on "Using a computer model to derive numeric nutrient criteria." There are relatively few errors in the draft, which made reviewing clear. The use of multiple sources of information, including a computer model, is a very good idea for establishing nutrient criteria. The many concepts developed and employed in this effort are innovative, well founded, and sound. However, I disagree with the conclusions that model conditions warrant more credibility than other sources of information and that model results should be used to set nutrient criteria for the Yellowstone River.

In summary, my short responses to the questions are:
1. The data used to run, calibrate, and validate the model were appropriate, but not sufficient.
2. Model calibration and validation were not good, because the fit of data to model runs was poor for a key endpoint variable, benthic algal biomass, and many results were biased.
3. The uncertainty of model predictions was problematic because: the model was not validated well for a key endpoint variable; the model was used to extrapolate to nutrient conditions outside the range for which it was calibrated and validated; and the model did not simulate extreme values well.
4. pH and algal biomass response endpoints should be used to establish nutrient criteria. The most sensitive response to a stressor (i.e. nutrients in this case) should be used to establish stressor criteria, even if different response endpoints are most sensitive in different types of habitats (in this case shallow and deep river habitats).
5. The appropriate methods were used to gather information about the development of nutrient criteria, but the results of the computer model were overstated and overweighted in a premature decision on nutrient criteria.

1. Please evaluate the sufficiency and appropriateness of the data used to run the model.

The data used to run, calibrate, and validate the model were appropriate, but not sufficient.

The computer model was designed to measure important response variables, such as benthic algal biomass, pH, and DO. These parameters respond either directly or indirectly to variation in nutrient concentrations and are used in either narrative or numeric water quality criteria in many states. These variables are highly appropriate from the perspective that we want to protect uses of waters. We know enough about nutrients to know the effects of nutrients instream and downstream. With proper research and synthesis of results, we should be able to set nutrient criteria above minimally disturbed conditions without threatening designated uses, such as drinking water, recreational uses and aesthetics, and support of biodiversity. Although we may not be protecting aquatic biodiversity of taxa that are highly sensitive to moderately increased nutrient concentrations in a habitat with nutrients above minimally disturbed condition, presumably those taxa are being protected in other habitats in which minimally disturbed condition is being protected (invoking tiered aquatic life uses). With the knowledge that biodiversity of some nutrient sensitive taxa will not be protected at nutrient concentrations that generate algal biomasses greater than 150 mg chl a m^{-2} and pH and DO standard violations, benthic algal biomass, DO, and pH can be appropriate endpoints for managing nutrients.

The right variables were modeled, measured, and calibrated in the field, but the sample size was low. Many of the key environmental variables were measured in the field, but they were measured at less than 10 locations. This limits the power of the comparison, much as a low sample size limits the statistical power in hypothesis testing. Was the fit or the lack of fit of the model to data due to chance or was it true?

The study should have been designed to have the calibration and validation datasets at the same time of year, perhaps sampling during summers of 2007 and 2008. The differences in temperature and light (day length and sun angle) between August and September could be substantial given they are within range that macroalgae like *Cladophora* are especially sensitive. August and September also have very different algal accumulation histories and processes regulating algal ecology probably differ as a result. Interannual variation in physical and chemical conditions in the Yellowstone River are relatively predictable, because of discharge regulation by snowpack melting, compared to rivers in parts of the country where unpredictable rain events have great effects on discharge and resulting physical and chemical conditions (e.g. light and nutrient concentrations).

Another concern was having sufficient scientific foundation for model coefficients. Admittedly, some knowledge is better than none, but assuming that coefficients developed in lakes or other parts of the country and for different kinds of algae in one condition or another would apply to this location seems premature. Many of the parameters were developed in the 1970s or earlier, not that old is necessarily bad, but it is an indication that few new components were available or were found in the literature for use in the computer model. More field and laboratory research is needed to quantify the parameters being used in processed based models.

2. Please evaluate model calibration and validation.
Model calibration and validation were not good, because the fit of data to model runs was poor for a key endpoint variable, benthic algal biomass, and many results were biased.

Not much change was needed in many model parameters to calibrate the model, but many parameters for benthic algal growth were substantially different between the initial estimate and calibrated value (Tables 9-5, 9-6, and 9-7). Almost no discussion followed on the magnitude of these changes and if they were reasonable.

At least one set of the changes in parameters was relatively easy to evaluate and determine if they were reasonable. The mass ratio of N:P in algal cells is assumed to be 7:1, and in the Yellowstone River was often lower because of the relatively low supply of N versus P in the river. The initial mg N and P per mg algae (subsistence quotas for N and P) for benthic algae were assumed to be 0.7 and 0.1, respectively (Table 9-6).

- The real issue is the relatively large change in one value during calibration and the unrealistic ratio for parameter values resulting from that calibration. The resulting calibration values of parameters for subsistence quotas for N and P were 3.20 mg N and 0.13 mg P, respectively. Even though each of these parameters independently fit within the range of possible values reported in the literature (remembering that one outlier in the literature has great effects on this range), the ratio seems very high for conditions within the Yellowstone River. The resulting mass ratio of subsistence levels of N and P was 3.20:0.13, which is more than 3 times the expected 7:1 ratio and 6 times the 4:1 ratios observed in low N habitats like the Yellowstone.

- Although internal N and P half-saturation constants are substantially different types of parameters than subsistence quotas, both are involved with algal growth, both were changed substantially during calibration, and ratios for both were unusually high.
- The same kinds of problems were noted for the phytoplankton (Table 9-7).
- A confusing issue initial parameter values (e.g. 0.7 mg N or 0.1 mg P per mg algae) indicate 70 and 10% of the algae were composed of N and P. Most of algal mass is carbon, not N or P. Presumably the units or my understanding of what these parameters mean were wrong.

Fit of the model, similarity between predicted and observed conditions, was better for physical than chemical parameters, and better for chemical than biological parameters. QAPP criteria were not met for 1 out of 5 of the parameters assessed (Table 10-1). The variable with poor fit based on RMSE and RE was benthic algal biomass, either by using the Q2K or AT2K model. Since benthic algal biomass was a key response endpoint, and an endpoint for which nutrient criteria were eventually going to be made, it was important that the model predict benthic algal biomass well.

As suggested on page 10-21, I agree that the AT2K model “allows us the ability to gain better information about spatial relationship of biomasses across a river transect,” but I don’t agree that AT2K model predictions were sufficiently accurate for the purposes intended for the modeling effort. High benthic algal biomasses were consistently under-predicted.

During review of figures, I became concerned that deviations between observed conditions and conditions predicted by the model are more serious if they are biased than if they are randomly distributed above and below model predictions. This bias would not be captured in the RMSE and RE statistics for goodness of fit. For example, even though the RE is only 7.3% for TN calibration and 1.38% for validation (Figure 10-7, the model overestimates TN concentrations). The bias in predictions (residual error) is common in many of the nutrient and biological parameters. In most cases, bias was either high or low along the river, but in some cases it systematically switched from high to low, which you could imagine was the case for the August 2000 phytoplankton validation (Figure 11-9). Systematic bias along the river is a concern because habitat conditions change systematically along the river.

The model did not capture extreme conditions well, especially for benthic algae. If there was little variation, the model tended to fit much better than if a parameter varied greatly over the range of nutrient and habitat conditions in the river. For example, diurnal variation in dissolved oxygen and discharge were simulated well by the model, but pH and benthic algal biomass which varied much more than DO and discharge were not simulated well by the model.

The model may not have been able to simulate the high algal biomasses that accumulate in the river. For example in Figure 10-15, the model never predicted algal biomass to be greater than about 70 mg chl a m^{-2}. However, several observations of higher chlorophyll were observed. In addition, most of the observed levels of chlorophyll a were less than 50 mg chl a m^{-2} and fell within a confidence envelop that probably had a width of 40 mg chl a m^{-2}. So it would have been difficult for the model to be wrong when benthic algal biomass was less than 50 mg chl a m^{-2}. When benthic algal biomass was predicted or observed to be greater than 50 mg chl a m^{-2}, only 1 of the 10 prediction/observation points were within the RMSE confidence envelop. Another issue with this model fit analysis is also the skewness of the distribution of observed and predicted values, with most points within 1/6th of the range of potential values (<50 mg chl a m^{-2} with a range of 0-300 mg chl a m^{-2}). Basically, it seems the model was not tested in the range of conditions in which it is intended to be applied.

3. Please comment on uncertainty in the model predictions.

The uncertainty of model predictions was problematic because: the model was not validated well for a key endpoint variable; the model was used to make predictions for nutrient conditions outside the range for which the model was calibrated and validated; and the model did not simulate extreme values well. In particular, the inability of the computer model to simulate extreme values in benthic algal biomass was a concern.

The poor prediction of algal biomass and inability to really evaluate model prediction of pH and other important response variables was discussed above.

A basic tenet of modeling, either statistical or highly calibrated computer models, is limiting extrapolation of results outside the range of conditions in which the model was developed. This model was employed outside the range of conditions for which it was calibrated. Since the computer model performed much worse when applied to September than August conditions, due to likely seasonal effects, wouldn't we also expect the same issues with performance outside the range of nutrient concentrations in which the model was calibrated?

Process based models (i.e. computer models) are theoretically better than statistical models for predicting outside the range of original conditions in which they were calibrated. However, the extent and magnitude of calibration from an initial values used in model is a key issue for using process based models to predict outside the range of calibration. Prediction outside the range of conditions for which either the statistical or process based model was calibrated requires that we know enough about the system and the behavior of the system in the two ranges of conditions (e.g. August versus September, or low and high nutrient concentrations) that we are confident that the models accurately describe behavior of the system. The less that you have to calibrate a model to new conditions to get a good fit, the more confident you can be that the model will perform well in a new set of conditions. The more fundamental the processes are that are simulated in the model and the fewer number of assumptions made for use of the model, the more certain you can be that the model will predict responses well in a set of conditions for which it was not calibrated.

Since there is little evidence that the model did perform well, either calibrating for key endpoints or predicting responses during validation, we should have concerns about accuracy of predictions by the model for ecological responses in higher nutrient concentrations for which the model was tested. In addition, many key parameters in the model were changed greatly during calibration from what were initially thought to be appropriate. So based on model performance, we cannot be certain that it will perform well outside the range of conditions in which it was calibrated, or even within that calibration range for some key parameters.

Many assumptions needed for the model also seemed to reduce credibility of its results. Some assumptions were probably met as well in the Yellowstone River as anywhere. For example, the assumption about the model simulating a steady state equilibrium is certainly more appropriate for rivers like the Yellowstone with snow-melt dominated and relatively predictable hydroperiods versus many other rivers where storm events have dramatic and unpredictable effects on hydroperiod.

Violation of model assumptions by the ecosystem may also explain why the model simulated the ecosystem poorly. Of course assumptions are necessary, but some violations of assumptions or combinations of violations may accumulate explain the unsatisfactory behavior in the model. Here are a few examples:

- The assumption that velocity and channel substratum are "sufficiently well mixed vertically and laterally" (pg 5-8, lines 3-4) may explain why the high algal biomasses were not simulated. If average, versus optimal velocity and substratum were used, that would underestimate the high algal accrual possible in optimal velocity and substratum conditions.
- Why assume dynamic equilibrium between particle re-suspension (drift) and deposition (settling)(pg. 8-20, lines 24-25)?
- Why assume the typical meterological year during a ten year period. For example, to understand the conditions under which problems would arise 1 in 10 years, aren't regional weather patterns a likely cause of those problems. Rather than running a typical meteorological year, shouldn't the 10-year extremes be boundary conditions for a run to understand the effects of less common conditions?

In addition to violation of the assumptions in the model, there may be issues with the analytical foundation of the model to accurately represent ecosystem processes; but I am not sufficiently familiar with the model to make that judgment. For example:

- Were growth patterns and differing spatial resource limitation (density dependence) for macroalgae and microalgae or algal taxa included in the model?
- Space limitation in the model, if I understand it correctly, is not the correct conceptualization of the process that regulates density dependent growth of benthic algae. Developing a more realistic characterization of the processes regulating benthic algal accumulation and density-dependent depletion of nutrients within mats would be very interesting and perhaps improve model predictions. Effects of mixing and diffusion vary greatly between different types of algae that grow in differing nutrient and temperature ranges, such as macroalgae (*Cladophora*) and microalgae (diatoms).
- Was N-fixation included in the model and the potential for N transfer between epiphytic diatoms with cyanobacterial endosymbionts on *Cladophora*? It is possible that *Cladophora* cells close to the substratum take up nutrients and transfer them to younger, actively growing cells in the ends of the filaments suspended in the water column. Only cells at the tips of *Cladophora* filaments reproduce, so they are younger and have fewer

epiphytes than cells at the base of filaments. *Cladophora* cells that are closer to the substratum, having more epiphytes, bacteria, and entrained detritus as well as slower currents, have greater potential for uptake of recycled nutrients in the epiphytic assemblages around them than younger cells in the water column. *Cladophora* does not have complete cross walls between cells, so fluid in cells can theoretically mix between cells, which would be facilitated by the movement and bending of filaments in currents. Thus, nutrient concentrations in the water column may be poor estimators of nutrient availability to *Cladophora,* as well as other benthic algae, because of nutrient entrainment and recycling in the mats.

If many potentially important processes are not included in the model, they may either independently or cumulatively have great effects on model outcome and prediction of extreme conditions and risk of problems required for criteria development.

Another reason for questioning model predictions could be the high nitrogen and phosphorus concentrations that are predicted to generate nuisance blooms of benthic algae: 700 μg TN L^{-1} and 90 μg TP L^{-1} in Unit 3 to prevent pH violations and 1,000 μg TN L^{-1} and 140 μg TP L^{-1} in Unit 4 to prevent nuisance benthic algal problems. Although we know relatively little about nutrient concentrations affecting pH in river, these phosphorus concentrations are many times higher than phosphorus concentrations thought to cause nuisance levels of benthic algal biomass, e.g. greater than 150 mg chl a m^{-2}. Admittedly, there's a great range limiting and saturating nutrient concentrations in the literature, but a 30 μg TP/L benchmark was proposed in the Clark Fork, which is upstream from this location. Why have higher numbers in the larger mainstem of the Yellowstone River? If we assume Leibig's law of the minimum, and nitrogen and light are sufficiently great to allow algae to grow, why wouldn't the marginal habitats of the Yellowstone River generate nuisance algal biomasses at 30 μg TP/L? At least one reason could explain that discrepancy. The reactive portion of the TP may be lower in the Yellowstone River than in smaller streams where nuisance blooms of benthic algae commonly occur at TP concentrations around 30 μg TP/L. The soluble fractions of total nutrient concentrations, assumed to be the most readily available fractions, were very low in the Yellowstone River during low flow conditions (Table 6-6). However, caution should be exercised when assuming only the soluble fraction of TP is bioavailable; mounting evidence indicates that entrained particulate P and N are recycled in benthic algal mats.

The model prediction that low DO is not likely in the Yellowstone River seems reasonable. The Yellowstone River is relatively hydrologically stable, so it is probably not prone to types of extreme low flow events that allow development of low DO with resulting fish kills. Rivers and streams are probably much more susceptible to high pH and fluctuating pH conditions than to low DO; but both phenomena have not been studied sufficiently to understand thoroughly.

4. Please comment on the appropriateness of using response variables, such as chl-a and pH, as model endpoints for numeric criteria derivation, and thus protection of water quality from nutrient pollution. Please comment on the spatial application of different response variables for deriving numeric nutrient criteria (pH was used for the upstream segment while benthic algal biomass was used in the downstream segment).

pH and algal biomass response are appropriate endpoints for justification of nutrient criteria. pH is more directly linked to negative effects on aquatic fauna than nutrient concentrations, so pH is a more proximate threat to a valued ecological attribute. High algal biomass is known be an aesthetic problem in rivers, as established in the great study by Suplee et al. As described above, nutrient criteria above minimally disturbed conditions that prevent nuisance algal accumulations and violation of pH and DO standards may not protect biodiversity of some nutrient-sensitive taxa; however chl a and pH, as well as DO, are appropriate endpoints for protecting designated uses.

The most sensitive response (e.g. chl a, pH, or DO) to a stressor (i.e. nutrients in this case) should be used to establish stressor criteria, even if different response endpoints are the most sensitive in different types of habitats (in this case shallow and deep river habitats). An important goal of environmental management should be protection of ecosystem services. Of course all ecosystem services should not have to be protected in all waters, but appropriate protection is warranted. Montana DEQ and presumably a majority of the people of Montana have supported water quality criteria related to pH and benthic algae. So nutrient concentrations should not be allowed that would generate unacceptable risk of violating the pH and nuisance algal biomass criteria.

The focus on shoreline algal biomass was also appropriate because that is where people most commonly observe the water as they use the resource for recreational purposes.

5. What other analytical methods would you suggest for deriving numeric nutrient criteria for the mainstem Yellowstone River?

The appropriate methods were used to gather information about the development of nutrient criteria, but the results of the computer model were overstated and overweighted in a premature decision on nutrient criteria.

Processed based (computer) models are very informative and valuable, but they are just one line of information. Three basic research approaches can be used to develop numeric nutrient criteria: observing patterns in nature and quantifying relationships between nutrients and key endpoint variables with by statistical models (e.g. regression models); simulating patterns in nature using process-based models; and experiments in controlled environments in which environmental conditions are purposefully manipulated. Each of these methods complement each other. When they all do not agree, then conclusions are suspect. In this case, the predictions of the computer model do not match results of other research based on statistical models and experiments. Even though there are plausible reasons for those discrepancies, there is little reason that the computer model is accurate.

Despite that lack of fit between computer model predictions and measured conditions in the river, during both calibration and validation, the computer model was used. In a simple comparison of accuracy of the computer model predictions of high algal biomass as a result of higher nutrient concentrations (Figure 10-5) and the regression model characterizations between algal biomass and either TN or TP (Figure 15-2), show the regression model warranted more credibility. For the computer model, there was no relationship between algal biomass predicted and the algal biomass observed at stations (Figure 10-5). Plotting these abundances in Figure 10-5 on a log-log scale may have improved the apparent fit, but lack of fit at higher biomasses is likely. Remember the discussion above about lack of data points above 50 mg chl a m^{-2} and poor range of observed conditions. For the regression models, the results were variable but plausible (Figure 15-2). If N:P ratios are low and N limits algal growth, then we'd expect a relationship between algal biomass and TN and not between algal biomass and TP concentration. The range of TP concentrations (and bioavailable P indicated by those concentrations) may have been above the TP concentration considered to have strong effects on benthic algal growth (e.g. 30 μg TP/L). The range of TN concentrations may have crossed the sensitive range and below the limiting nutrient concentration for TN; therefore TN may have been the primary limiting nutrient in the Yellowstone River. Thus, the Montana DEQ got a relationship between TN concentrations and benthic algal biomass, but not TP concentrations and benthic algal biomass. I disagree with the interpretation by Montana DEQ about these relationships. These relationships do show that TN concentrations below 505 μg TN/L should constrain average algal biomass to less than 150 mg chl a m^{-2}, but the lack of significance in the TP algal biomass relationship indicates it should not be used to set a TP criterion. This relationship between TN and algal biomass is really the only evidence in the report for nutrient regulation of benthic algal biomass.

If benthic algal biomass is not simulated accurately by the computer model, can we trust predictions of pH and DO that respond to changes in algal biomass? pH and DO predictions of the computer model were also not validated well because of low sample sizes and ranges of conditions in which the model was calibrated.

Another question develops about whether TP concentrations need to be kept below a TP criterion that would constrain algal biomass, if TN concentrations are below that 505 μg/L; but that question is a policy deeper policy question. If TN is kept below 505 μg/L, then presumably there would not be a response of benthic algae to TP if N is the primary limiting nutrient. However, the 505 TN and 30-60 TP range seem close to what I would expect to be saturating nutrient concentrations. So, a combination of TN and TP criteria would provide double protection against risk of high algal biomass.

Good calibration of models, computer or regression, should not be expected in a river without a good range of nutrient that result in algal problems at some place across the range of nutrient conditions. In habitats in which no algal problems are observed, it is possible that sediments and low light constrain algal accumulation such that nutrients have no effect on instream algal-related conditions. In this case, downstream effects should be the concern/endpoints of criteria. Alternatively, it is possible that most that we know about the asymptotic relationship between nutrient concentrations and algal biomass is not true; or for some other reason, TP concentrations above 50-100 μg TP/L do regulate benthic algal biomass. Then the high nutrient concentrations as those proposed (700 μg TN L^{-1} and 90 μg TP L^{-1} in Unit 3 to prevent pH violations and 1,000 μg TN L^{-1}

and 140 μg TP L^{-1} in Unit 4 to prevent nuisance benthic algal problems) would be appropriate in the Yellowstone River.

Continued research in the form of monitoring of the Yellowstone River, surveys of other large rivers, experimental research, and computer modeling will be needed to develop nutrient criteria that protect ecosystem services of large rivers without overprotection. Continued monitoring in the Yellowstone River will enable assessment of whether nutrient concentrations are increasing and nuisance algal biomasses and high pH are becoming more frequent. This will forewarn managers that nutrient related problems are developing and will provide the additional information needed for better computer and regression models used to establish nutrient criteria. In the report, Montana DEQ did propose continued monitoring and data analysis with one goal being learning more about nutrient effects in the river for potential revision of the proposed nutrient criteria. But will reducing the nutrient criteria, based on new science, be practical politically. Why will the public believe the new science if the old science was not sufficient? Why hurry to have nutrient criteria if there are no known problems? Was this the wrong place to try to develop nutrient criteria for large rivers?

A concerted national effort should be developed and maintained to gather the kind of information needed for developing nutrient criteria in large rivers. Monitoring data as well as experimental results should be gathered and evaluated with statistical models and integrated in processed based models to provide sufficient information for development of nutrient criteria in large rivers. Great similarities exist among the large rivers of the world, such that information learned in multiple rivers should be able to be synthesized and related to other large rivers. Until this information is gathered and analyzed, perhaps the most prudent nutrient management strategy is to try to maintain current conditions if there are no existing problems.

A couple editorial changes worthy of note:
Figure 9-1 makes much more sense to me if Table 8-1 were changed to Table 9-1.
Figures 13-4 and 15-2 were hard to understand because the independent variable (nutrient concentration) was not on the X axis.

Memorandum

To:	**NSTEPS**	**Date:**	**January 10, 2012**
From:		**Subject:**	**Yellowstone River Nutrient Criteria**
Cc:		**Proj. No.**	**100-FFX-T94271-06A7**

Using a Computer Water Quality Model to Derive Numeric Nutrient Criteria, Lower Yellowstone River, MT (Montana DEQ, 2011) provides a comprehensive discussion of Montana Department of Environmental Quality (DEQ) efforts to develop nitrogen and phosphorus criteria for the lower Yellowstone. This is done through the development of a site-specific mechanistic nutrient response model that links nutrient loads to measurable endpoints associated with the support of designated uses in the river. The approach is consistent with EPA guidance on establishing TMDLs to address narrative nutrient criteria, which also results in site-specific objectives.

The result of the study is recommendations on site-specific nutrient criteria for the Lower Yellowstone. The results are truly site-specific as they depend on the conditions present in the Lower Yellowstone and it is not clear that they would be applicable to other, similar waterbodies. The results could serve as a template for the derivation of site-specific criteria for other large rivers; however, the evidently high level of effort required to complete this study may preclude wide application.

In general, the modeling and analysis presented here is well done and adequately documented. There are, however, some specific questions that should be resolved before finalizing the analysis. These are described below.

The site-specific nutrient response approach is attractive for several reasons. As noted by DEQ, there is a lack of reference watersheds for large rivers, and methods appropriate to wadeable streams are not transferable to large rivers. In addition, nutrients themselves (except at extreme concentrations) generally do not directly impair designated uses; instead, it is the secondary effects of elevated nutrients, generally involving algal growth, that lead to use impairment. These secondary effects differ according to site characteristics, such as light availability, residence time, and scour regime, which means that the assimilative capacity of a waterbody for nutrients is inherently site-specific and determined by a variety of co-factors; thus the most economically efficient nutrient criteria should also be site-specific.

DEQ has developed site-specific criteria for the lower Yellowstone that reflect specific characteristics of the basin. Notably, the river is deep and turbid, both of which characteristics reduce light availability and thus also reduce the expression of nutrient impacts through algal growth. In other words, these characteristics of the Yellowstone River serve to increase its assimilative capacity for nutrients.

It is clearly appropriate to consider the hydrologic characteristics of the Yellowstone in developing site-specific criteria. In particular, the amount of flow and depth of the river, which reduce the area in which benthic algae can grow, is a largely natural condition. The case for turbidity is a little less clear. The tributaries of the Yellowstone, especially the Powder River, are believed to be naturally turbid. However, the present day turbidity is also affected by land use practices (silviculture, agriculture, grazing, mineral extraction, etc.). If turbidity is greatly elevated by anthropogenic sources then it would appear inappropriate to count the full effect of high turbidity on reducing algal growth as a "credit" that allows for higher nutrient concentrations.

The report (p. 4-8) says, regarding sediment loads in the Powder River, "Much of its contribution may be natural. A number of other anthropogenic non-point sources are believed to occur…" There are turbidity standards for the lower Yellowstone. These allow a maximum increase of 10 NTU relative to natural conditions (Table 4-3). The lower Yellowstone has not been assessed as impaired by turbidity, but it is not clear if an analysis of natural turbidity levels in the system has been performed. It would appear most appropriate to evaluate nutrient criteria with turbidity constrained to meet standards – i.e., the natural turbidity regime plus 10 NTU. At a minimum, the report should discuss these issues and make a case for the selected approach.

One additional caution regarding the study in general is that the authors take some liberties in reinterpreting numeric criteria from the Administrative Rules of Montana into "more appropriate" forms.

- *Total dissolved gas levels must be ≤ 110 percent of saturation*: The Montana administrative code seems to establish a clear limit of 110 percent of saturation. The authors argue (p. 13-15) "the standard is mainly intended to control super-saturation of atmospheric gas below dam spillways… A thorough literature review… shows that fish are tolerant of much higher total gas levels than the state's standard when the gas pressure is driven by oxygen. For example, fish have been found to tolerate DO saturation levels to 300% DO without manifesting [gas bubble] disease… DO supersaturation levels observed in our model runs were never greater than 175% of saturation and were therefore not an endpoint of consideration with respect to gas bubble disease…" In my opinion, this argument is sensible; however, it is not what the rule says. Presumably, a modification to the criterion should be needed to eliminate consideration of meeting the dissolved gas target from the analysis.
- *Induced variation in pH must be less than 0.5 pH units within the range of 6.5 to 9.0 or without change outside this range*: This requirement is also established in the Montana code. The authors (p. 13-12) contend that this is mistaken and should reflect a two-part test (greater than 9 units and induced variation of 0.5) "as pH in the range of 6.5-9.0 is considered harmless to fish and diurnal changes (delta) greater than 0.5 are only unacceptable when they push the pH outside the 6.5-9.0 range." As with total dissolved gas, this argument makes some sense, but appears to be at odds with existing regulations.

The following comments address specific peer review questions:

1 Sufficiency and Appropriateness of Data

Please evaluate the sufficiency and appropriateness of the data used to run the model.

An extensive data collection effort was undertaken to support the modeling. This effort was specifically designed to support QUAL2K application. The data included two 10-day synoptic surveys (August and September 2007) at multiple sites, along with YSI sonde deployment at 20 or so mainstem and tributary sites throughout the summer. All water quality data were collected in accordance with a QAPP. In addition, a variety of historical data were also located and documented, including a synoptic USGS data set from August 2000. Data were also collected via algal growth rate experiments.

Three good synoptic data sets should be sufficient to test, calibrate, and validate and steady-state model such as QUAL2K. Additional inputs, such as climate forcing, are well documented

Estimates of reaeration and SOD are key inputs to QUAL2K modeling and often difficult to disentangle. DEQ used the approximate delta method of McBride and Chapra to estimate reaeration rates from continuous sonde data. The resulting estimates of k_a have 95% confidence limits on the order of about 1 – 1.5 day^{-1} on mean values from 2 – 7 day^{-1}. An attempt was made to estimate SOD with *in situ* chambers, which is the preferred method, but these failed due to the coarse nature of the river substrate that prevented a good seal. Therefore, estimates were instead estimated from incubated cores, resulting in values that are consistent with literature values for sand bottoms (around 0.5 $g/m^2/d$). However, the authors then state that “percent SOD coverage was visually estimated at each field transect”, resulting in values of either zero or 5 percent “cover by SOD” by reach. This percent cover operates as a scaling factor on SOD; thus the authors have effectively reduced SOD in the model to near zero. How it is possible to determine SOD cover visually is not explained, as the levels cited are typical of sands, not mucks. It further seems unreasonable that reaches can have 80 – 100 percent cover by algae but zero “cover” by SOD. Thus, SOD may be underestimated in the model. This in turn may introduce some bias into the benthic algae and diurnal DO calibration.

Another potential area where data are somewhat weak is in the estimates of groundwater quality. This input is based on wells less than 200 feet deep and within 5 km of the river. The problem is the assumption that well measurements are equivalent to the quality of water that discharges from groundwater to the river. Typically there can be significant amounts of nutrient uptake by sediment bacteria during the seepage process. This, however, appears to constitute only a very small portion of the total nutrient mass balance and so is not a significant cause for concern.

2 Model Calibration and Validation

Please evaluate model calibration and validation.

Calibration was performed on the August 2007 dataset with validation on the September 2007 dataset. An additional validation test was undertaken with 2000 USGS data. The calibration was carried out in accordance with a plan and criteria pre-specified in the QAPP for temperature, DO, phytoplankton chlorophyll *a*, and bottom algae chlorophyll *a*. The authors are commended for using the approach of pre-specifying criteria, which is consistent with EPA QA recommendations, but often not done in modeling studies. One concern with the approach is that the QAPP criteria are not based on an analysis of the level of precision needed to meet decision needs under a systematic planning approach but rather seem to be mostly derived from literature recommendations. (The QAPP does not actually state the basis for the selection of the criteria). The specified criteria for Relative Error and Root Mean Squared Error are aggressive but feasible for temperature (±5% or 1° C) and dissolved oxygen (±10% or 0.5 mg/L). The targets for chlorophyll *a* (±10% for phytoplankton and ±20% for bottom algae) are, in my experience, more stringent than is likely to be attainable for models of this type – particularly for bottom algae chlorophyll *a*, as this is affected by a variety of processes, including grazing, scour, and variability in the carbon:chlorophyll *a* ratio, that make precise prediction difficult. The QAPP did not specify acceptance criteria for the pH calibration, as pH was not identified as an important decision variable until after development of the model. It would also have been desirable to specify acceptance criteria for the nutrient simulation (e.g., ±25%), but it would not be appropriate to add acceptance criteria after the fact.

Model parameters and rate coefficients adjusted during calibration are clearly documented and compared to literature values – in most cases. For some reason, the literature ranges for algal stoichiometry and various Arrhenius temperature coefficients are cited as "n/a", although citations are available; however, none of these values look to be unreasonable.

Results of model calibration and validation (both September 2007 and August 2000) are summarized in Table 10-1, where it is stated that the QAPP criteria are met except for benthic algae. This is not quite correct, as the Relative Error for DO in the 2nd validation is 18.5%, greater than the criterion of ±10%.

Most aspects of the model fit appear quite good. One problem area is the nitrogen simulation. While total N is fit well, there are large relative errors in the nitrate and ammonium simulations. The model consistently underestimates observed NH_4-N concentrations, while overestimating NO_2+NO_3-N during the calibration and underestimating it during the validation. The authors suggest that this is mostly due to changes in trophic condition between August and September, but it looks as though there is something else occurring, probably associated with estimated boundary conditions for incremental inflows.

In addition to the base QUAL2K model, the authors made use of several related tools. First, they worked cooperatively with Tufts University to develop a new model, AlgaeTransect2K (AT2K) that relates longitudinal QUAL2K model output to lateral benthic algal density. This tool was designed to account for lateral heterogeneity in areas where only the wadeable, nearshore areas have sufficient light to support significant bottom algae growth.

It is not entirely clear how well AT2K works when applied essentially as a post-processor to QUAL2K. That is, the QUAL2K model calibration relies on laterally averaged conditions – including the effects of benthic algal growth calculated based on mean depth. As the relationship between depth and light attenuation is not linear it would not seem appropriate to apply AT2K as a post-processor to QUAL2K results; rather the laterally averaged bottom algae density from AT2K would seem to need to be re-input to QUAL2K in an iterative process until convergence was obtained.

The apparently weak fit to observed benthic algae chlorophyll *a* is of less concern, as this measure is typically highly variable both in space and time. The fact that both the longitudinal and diurnal profiles of DO and pH are well simulated suggests that the algal simulation is acceptable.

Several additional minor criticisms regarding the calibration are:

- The groundwater contribution was treated as the only unknown in the flow balance (p. 7-9). In fact, irrigation lateral return flows are entirely estimated, although a regression relationship is cited. This uncertainty in the estimate of groundwater accrual should be noted.
- Evaporation losses from the river are modeled as diffuse abstractions, which remove constituent mass as well. DEQ recognized this as an issue, but the model has not yet been modified to allow removal of water only.

3 Uncertainty in Model Predictions

Please comment on uncertainty in the model predictions.

Uncertainty in model predictions, as shown by the calibration and validation exercises, is fully acknowledged and discussed in some detail in the text. In addition, Chapter 14 presents an error propagation analysis in which the effect of uncertainty in boundary conditions, model parameters, and rate coefficients on model predictions is examined. This was accomplished through Monte Carlo analysis using QUAL2K-UNCAS, a re-write of the original QUAL2E-UNCAS uncertainty analysis. (This model version does not appear to be publicly available.). Headwater boundary conditions appear to be the most sensitive parameter controlling pH (which is significant, as pH becomes the decision criterion for the upper reach). However, this conclusion would be better supported if sensitivity to irrigation return flows was also evaluated.

The major problem with the uncertainty analysis is the interpretation of results. These focus on the variance in output for TN and TP as a function of input uncertainty (excluding nutrient loads), which are used to suggest that the confidence limits on the proposed criteria are small. This approach is incomplete. Instead of TN and TP, the authors should be examining the effect of error propagation on response variables used to derive the criteria. For example, if the error propagation analysis resulted in large confidence limits in predicted benthic algal density it would be appropriate to set lower nutrient criteria to account for this uncertainty.

4 Use of Response Variables

Please comment on the appropriateness of using response variables, such as chl-a and pH, as model endpoints for numeric criteria derivation, and thus protection of water quality from nutrient pollution. Please comment on the spatial application of different response variables for deriving numeric nutrient criteria (pH was used for the upstream segment while benthic algal biomass was used in the downstream segment).

The approach of using response variables is wholly appropriate for establishing site-specific nutrient criteria. The response variable analysis (if comprehensive) ensures that factors that actually impair designated uses are controlled to acceptable levels as a result of nutrient limits while protecting against the economic impacts of unnecessarily stringent limits based on generic nutrient concentration objectives. It is important, however, to ensure that all secondary impacts of nutrient concentrations that have a potential to impair uses are considered in this type of approach.

The response variable approach appropriately relies on the most limiting response in each reach. That is, each response variable must be controlled within criterion concentrations and other appropriate limits. pH is the most limiting response in the upstream segment and benthic algal biomass the most limiting response in the downstream segment; however, the proposed criteria will protect both pH and benthic algal biomass in all analyzed segments of the river. Thus, the approach is appropriate.

Application of the model was conducted using 14Q10 flows, typical August meteorology, and low-flow tributary boundary conditions. Selection of these conditions is well supported and documented in Chapter 12.

The model predicts that there is additional assimilative capacity for nutrients under current conditions. Therefore, the model was used to evaluate nutrient criteria by simulating nutrient additions of NO_3 or soluble reactive P (SRP) that achieve new concentration levels in stream – requiring an iterative procedure. Ten levels of NO_3 (with SRP at non-limiting levels) and ten levels of SRP (with NO_3 at non-limiting levels) were tested. Resulting TN and TP concentrations were calculated by the model. Output from each test was compared to nutrient-related criteria or recommendations for DO, pH, benthic algal biomass, total dissolved gas, and TOC. Of these, the benthic algal biomass and TOC targets are recommendations, not standards.

The benthic algal biomass target of 150 mg/m^2 chlorophyll *a* (as an average for the wadeable region) is DEQ's recommendation to protect recreational uses. This is certainly relevant to use support; however, some justification should be provided as to whether 150 mg/m^2 as a wadeable zone average is adequate to support aquatic life uses as well as recreational uses – especially in light of recommendations for the Clark Fork of 100 mg/m^2 as an average and 150 mg/m^2 as a *maximum* density.

TOC was compared to EPA recommendations for treatment thresholds to minimize harmful disinfection byproducts, and a footnote states "primarily we are concerned with whether or not any scenario would push the river over a required treatment threshold…", thus requiring a higher level of TOC removal. While this is related to drinking water uses, it appears to be more of an economic than a use-protection argument. The issue is moot, however, as TOC was not a limiting factor in the determination of assimilative capacity.

As mentioned in my introductory remarks, there are some issues with how the authors have interpreted (or re-interpreted) existing Montana water quality standards for pH and dissolved gas. The dissolved gas criterion would exceed the 110 percent threshold defined in the rule, if it was deemed applicable, and might thus require more stringent nutrient limits; however, the authors argue that this is not appropriate. It is stated (p.13-16) that the nutrient addition runs resulted in dissolved gas concentrations up to 175 percent of saturation; however, full details are not provided.

Regarding pH, this becomes a limiting factor for nutrients primarily because the natural pH of the system seems to be high (> 8.5 at the headwater reach for this analysis); thus only a small increment is needed to push it over the level of 9 standard units. The authors should likely discuss whether there are other anthropogenic causes contributing to elevated pH in the system.

5 Other Analytical Methods for Deriving Numeric Nutrient Criteria for the Mainstem Yellowstone

What other analytical methods would you suggest for deriving numeric nutrient criteria for the mainstem Yellowstone River?

In my opinion, the approach used is the appropriate one for the lower Yellowstone River as it provides a fairly comprehensive evaluation of stressor-response relationships specific to the site. A variety of other methods could also have been attempted. Most of these are summarized in Chapter 15 and would generally result in lower criteria. This is expected because (except for the continuous modeling option) they do not fully account for (or wholly ignore) the site-specific characteristics of the Yellowstone. Briefly:

- Literature provides a wide range of potential nutrient criteria values, some lower and some higher than the proposed lower Yellowstone criteria. None of the identified literature sources is fully applicable to a deep, turbid river in the High Plains. General recommendations (such as Dodds, 1997, guidance of 350 µg/L TN and 30 µg/L TP to keep benthic biomass below 150 mg/m^2 chlorophyll *a* can be regarded as a lower bound that might apply if other mitigating factors (turbidity, depth) were not present.
- Reference site approaches are in theory applicable; however, an appropriate unimpacted reference for the Yellowstone does not seem to be available. Setting criteria to an unimpacted reference condition would also tend to establish a lower bound level of no anthropogenic effect and not a site-specific estimate of assimilative capacity.
- Level III Ecoregional Criteria recommendations are, in essence, a formal summary of available reference site data. These recommendations are most applicable to wadeable streams and do not take conditions specific to the Yellowstone into account.
- Regression analysis is presented by DEQ relating benthic algal chlorophyll *a* to TN and TP in the Yellowstone. This implicitly takes into account some of the site-specific conditions present in the river. These regressions could be used to predict concentrations at which nuisance levels are exceeded; however, the coefficients of determination are quite low, indicating weak predictive

ability. Thus the approach of using a calibrated, mechanistic model is preferable. I do suggest that the authors present a multiple regression analysis of benthic algae as a function of both TN and TP, similar to the equations developed by Dodds on the Clark Fork.

- Continuous simulation modeling could also be used to provide a more detailed analysis of nutrient and algal dynamics over time in the Yellowstone. This would primarily be of academic interest, as the identification and simulation of critical conditions using the steady state QUAL2K model appears adequate for the purposes of establishing criteria.

March 28, 2014

MT Department of Environmental Quality
Board of Environmental Review
c/o Ms. Elois Johnson
P.O. Box 200901
Helena, MT 59620-0901

MT Department of Environmental Quality
c/o Ms. Carrie Greeley
P.O. Box 200901
Helena, MT 59620-0901

RE: Statewide Numeric Nutrient Standards - Comments

To Whom it May Concern:

Stillwater Mining Company (Stillwater) offers the following comments on the proposed amendments to the Administrative Rules of Montana with respect to the numeric nutrient criteria contained in draft Circulars DEQ-12A and DEQ-12B, as well as the draft implementation guidance document. Stillwater has operating mines in Stillwater and Sweetgrass Counties along with an operating Smelter and Base Metals Refinery in Columbus. As you'll note, many of our comments will echo those expressed by the Montana Petroleum Association, Montana Mining Association, and other vested dischargers and interested members of the public.

Stillwater currently employs over 1,700 Montanans in addition to numerous independent contractors, consultants, and supporting industries making Stillwater the largest industrial employer in the State. On average, Stillwater's annual expenses exceed $750,000,000, a significant portion of which is spent in the State. In addition, during 2012, Stillwater paid over $17,500,000 in taxes including $7,400,000 to Stillwater County; $3,300,000 to Sweetgrass County; and $6,800,000 to the State of Montana.

Stillwater maintains MPDES discharge permits at the Stillwater and East Boulder mines, as well as MPDES stormwater permits at both mines and the processing facilities. The proposed numeric nutrient criteria and associated variance process will directly impact Stillwater's operations at all three locations, and therefore, Stillwater has a vested interest in the development of reasonable nutrient standards and associated rules.

1. Stillwater recognizes and appreciates the fact that the Montana Code requires promulgation of a rule establishing base numeric nutrient standards and that the Board and Montana Department of Environmental Quality (Department) have a non-discretionary duty to do so. However, Stillwater also recognizes the efforts in the 2011 Legislature in SB-367 to create authority for the Department to grant variances for point source dischargers for nitrogen and phosphorous limits in numeric nutrient stands which cannot be met given existing technology. The limits of technology and the fact that the technology is not cost-effective were the bases for the Legislature's

242 South Diamond
P.O. Box 1330
Columbus, Montana 59019

STILLWATER
MINING COMPANY

telephone 406.322.8700
facsimile 406.322.8841
www.stillwatermining.com

decision to adopt variances. It is our interpretation that SB-367 provided that if a discharger, compliant with the caps set in Section 75-5-313(5)(b)(i) and (ii), MCA, cannot meet the applicable numerical nutrients standard, the discharger will be granted a variance. Without the variances, substantial and widespread economic impacts would result if Montana Law required immediate compliance with numeric nutrient limits imposed by the new standards.

Stillwater further recognizes and appreciates the Department's efforts to work with all stakeholders, including industry, to develop rules and guidance to implement the provisions of SB-367. Significant time and effort was spent in evaluating and clarifying the effects of the proposed numeric nutrient criteria on 'new or expanded' dischargers, specifically in cases where proposed new or increased point sources are subject to Montana's non-degradation rules, Section 75-5-303, MCA. However, Stillwater does offer the following comments specific to the Department's Authority on Variances.

- The Department has included the following comment on this issue within the Base Numeric Nutrient Standards Implementation Guidance document: "The provisions for general, individual, and alternative variances in section 75-5-313, MCA, are available to all discharge permit holders and are not limited to dischargers under permit on the effective dates of DEQ Circular DEQ-12A or DEQ Circular DEQ-12B." Stillwater appreciates the Department's effort to clarify the availability of General Variances to all discharge permit holders in the Guidance document, but this provides a lesser degree of regulatory and legal protection than inclusion of the same statement in the Administrative Rules of Montana and DEQ-12B. For this reason, Stillwater requests the aforementioned variance language in the Implementation Guidance also be included within the Administrative Rules of Montana and DEQ Circular DEQ-12B; or at a minimum, the Department state for the record that the Department's position on issuance of the General Variance will be the same for private and public entities and that the General Variance will be available to new and increased discharges on the same basis as for existing permit holders.

- On pages 10 and 11 of DEQ-12, the Department describes the rationale for amending the rule as being required, in part, to "incorporate the nutrient standards variance limits". Stillwater does not believe that the draft language is accurate. Rather, Stillwater recommends that the Board modify the language in all three sections to strike "nutrient standards variance limits" and replace it with "the Department's authority to grant variances from the numeric standards for permittees."

- In describing the inability to meet proposed numeric nutrient criteria, the Department's draft refers to the inability of permittees to meet the numeric concentrations imposed by the new standards as a problem which would arise "in many cases". The use of "many" is inappropriate in this context. It is clear from the actions of the Legislature and the plain language of SB-367 that "most" or "virtually all" should be insert in the place of "many" in describing the reason for the adoption of the draft rule. In addition, the Department has written that the "statute allows dischargers to be granted variances from base numeric nutrient standards in those cases where meeting the standards today would be an unreasonable economic burden or technologically infeasible." This should be rewritten to reflect that "the statute requires the Department to grant general variances from base numeric nutrient standards in those cases where meeting the standards today would be an unreasonable economic burden or technologically infeasible and the permittee meets the end-of-pipe treatment requirements in DEQ-12B."

2. On page 7 of DEQ-12, the Department proposes to add a section 2 to ARM 17.30.619 as a non-severability clause. It is recognized that the general variance provision internalized in the rule to be promulgated by the Department and amplified in DEQ-12B will be of no effect if, after promulgation of the rule, EPA disallows a permit with a general variance for the reason that the Department allowed the permittee to deviate from the numeric nutrients standards based upon the application of a general variance. The 2011 Legislature, without opposition from EPA, used mandatory language in Mont. Code Ann. § 75-5-313(5)(b) to require the Department to incorporate a general variance in permits if the permit applicant meets certain conditions. If EPA, in turn, refuses to allow a permit with a general variance to take effect as a result of the inclusion of the variance, the intent of the statute has been nullified with respect to the permittee. In such a circumstance, the rules should not continue to bind permittees. Therefore, Stillwater asks the Board to amend the language employed by the Department in the rule as noted in the italicized language as follows:

> If (1) a court of competent jurisdiction declares 75-5-313, MCA, or any portion of that statute invalid, (2) the United States Environmental Protection Agency disapproves 75-5-313, MCA, or any portion of that statute, under 30 CFR 131.21, or if rules adopted pursuant to 75-5-313(6) or (7), MCA, expire and general variances are not available, *or (3) after the date of the promulgation of this rule, the United States environmental protection agency nullifies or otherwise disallows a permit with a general variance issued by the Department based upon the Department's inclusion of a general variance in the permit*, then (1)(e) and all references to DEQ-12A, base numeric nutrient standards and nutrient standards variances in ARM 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.622 through 17.30.629, 17.30.635, 17.30.702, and 17.30.715 are void, and the narrative water quality standards contained in ARM 17.30.637 are the standards for total nitrogen and total phosphorus in surface water, except for the Clark Fork River, for which the standards are the numeric standards in ARM 17.30.631.

Without the addition of this language to the rule, the rule will remain in force if EPA rejects a permit with a general variance for the permittee because EPA does not believe the permittee is entitled to a general variance.

3. Stillwater requests that language be added to DEQ-12A that future violations of numeric nutrients standards should only be considered in context to the "nuisance level threshold" for algae in stream, at that time. In short, a violation of the numeric nutrients standards should only be considered in combination with the the amount of *chlorophyll a* in-stream, in determining the site-specific and potential impact of water quality exceedences, thus allowing for the numerous site-specific and seasonal variations and assimilative capacity of the stream to be taken into account.

4. It is recognized that Montana is among a small number of states which have moved to adopt numeric nutrient standards for rivers and streams. At present, none of our neighboring states have adopted numeric nutrient standards, and these states, among many others, have retained narrative standards for nutrients because they remain legally viable under federal law. With Montana's leadership role comes an added responsibility to guarantee that all of the associated regulations are accurately updated to ensure a regulatory process that continues to function smoothly and effectively.

   Stillwater, as a leading industrial employer in the state, is generally concerned about Montana taking a leading role, ahead of most of the United States, in implementation of numeric nutrient standards that are not achievable by current water treatment technology. If not managed properly, this could hinder economic growth and/or a protracted legal battle, as demonstrated by the state of Florida recently. Specifically, without carefully planning and addressing all implementation issues, this action could affect natural resource development; economic growth in Montana's cities, towns, and counties; as well as the agricultural industry. EPA's own Science Advisory Board (SAB) in 2009 advised EPA that "numeric nutrient criteria developed and implemented without consideration of system specific conditions can lead to management actions that may have negative social and economic and unintended environmental consequences without additional environmental protection."

5. Stillwater remains concerned about how the Department will analyze whether downstream uses are adequately protected when an applicant seeks a variance based upon water quality modeling. In principle, Stillwater tends to agree with comments submitted by the League of Cities and Towns, in which the League noted:

   > The reference to "protection of downstream use" should be removed from the proposed documents or use language similar to the following: "dischargers shall only be responsible for the protection of downstream use to the first location of a non-point source loading". Without defining the extent a point source discharger is responsible for protection of downstream use and without recognition of non-point source contribution, the language is not acceptable.

   Unfortunately, the lack of clarity on this point has continued through the development of the rule package. In fact, in the guidance document, the Department states, "[a]ny reach-specific criteria developed for a receiving stream using a mechanistic or empirical model will also need to protect downstream beneficial uses. … "How far downstream" is a consideration which will vary from case-to-case...." It is problematic to promulgate the rule packages without a better idea of the touchstones for the Department's analysis because parties are left to their own devices to determine whether the answer is the point of the next discharge downstream or the Gulf of Mexico.

6. Stillwater is concerned that the overarching problem of non-degradation has not been addressed by Rule and urges the Department and Board to address the non-degradation issue prior to finalization of these rules and DEQ-12B. Stillwater believes this is appropriate as the intent of SB-367 was not to establish a variance system only to have it nullified by the non-degradation review process.

We appreciate and express our gratitude to the members of the Nutrient Working Group and the staff and officials in the Department of Environmental Quality for their significant efforts and time in developing the draft numeric nutrient standards and circulars. However, Stillwater is generally in opposition to promulgation of the proposed numeric nutrient rules at this time, noting that there remain significant 'unknowns' or 'uncertainties' about how the new rules will be implemented and their impact on both private and public dischargers. If proposed nutrient rules and standards are adopted, Stillwater believes these comments need to be considered and acted upon in advance. As such, we look forward to continuing to work with the Department in resolution of these concerns and the successful development and implementation of numeric nutrient standards.

Respectfully,

Randy Weimer

Randy Weimer
Corporate Manager – Environmental and Governmental Affairs
Stillwater Mining Company

/rw

Cc: Mr. Bruce Gilbert, SMC
Mr. Matt Wolfe, SMC
Mr. David Johnson, SMC

From: Suplee, Mike
To: Greeley, Carrie
Subject: FW: proposed amendments to adopt Circular DEQ-12A
Date: Tuesday, April 01, 2014 1:51:14 PM

From: Robin Steinkraus [mailto:lakers@flatheadlakers.org]
Sent: Tuesday, April 01, 2014 1:19 PM
To: Johnson, Elois
Cc: Suplee, Mike; Mathieus, George; Jason Gildea; Laidlaw.tina@Epa.gov
Subject: proposed amendments to adopt Circular DEQ-12A

Elois Johnson,

Attached is the Flathead Lakers' comment letter regarding the proposed amendment to adopt Department Circular DEQ-12A base nutrient standards for total nitrogen and total phosphorus. I am also mailing a hard copy of this letter to you today. Thank you.

Robin

Robin Steinkraus, Executive Director
Flathead Lakers
PO Box 70
Polson, MT 59860
406-883-1346

P.O. Box 70 · Polson, MT 59860
(406) 883-1346
Fax (406) 883-1357
lakers@flatheadlakers.org
www.flatheadlakers.org

FLATHEAD LAKERS

Flathead Lakers:
Working for clean water, a healthy ecosystem, and lasting quality of life in the Flathead Watershed

March 31, 2014

Robin Shropshire, Chair
Montana Board of Environmental Review
c/o Elois Johnson
Department of Environmental Quality
P.O. Box 200901
Helena, Montana 59620-0901

Re: Proposed amendment to adopt Department Circular DEQ-12A base nutrient standards for total nitrogen and total phosphorus

Dear Chairman Shropshire and Montana Board of Environmental Review:

The Flathead Lakers support adoption and implementation of the proposed base numeric nutrient standards for total nitrogen and total phosphorus described in Department Circular DEQ-12A.

The Flathead Lakers is a nonprofit organization working for clean water, healthy ecosystems and lasting quality of life in the Flathead Watershed. Founded in 1958, our organization currently has 1,500 members.

Our organization has long supported the Flathead Basin Water Quality Monitoring Program and we have been engaged in the development of Flathead Lake and watershed TMDLs since the 1990s. Our education, advocacy, and stewardship programs are aimed toward protecting water quality in Flathead Lake and throughout its watershed and improving water quality through reductions in point and nonpoint source pollution.

We appreciate the Nutrient Work Group and Montana Department of Environmental Quality (DEQ) staff's diligence, proficiency, and perseverance in developing the proposed nutrient standards for wadeable streams and rivers, including the technical and scientific analysis that provides the foundation for the standards. The proposed standards will be critical for ensuring that the Flathead Watershed's, as well as all of Montana's, vital streams and rivers remain clean and healthy or are restored to health to support their many beneficial uses critical for healthy ecosystems and economies.

We are disappointed that DEQ did not provide the technical and scientific basis for the proposed Flathead Lake numeric nutrient standards to the public and the Board for review prior to the March 24 Board hearing and that the agency did not recommend that the Board adopt those standards at this time. We continue to support adoption of the 2001 Flathead Lake TMDL Phase I targets for nutrient loading in Flathead Lake as the Flathead Lake nutrient standards. However, a document describing the technical and scientific support for those standards is needed.

The explanation given by DEQ Water Quality Specialist Dr. Michael Suplee at the Board hearing for his request to postpone adopting the nutrient standards proposed for Flathead Lake was related to concerns raised about changes in lake conditions since the 2001 Flathead Lake TMDL Phase I nutrient target levels were set. We wish to point out that although current water quality is a consideration in setting water quality standards, the Clean Water Act requires that standards be based on what is necessary to achieve and maintain designated beneficial uses. It is important that the state provide technical justification for the standards on this basis so that there is no delay in EPA approval.

The Flathead Lakers support the adoption of the proposed numeric nutrient standards for wadeable streams and rivers. Thank you for your consideration.

Sincerely,

Greg

Greg McCormick
President

Robin

Robin Steinkraus
Executive Director

cc: Dr. Michael Suplee, DEQ
George Mathieus, DEQ
Jason Gildea, EPA
Tina Laidlaw, EPA

From: North, John
To: DEQ WQP Admin
Subject: Small Business Economic Analysis
Date: Tuesday, April 01, 2014 4:51:59 PM

Board of Environmental Review:

Attached is the small business economic impact analysis prepare be the Department's economist for the numeric nutrient standards in MAR Notice No. 17-356. Although we do not believe that the variance rule contained in MAR Notice 17-355 requires a small business economic analysis, the effects of the variance rule are evaluated in the attached analysis because the standards do not become effective until the variances are effective, and therefore the economic effect must be determined based on the combined operation of the standards and the variances.

John North
Chief Legal Counsel
Department of Environmental Quality

Small Business Impact Analysis of the Nutrient Standards Rule

The Nutrient Standards Rule (MAR Notice No. 17-356, Circular DEQ 12A) and Variance Rule (MAR Notice No. 17-355, Circular 12B) will significantly and directly impact a couple of small businesses in the state of Montana. For the purposes of this study, small businesses are defined as less than 50 employees. An estimated two existing small businesses will see larger cost impacts from having to meet higher water quality standards. These two businesses are located in a small town and rural area in Montana. The two businesses that might have to upgrade would likely experience significant impacts from the rule.

Impacts

Direct Costs to Two Existing Small Businesses to Meet WERF Level 2 and Level 3[1]

Out of the thousands of businesses in Montana, about 50 were identified as ones that would be affected directly by the nutrient criteria. This identification was done in the document entitled, " Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met by Entities in the Private Sector in 2011/2012". Included were businesses that have a discharge permit into state waters, and are not otherwise hooked up to a municipal system. Therefore, the numeric nutrient water quality standards only apply to those few business entities that have a surface water discharge permit and treat their own water. Out of these 50 businesses, only about 4 or 5 would qualify as small businesses (less than 50 employees) that would be directly affected by this rule. Indeed, most of the 50 businesses are mines, refineries, natural gas companies and other large entities over 50 employees. Only two very small businesses out of the four were found to have treatment levels that do not currently meet WERF Level 2 and thus the general variance level allowed at this time under the Variance rule. Thus, only two existing small businesses in Montana would likely have to upgrade their self-owned wastewater treatment systems to meet WERF 2 (or find alternative forms of water disposal). The other two or three small businesses were estimated to be already meeting WERF level 3 and 4, or meeting higher standards than the general variance.

DEQ examined the wastewater permits and the Statement of Basis for these 2 affected businesses. These records are located within DEQ's Permitting Division. Current effluent nutrient levels and estimates of current treatment costs at the two businesses were compared to costs that would be needed to meet WERF levels 2 and 3 based on the WERF study. In this way, annual capital and operations costs needed for meeting base nutrient criteria (above current nutrient treatment costs) were applied to each business.

[1] Much of this section is taken from From 'Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met by Entities in the Private Sector in 2011/2012'

Costs are under-estimated for small facilities and those with low flows, because the WERF cost data was multiplied by effluent flow providing a linear cost estimate based on flow. Clearly, there will be a minimum capital cost of treating to WERF level 2 for facilities with small flows such as pouring concrete, hiring labor, etc. that is greater than the linear cost estimates for these low-flow and small facilities. Thus, we multiply the WERF cost numbers by 5 to account for a lack of economies of scale in these two businesses.

An analysis of the life-cycle costs for a number of technologies used to control nitrogen and phosphorus in wastewater treatment plants estimated that labor costs are between 15-21 percent of the annualized capital costs for nitrogen and 15-48 percent of annualized capital costs for phosphorus to treat nutrients (Kang and Omstead, 2011).[2] Thus, we add the addition of the high estimate of labor costs (48 percent additional costs annually) as a percentage of capital costs were considered across each scenario.

The two businesses were assumed to be at WERF Level 1 which suggests little or no nutrient treatment. The businesses were calculated to have annual costs of $6,700 and $1,500 annually to meet WERF level 2. These figures were calculated from numbers found in the DRAFT Interim WERF study "*Striking the Balance Between Wastewater Treatment Nutrient Removal and Sustainability, Considering Capital and Operating Costs, Energy, Air and Water Quality and More*" (Falk, et al., 2011a). Because these businesses have such small outflows, these business might choose to either find an alternative discharge method (that is not into a state water), apply for an individual variance or perhaps change some operating procedures rather than investing in the capital expenditures to reach WERF level 2.

If the two businesses did invest in the equipment to get to WERF level 2, the $6,700 and $1,500 numbers for annual expenditure are probably significantly underestimated because the WERF numbers were for larger businesses with economies of scale. If we take into account economies of scale, it might be reasonable to multiply these numbers by a factor of 5. If we add 48 percent for labor costs on to this multiple of 5, then the numbers would be about $49,500 and $11,000 annually for these two businesses to reach WERF level 2. It is not clear how far beyond WERF level 2 these businesses would have to go as time goes on. These annual costs could effectively harm or shut down these two businesses. Thus, it is likely that both businesses would apply for an individual variance or find an alternative discharge method to avoid substantial impacts and these annual costs.

For the two businesses (one located in Madison County and the other in the northern Powder River Basin), non-discharge options include, a. land application, b. total/seasonal retention, c. piping water long distances away from state waters or to larger state waters with dilution, and d. trading. These non-discharge options, including land application, could be expensive or might not be feasible in certain areas (such as places far from open land or with few trade partners) or during the cold months.

[2] Based on information in: Introduction of Nutrient Removal technologies Manual, EPA, 2008 and WEF/WERF Cooperative Study of Nutrient Removal Plants: Achievable Technology Performance Statistics for Low Effluent Limits)

As noted above, these two affected businesses are located in or near small towns. Since most small towns do not have diverse economies, even a small decrease in business and in population can have a large effect on them. For example, some small Montana towns have less than 10 businesses total (e.g. Fromberg, MT).

There would be paperwork and other reporting tasks and sampling costs that the two small businesses directly affected would have to undertake in meeting the nutrient standards rule. It is difficult to say what these costs would be, but water sampling could be the largest cost along with extra time needed to comply. The other two or three business that already meet general variance standards would also have additional paperwork to comply with in their water discharge permit, although probably not significantly more than the current paperwork.

New small businesses with new wastewater discharges that want to located in Montana could also be affected greatly by the Nutrient Standards rule and may need to install water treatment technologies up to the Limits of Technology (stricter than the general variance) due to Non-Degradation rules. Some new businesses may choose to not locate in Montana if they were required to be in compliance immediately, while other states may not have this requirement. Eventually, all U.S. states will have to meet nutrient criteria, so this effect will probably decline over time.

The multiplier effects from the two small businesses directly affected would be minimal for the state as a whole and could be significant to the small towns where they are located.

Appendix A: Company Sizes

Matt Betcher of the Montana Department of Labor and Industry, Research and Analysis Bureau, entered the size classes of the businesses from the approximately 50 affected businesses (that have discharge permits) studied in the 'Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met in 2011/2012'. The obvious large businesses greater than 50 employees were not looked at, but businesses that might be under 50 employees were looked at by Mr. Betcher. He was able to find ranges of numbers of employees for these businesses using an alpha search of business names. The size classes are based on the average employment over last year of Quarterly Census of Employment and Wages (QCEW) data. Jeff Blend looked up the companies that were not in that database.

Quarterly Census of Employment and Wages is found at: http://www.bls.gov/cew/

Appendix B-Small Business Statute

http://openstates.org/mt/bills/2013/SB139/documents/MTD00006286/

2013 Montana Legislature, SENATE BILL NO. 139, INTRODUCED BY E. WALKER

NEW SECTION. **Section 1. Small business impact analysis -- assistance.** (1) Prior to the adoption of a proposed rule, the agency that has proposed the rule shall determine if the rule will adversely or positively impact small businesses. If the agency determines that the proposed rule will impact small businesses, the determination must be published in the register when the proposed rule is published. If the agency determines that the proposed rule may have an adverse or positive impact on small businesses, the agency shall prepare a small business impact analysis that, at a minimum, must:

(a) identify by class or group the small businesses probably affected by the proposed rule;

(b) include a statement of the probable adverse or positive effects of the proposed rule on the small businesses identified in subsection (1)(a); and

(c) include a description of any alternative methods that may be reasonably implemented to minimize or eliminate adverse effects described in subsection (1)(b), while still achieving the purpose of the proposed rule.

(2) The agency shall provide documentation for the estimates, statements, and descriptions required under subsection (1).

(3) The office of economic development, established in 2-15-218, shall advise and assist agencies in complying with this section.

Section 2. Section 2-4-102, MCA, is amended to read:

"**2-4-102. Definitions.** For purposes of this chapter, the following definitions apply:

(1) "Administrative rule review committee" or "committee" means the appropriate committee assigned subject matter jurisdiction in Title 5, chapter 5, part 2.

(2) (a) "Agency" means an agency, as defined in 2-3-102, of state government, except that the provisions of this chapter do not apply to the following:

(i) the state board of pardons and parole, except that the board is subject to the requirements of 2-4-103, 2-4-201, 2-4-202, and 2-4-306 and its rules must be published in the ARM and the register;

(ii) the supervision and administration of a penal institution with regard to the institutional supervision, custody, control, care, or treatment of youth or prisoners;

(iii) the board of regents and the Montana university system;

(iv) the financing, construction, and maintenance of public works;

(v) the public service commission when conducting arbitration proceedings pursuant to 47 U.S.C. 252 and 69-3-837.

(b) The term does not include a school district, a unit of local government, or any other political subdivision of the state.

(3) "ARM" means the Administrative Rules of Montana.

(4) "Contested case" means a proceeding before an agency in which a determination of legal rights, duties, or privileges of a party is required by law to be made after an opportunity for hearing. The term includes but is not restricted to ratemaking, price fixing, and licensing.

(5) (a) "Interested person" means a person who has expressed to the agency an interest concerning agency actions under this chapter and has requested to be placed on the agency's list of interested persons as to matters of which the person desires to be given notice.

(b) The term does not extend to contested cases.

(6) "License" includes the whole or part of an agency permit, certificate, approval, registration, charter, or other form of permission required by law but does not include a license required solely for revenue purposes.

(7) "Licensing" includes an agency process respecting the grant, denial, renewal, revocation, suspension, annulment, withdrawal, limitation, transfer, or amendment of a license.

(8) "Party" means a person named or admitted as a party or properly seeking and entitled as of right to be admitted as a party, but this chapter may not be construed to prevent an agency from admitting any person as a party for limited purposes.

(9) "Person" means an individual, partnership, corporation, association, governmental subdivision, agency, or public organization of any character.

(10) "Register" means the Montana Administrative Register.

(11) (a) "Rule" means each agency regulation, standard, or statement of general applicability that implements, interprets, or prescribes law or policy or describes the organization, procedures, or practice requirements of an agency. The term includes the amendment or repeal of a prior rule.

(b) The term does not include:

(i) statements concerning only the internal management of an agency or state government and not affecting private rights or procedures available to the public, including rules implementing the state personnel classification plan, the state wage and salary plan, or the statewide accounting, budgeting, and human resource system;

(ii) formal opinions of the attorney general and declaratory rulings issued pursuant to 2-4-501;

(iii) rules relating to the use of public works, facilities, streets, and highways when the substance of the rules is indicated to the public by means of signs or signals;

(iv) seasonal rules adopted annually or biennially relating to hunting, fishing, and trapping when there is a statutory requirement for the publication of the rules and rules adopted annually or biennially relating to the seasonal recreational use of lands and waters owned or controlled by the state when the substance of the rules is indicated to the public by means of signs or signals; or

(v) uniform rules adopted pursuant to interstate compact, except that the rules must be filed in accordance with 2-4-306 and must be published in the ARM.

(12) (a) "Significant interest to the public" means agency actions under this chapter regarding matters that the agency knows to be of widespread citizen interest. These matters include issues involving a substantial fiscal impact to or controversy involving a particular class or group of individuals.

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(13) "Small business" means a business entity, including its affiliates, that is independently owned and operated and that employs fewer than 50 full-time employees.

~~(13)~~(14) "Substantive rules" are either:

(a) legislative rules, which if adopted in accordance with this chapter and under expressly delegated authority to promulgate rules to implement a statute have the force of law and when not so adopted are invalid; or

(b) adjective or interpretive rules, which may be adopted in accordance with this chapter and under express or implied authority to codify an interpretation of a statute. The interpretation lacks the force of law."

From:	Blend, Jeff
To:	Greeley, Carrie
Subject:	Small business Impact Study for Nutrients
Date:	Wednesday, April 02, 2014 10:05:43 AM

Jeff Blend
(406) 444-0218
jblend@mt.gov

Economist and Energy Analyst
Energy and Pollution Prevention Bureau
Montana Dept. of Environmental Quality
1520 East Sixth Ave
P.O. Box 200901
Helena, MT 59620-0901

Montana Department of

Environmental Quality — Memo

TO: Board of Environmental Review

FROM: Jeff Blend, Economist

DATE: April 1, 2014

SUBJECT: Small Business Impact Analysis for MAR Notice No. 356

Attached is a Small Business Impact analysis for MAR Notice No. 356. This analysis fulfills the statutory obligations of Senate Bill 139 passed in the 2013 Legislature, which requires an analysis of the direct impacts to small businesses from a new rule.

Small Business Impact Analysis of the Nutrient Standards Rule

The Nutrient Standards Rule (MAR Notice No. 17-356, Circular DEQ 12A) and Variance Rule (MAR Notice No. 17-355, Circular 12B) will significantly and directly impact a couple of small businesses in the state of Montana. For the purposes of this study, small businesses are defined as less than 50 employees. An estimated two existing small businesses will see larger cost impacts from having to meet higher water quality standards. These two businesses are located in a small town and rural area in Montana. The two businesses that might have to upgrade would likely experience significant impacts from the rule.

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Appendix B-Small Business Statute

http://openstates.org/mt/bills/2013/SB139/documents/MTD00006286/

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From: Johnson, Elois
To: Orr, Katherine; Mathieus, George; Suplee, Mike; Greeley, Carrie
Subject: FW: Montana Mining Association Comments RE: Numeric Nutrient Standards Rule Package
Date: Tuesday, April 01, 2014 2:12:54 PM

Since these comments reference both MAR Notice No. 17-355 (Department) and 17-356 (Board), I included Carrie Greeley as a recipient of this e-mail. She was to receive comments pertaining to MAR Notice No. 17-355.

Elois

From: Tammy Johnson [mailto:tjohnson@montanamining.org]
Sent: Tuesday, April 01, 2014 1:04 PM
To: Johnson, Elois
Cc: 'Laura Feist'
Subject: Montana Mining Association Comments RE: Numeric Nutrient Standards Rule Package

Dear Elois,

Attached please find the comments of the Montana Mining Association with regard to the Numeric Nutrient Standards Rule Packages before the Montana Board of Environmental Review and the Montana Department of Environmental Quality. The hard copy has been sent to you via USPS Priority Mail.

Sincerely yours,

Tammy Johnson

Tammy Johnson, Executive Director
Montana Mining Association
P.O. Box 5567
Helena, MT 59604
(406) 495-1444 / Office
(406) 491-1714 / Cell
tjohnson@montanamining.org
www.montanamining.org

MONTANA MINING ASSOCIATION

Office Address: 2301 Colonial Drive, Suite 2A ~ Helena, MT 59601
Mailing Address: P.O. Box 5567 ~ Helena, MT 59604
Telephone: (406) 495-1444
Email: info@montanamining.org
Website: http://www.montanamining.org

March 31, 2014

Montana Department of Environmental Quality
1520 E. 6th Avenue
P.O. Box 200901
Helena, Montana 59620-0901
Attention: Elois Johnson
Submitted via Email: ejohnson@mt.gov and USPS

RE: Draft Department Circulars DEQ-12A and 12-B; Associated Rules (MAR Notice 17-356 and MAR Notice 17-355); Base Numeric Nutrient Standards Implementation Guidance

Dear Ms. Johnson:

Thank you for the opportunity to submit comments on the above referenced rule package. The Montana Mining Association is a trade association of mineral developers, producers, refiners and vendors in the State of Montana. The mining industry is a major employer and taxpayer in Montana and we believe the continued viability and growth of our members' operations are significant factors in the economic health of our state and its citizens.

The Montana Mining Association (MMA), as a member of the Nutrient Work Group, has been engaged in the multi-year process which has led to this public comment period on the rule package which adopts numeric standards for nutrients and establishes the legislatively required variance package and DEQ guidance for implementation.

The MMA would like to express our appreciation to the Montana Department of Environmental Quality (DEQ) for the hard work and dedication of its staff in developing a five-piece package, that when taken as a whole, should provide the means to improve Montana's water quality by preventing detrimental levels of nutrients being discharged to waters in the state. The five documents are interrelated and though the Montana Board of Environmental Review and the Montana Department of Environmental Quality each has authority for adoption of separate pieces of the package, it is important to understand not only the correlation but to understand the importance of all five pieces moving forward together.

The MMA supported SB 367, the legislation authorizing this rulemaking effort. The following fundamental components are at the heart of our support for the legislation, our comments in this letter, and our unresolved concerns.

- Proposed nutrient standards are below the limits of current wastewater treatment technology.
- Compliance with proposed numeric nutrient concentrations, without the availability of variances will result in substantial and widespread economic impacts.
- Legislative intent was to make variances available to current dischargers and to new dischargers.
- Legislative intent was not to make the variances available to all dischargers only to have it nullified by the nondegradation review process.
- Unambiguous non-severability language must be included throughout the package to ensure, with respect to all dischargers, that the rules do not continue to bind permittees should permits with a general variance be disallowed or are not allowed to take effect.

MMA's comments include both general comments on the entire rule package and approach to implementing SB 367 and some specific comments on the proposed rules and DEQ Circulars 12A and 12B.

GENERAL COMMENTS

It is MMA's understanding that the legislative intent was that nutrient standards variances were to be made available to all dischargers. We wish to acknowledge the following found in the Guidance Document in Section 1.1 Scope – The provisions for general, individual, and alternative variances in Section 75-3-313, MCA, are available to all discharge permit holders and not limited to dischargers under permit on the effective dates of DEQ Circular DEQ 12-A or DEQ Circular 12-B. We sincerely appreciate the DEQ's acknowledgement of what we believe to be the legislative intent of SB 367. However, the fact that the Guidance document can be modified without any notification or process and that the Rule and DEQ Circular 12B are silent on the availability of the General Variance to private entities is a concern.

- Please state for the record in reply to this comment that the Department's position on issuance of the General Variance will be the same for private and public entities and that the General Variance will be available to new and increased dischargers on the same basis as for existing permit holders.

The Base Numeric Nutrient Standards Implementation Guidance document is not referenced by Rule or in DEQ-12A or DEQ-12B. This document is important in understanding how these standards will be interpreted and implemented and the lack of this reference limits the publics' understanding of the rule package.

Additionally, there are several problems and inconsistencies with the Guidance that need to be corrected in order for there to be effective and transparent implementation by the Department.

- Please respond to this comment by indicating that DEQ intends to utilize the Guidance document to implement the Rule and to issue permits.

The economic test for applicability of the Individual Variance for private entities remains based on and referenced to the 1995 EPA draft guidance. The legislature established, and 75-5-313(5), MCA codifies, the finding that "treatment of wastewater to base numeric nutrient standards would result in substantial and widespread economic impacts". This finding should preclude the need for an individual finding for either the Individual or Alternative Variances (as is the case for the General Variance).

- A modification of the procedure outlined in DEQ Circular 12B 3.2 could potentially be used to determine the level of capital and O&M expenditures that a discharger could afford, but should not be used to determine the availability of a variance.

Neither the Rule changes nor DEQ Circular 12B appropriately address what MMA sees as the legislative intent with regards to the application of the nondegradation requirements. MMA is of the opinion that it was not the intent of the Montana Legislature in SB 367 to establish a variance system only to have it nullified by the nondegradation review process.

- Clarification of the implementation of nondegradation for existing and future permits should be included in this rule package.

17.30.715(1) ARM retains nondegredation review for NO2 and NO3 and adds nondeg review for TN and TP. With the adoption of numeric standards for TN and TP at sub ppm levels, the significance level for these parameters at 10% of the standard outside of a mixing zone (17.30.637(1)(f), ARM) are very low (e.g. 27.5 ppb and 2.5 ppb respectively in the northern Rockies ecozone). The existing standard nondegradation evaluation process should recognize the difference between nutrients and other parameters for which water quality standards have been adopted.

- The nondegradation review needs to recognize the seasonal nature of the nutrient standards as well as recognize that the traditional concepts of chronic and acute effects are not appropriate.

MMA is concerned about how numeric standards would be applied in permits. One issue is the application of seasonal standards. Although DEQ has begun using the proposed numeric nutrient standards as *de facto* standards in TMDLs and permits, it does not appear that the seasonal nature of the proposed standards are being recognized in permits (for example the Butte Highlands permit).

- The seasonal nature of potential impacts from nutrients is clearly documented in DEQ's standards development process and needs to be implemented in permits and TMDLs.

PROCESS

MMA feels as a member of the Nutrient Work Group that the instruction to "advise the department on the base numeric nutrient standards, the development of temporary nutrient criteria nutrient standards variances" was not adequately fulfilled at the end of the process when the Nutrient Work Group was not allowed to review or comment on the final rule package.

MMA requests as a member of the Nutrient Work Group and as indicated by 75-5-313(2)(b), ARM that we be informed of all draft permits that implement nutrient effluent limits or nutrient standards variances and any policy decisions the Department reaches in implementing the numeric nutrient standards or variances.

NON-SEVERABILITY CLAUSE

MMA supports the modification of the proposed non-severability clause. In the event of an EPA denial of an individual permit based on the inclusion of the general variance for a new or increased discharge should be added to the list of causes that would void the proposed rule package.

Specifically, The MMA asks the Board and the DEQ to amend the language employed by DEQ in the rule as noted in the italicized language as follows:

> If (1) a court of competent jurisdiction declares 75-5-313, MCA, or any portion of that statute invalid, (2) the United States Environmental Protection Agency disapproves 75-5-313, MCA, or any portion of that statute, under 30 CFR 131.21, or if rules adopted pursuant to 75-5-313(6) or (7), MCA, expire and general variances are not available, *or (3) after the date of the promulgation of this rule, the United States environmental protection agency nullifies or otherwise disallows a permit with a general variance issued by the Department based upon the Department's inclusion of a general variance in the permit*, then (1)(e) and all references to DEQ-12A, base numeric nutrient standards and nutrient standards variances in ARM 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.622 through 17.30.629, 17.30.635, 17.30.702, and 17.30.715 are void, and the narrative water quality standards contained in ARM 17.30.637 are the standards for total nitrogen and total phosphorus in surface water, except for the Clark Fork River, for which the standards are the numeric standards in ARM 17.30.631.

Without the addition of this language to the rule, the rule will remain in force if EPA rejects a permit with a general variance for the permittee because EPA does not believe the permittee is entitled to a general variance.

MONTANA SHOULD NOT BE ECONOMICALLY DISADVANTAGED

MMA continues to express our concern with regard to the adoption and implementation of numeric standards at levels below the level of viable treatment technology and in advance of numeric nutrient standards adoption by most other states. We wish to remain positive about this process and of this rule package, but admittedly have concerns when we see that our immediate neighbors in Idaho, the Dakotas and Wyoming have not ventured down this path.

One thing is certain for now. Montana will have numeric nutrient standards in place when our neighbors do not. We do not want to see companies making a decision to not locate in Montana or to leave our state because it is technologically and economically infeasible to acquire the necessary permits to operate in this state. This overall package cannot result in a regulatory moratorium on new business in Montana.

SPECIFIC COMMENTS – DEQ CIRCULAR 12-A

2.0 – Applying criteria "as an annual average, not to be exceeded more than once in any three year period, <u>on average.</u>" is unclear. Please clarify what "<u>one in any three year period, on average</u>" means?

SPECIFIC COMMENTS – DEQ CIRCULAR 12-B

1.1 – The definition of "Monthly Average" is confusing. The period in which the base numeric nutrient standards apply is generally July 1 to September 30. If this definition is to be applied to permit compliance then it seems that it should reference the sum of the measurements for a parameter divided by the number of samples during the reporting period.

2.0 2nd paragraph last sentence on page 1 – *…(or an individual variance)…* Alternative Variance should be included here.

3.1 3rd paragraph – S*ince the basis of this type of individual variance is related to the economic status of a community or permittee* … The basis for an individual variance can also be limits of technology, which should be noted here.

SPECIFIC COMMENTS – BASE NUMERIC NUTRIENT STANDARDS IMPLEMENTATION GUIDANCE

1.1 Scope – MMA appreciates the inclusion of this statement, but continues to feel that this should be included in Rule and Circular DEQ-12B.

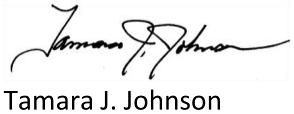
2.0 Reduction Steps – While it is useful for dischargers to have an idea of potential future changes in treatment standards associated with the General Variance, it would be appropriate in this section to indicate that these projected future treatment levels are targets and would be subject to rulemaking to implement. These target levels were not required by or specified by SB 367.

3.2 Private Sector Permittees – The finding by the legislature in SB 367 of significant and widespread impacts should apply to Individual Variances as well as the General Variance.

3.2.2 Completing the Substantial and Widespread Assessment Spreadsheet – The inclusion of a nondegradation waiver procedure based on significant and widespread economic impacts and generally following the EPA 1995 draft guidance was not discussed with the Nutrient Work Group and should not be included in this guidance package or the Work Sheet without input or consideration by the public. Further, it is not clear if the "cost model" work sheet available on DEQ's web site and the reference cited here are the same. DEQ should clarify which version of the worksheet it intends to use and allow for public comment on the details in the worksheet.

In conclusion, the MMA is keenly aware of the amount of time, effort and talent that has been devoted to this overall process and wishes to say thank you and to express our appreciation to everyone involved. It has been a very large task which corresponds to the very large step Montana is taking as one of the early pioneers or adopters of the numeric nutrient standards process. The MMA will continue to stay engaged and trust that the DEQ will continue to work with our membership as we move forward. Again, thank you for the opportunity to submit our comments.

Sincerely yours,



Tamara J. Johnson
Executive Director
Email: tjohnson@montanamining.org

Cc: Water Policy Interim Committee
Environmental Quality Council

Montana Department of
ENVIRONMENTAL QUALITY MEMO

TO: Board of Environmental Review
Tracy Stone-Manning, Director

FROM: John F. North, Chief Legal Counsel

DATE: March 24, 2014

SUBJECT: HB 521 Stringency and SB 311Takings Analyses for MAR Notices No. 17-355 and 356

HB 521, which is codified at 75-5-203, MCA, requires that the Department make certain findings before it may adopt water quality rules that are more stringent than comparable federal regulations that address the same circumstances. Section 75-5-309, MCA, contains a similar requirement.

In MAR Notice No. 17-355, the Board is proposing to adopt numeric nutrient standards. In MAR Notice No. 17-356, the Department is proposing to adopt procedures for variances from the numberic nutrient standards and to place in rule numeric nutrient general variance effluent limits that, purusnat to 75-5-313, MCA, will go into effect upon adoption of the numeric nutreint standards. Because the general variances and numeric nutrient standards will substantively take effect at the same time, I am analyzing them together in this memorandum.

The EPA has not adopted regulations imposing numeric nutrient standards. However, it has adopted guidance for states to use to set numeric nutrient standards. Using the formulas in the guidance, the Department has calculated numeric nutrient standards for Montana. Those calculations result in standards that are more stringent than standards proposed in MAR Notice 17-356, except for the proposed standards for Flathead Lake, which are more stringent than EPA guidance for phosphorus, chlorophyl A, and secchi depth. Findings pursuant to 75-5-203 and 308 would be necessary for the Flathead Lake. EPA has no regulations setting criteria for variances. The varienacc creiteria and procedures proposed in MAR Notice No. 17-356 are no more stringent than EPA.

SB 311 is codified as Title 2, Chapter 10, MCA. That chapter requires an agency to conduct a takings impact assessment for actions, including adoption of rules, with taking or damaging implications. It directs that the Attorney General provide a checklist for agencies to use in determining whether actions have taking or damaging implications. Enclosed is a checklist for these rule amendments. It indicates that adoption of these rule amendments does not have taking or damaging implications.

Enclosure

Attachment

PRIVATE PROPERTY ASSESSMENT ACT CHECKLIST: MAR Notices No. 17-355 and 17-356

DOES THE PROPOSED AGENCY ACTION HAVE TAKINGS IMPLICATIONS UNDER THE PRIVATE PROPERTY ASSESSMENT ACT?

Yes	No	
X		1. Does the action pertain to land or water management or environmental regulation affecting private real property or water rights?
	X	2. Does the action result in either a permanent or indefinite physical occupation of private property?
	X	3. Does the action deprive the owner of all economically viable uses of the property?
	X	4. Does the action deny a fundamental attribute of ownership?
	X	5. Does the action require a property owner to dedicate a portion of property or to grant an easement? [If the answer is NO, skip questions 5a and 5b and continue with question 6.]
		5a. Is there a reasonable, specific connection between the government requirement and legitimate state interests?
		5b. Is the government requirement roughly proportional to the impact of the proposed use of the property?
	X	6. Does the action have a severe impact on the value of the property?
	X	7. Does the action damage the property by causing some physical disturbance with respect to the property in excess of that sustained by the public generally? [If the

answer is NO, do not answer questions 7a through 7c.]

______ ______ 7a. Is the impact of government action direct, peculiar, and significant?

______ ______ 7b. Has government action resulted in the property becoming practically inaccessible, waterlogged, or flooded?

______ ______ 7c. Has government action diminished property values by more than 30% and necessitated the physical taking of adjacent property or property across a public way from the property in question?

Taking or damaging implication exist if YES is checked in response to question 1 and also to any one or more of the following questions: 2, 3, 4, 6, 7a, 7b, 7c; or if NO is checked in response to questions 5a or 5b.

________________________ March 24, 2014

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.619, 17.30.622, 17.30.623, 17.30.624, 17.30.625, 17.30.626, 17.30.627, 17.30.628, 17.30.629, 17.30.635, 17.30.702, and 17.30.715 pertaining to permit application, degradation authorization, and annual permit fees, specific restrictions for surface water mixing zones, standard mixing zones for surface water, definitions, incorporations by reference, A-1 classification standards, B-1 classification standards, B-2 classification standards, B-3 classification standards, C-1 classification standards, C-2 classification standards, I classification standards, C-3 classification standards, general treatment standards, definitions, and criteria for determining nonsignificant changes in water quality)	NOTICE OF AMENDMENT (WATER QUALITY)

TO: All Concerned Persons

1. On February 13, 2014, the Board of Environmental Review published MAR Notice No. 17-356 regarding a notice of proposed amendment of the above-stated rules at page 280, 2014 Montana Administrative Register, Issue Number 3.

2. The board has amended 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.619, 17.30.622, 17.30.623, 17.30.624, 17.30.625, 17.30.626, 17.30.627, 17.30.628, 17.30.629, 17.30.635, 17.30.702 exactly as proposed. It has amended ARM 17.30.619 as proposed, except that the reference to "Circular DEQ-12A, entitled 'Montana Base Numeric Nutrient Standards' (December 2013 Edition)" has been changed to "Circular DEQ-12A, entitled 'Montana Base Numeric Nutrient Standards' (July 2014 Edition)" to reflect the date of adoption of the circular and has amended ARM 17.30.715 as proposed, but with the following changes, stricken matter interlined, new matter underlined:

<u>17.30.619 INCORPORATIONS BY REFERENCE</u> (1) The board adopts and incorporates by reference the following state and federal requirements and procedures as part of Montana's surface water quality standards:

(a) through (d) remain as proposed.

(e) Department Circular DEQ-12A, entitled "Montana Base Numeric Nutrient Standards" (~~December 2013~~ July 2014 edition), which establishes numeric water quality standards for total nitrogen and total phosphorus in surface waters.

(2) and (3) remain as proposed.

17.30.715 CRITERIA FOR DETERMINING NONSIGNIFICANT CHANGES IN WATER QUALITY (1) The following criteria will be used to determine whether certain activities or classes of activities will result in nonsignificant changes in existing water quality due to their low potential to affect human health or the environment. These criteria consider the quantity and strength of the pollutant, the length of time the changes will occur, and the character of the pollutant. Except as provided in (2), changes in existing surface or ground water quality resulting from the activities that meet all the criteria listed below are nonsignificant, and are not required to undergo review under 75-5-303, MCA:

(a) and (b) remain as proposed.

(c) discharges containing toxic parameters, ~~inorganic nitrogen, or inorganic phosphorus, except as specified in (1)(d) and (e),~~ which will not cause changes that equal or exceed the trigger values in Department Circular DEQ-7. Whenever the change exceeds the trigger value, the change is not significant if the resulting concentration outside of a mixing zone designated by the department does not exceed 15% of the lowest applicable standard;

(d) through (e) remain as proposed.

(f) changes in the quality of water for any harmful parameter, ~~including~~ and parameters listed in Department Circular DEQ-12A, except as specified in (1)(g), for which water quality standards have been adopted other than carcinogenic, bioconcentrating, or toxic parameters, in either surface or ground water, if the changes outside of a mixing zone designated by the department are less than 10% of the applicable standard and the existing water quality level is less than 40% of the standard;

(g) for nutrients in domestic sewage effluent discharged from a septic system that does not require an MPDES or MGWPCS permit, except as specified in (1)(d) and (e), which will not cause changes that equal or exceed the trigger values in Department Circular DEQ-7. Whenever the change exceeds the trigger value, the change is not significant if the changes outside of a mixing zone designated by the department are less than 10% of the applicable standard and the existing water quality level is less than 40% of the standard;

(g) remain as proposed, but is renumbered (h).

(2) and (3) remains as proposed.

(4) If a court of competent jurisdiction declares 75-5-313, MCA, or any portion of that statute invalid, ~~or~~ if the United States Environmental Protection Agency disapproves 75-5-313, MCA, or any portion of that statute under 30 CFR 131.21, or if rules adopted pursuant to 75-5-313(6) or (7), MCA, expire and general variances are not available, then the significance criteria contained in (1)(g) are the significance criteria for total nitrogen and total phosphorus in surface water.

3. The following comments were received and appear with the board's responses:

COMMENT NO 1: The rule proposes uniform, relaxed mixing zone standards. In contrast, EPA policy recommends that mixing zone characteristics be defined on a case-by-case basis after it has been determined that the assimilative capacity of the receiving system can safely accommodate the discharge. EPA also states that the assessment should take into consideration the physical, chemical, and biological characteristics of the discharge and the receiving system; the life history and behavior of organisms in the receiving system; and the desired uses of the waters.

RESPONSE: The proposed rules provide for the use of the entire 14Q5 flow in dilution calculations involving a standard mixing zone. The issue of appropriate mixing zones for base numeric nutrient standards was discussed during Nutrient Work Group meetings and EPA was involved in the discussions. Although EPA's national policy is for case-by-case analysis, as noted by the commenter, much of the concern regarding such case-by-case analysis revolves around toxic compounds, which have both chronic and acute impact levels. Mixing zones are designed to make sure acute levels are not exceeded in the mixing zone because this would harm any aquatic life present. In contrast, nutrients at the concentrations of the base numeric nutrient standards behave like chronic criteria, and the changes to the mixing zone rules reflect this reduced potential for direct aquatic life impact.

COMMENT NO. 2: Endnote 4 in Circular DEQ-12A should be rephrased to clarify whether it applies to the development of permit limits or stream assessment for listing/delisting on Montana's 303(d) list. Further, it should be clarified that it is a monthly average, not a 30-day rolling average.

RESPONSE: The board agrees that this should be clarified. Section 2.2 of the circular states that permit limits for nutrient discharges are to be developed using the Average Monthly Limit (AML). Thus, the circular already provides-for permitting purposes-that the averaging timeframe (duration) for permitting nutrient discharges is 30 days. The board also agrees that the 30-day period should not be a rolling average. Section 2.2 has also been modified to provide that this is a calendar month.

In contrast to setting permit limits, when assessing a stream's ambient condition for 303(d) listing purposes, the department's monitoring and assessment unit collects nutrient samples throughout the growing season (a three-month period each year) and evaluates all data using statistical testing procedures. It does not restrict the evaluation to a calendar month. Footnote 4 has been updated to better reflect the monitoring and assessment process. It now reads: "The average concentration during a period when the standards apply may not exceed the standards more than once in any five-year period, on average." In relation to the duration and frequency requirements of the standards, it should be noted that, because permits are written to a shorter time frame (a calendar month), they are fully protective of the standard. In addition, the monitoring and assessment unit of the department evaluates biological data in concert with the nutrient concentrations to make a final assessment.

COMMENT NO. 3: Some commenters supported adoption of Flathead Lake TMDL Phase I targets as Flathead Lake water-quality standards, but stated that a document describing the technical and scientific support for the standards is needed first. Other commenters asked for postponement of the standards adoption for Flathead Lake pending a more thorough technical review and local stakeholder involvement.

RESPONSE: The board agrees that the standards for Flathead Lake should be postponed and that more details on the scientific and technical basis of the standards should be prepared. The standards for Flathead Lake have been removed from Department Circular DEQ-12A.

COMMENT NO. 4: Nutrient standards should be adopted for the Flathead River.

RESPONSE: Numeric standards for the Flathead River were not proposed for the Flathead River in the notice of public hearing, and therefore adoption of standards for that stream is not within the scope of this rulemaking. It would require commencement of a new rulemaking proceeding.

COMMENT NO. 5: Required reporting limits in Table 12A-3 of Department Circular DEQ-12A for total kjeldahl nitrogen (TKN) are not obtainable and should not be adopted in rule.

RESPONSE: The board agrees and has increased the required reporting value (RRV) for TKN from 150 µg/L to 225 µg/L. The board is aware that the typical practical quantitation limit for total kjeldahl nitrogen (TKN) is 0.5 mg/L, or 500 µg/L. However this limit is not low enough to meet some nitrogen standards proposed in Circular DEQ-12A. If a reporting limit is greater than or equal to an applicable standard, state waters that have non-detectable levels of TKN may be unnecessarily listed as impaired and/or may have requirements for extensive sampling. This is why the board calculates an RRV for parameters which have numeric standards. The board has worked with the department and statewide analytical laboratories to derive the updated RRV. The board understands that this number is still very low and that conditions must be optimal in order to achieve this number. Reporting value concentrations higher than 225 µg/L may also be acceptable on a case-by-case basis, as indicated in Table 12A-3 of Department Circular DEQ-12A. Additionally, if a sample must be diluted, it is understood that the reporting limit will be raised.

COMMENT NO. 6: The board should consider expanding the period of application of the standards, where needed, because nutrient problems can manifest at times other than in summer/early fall due to low snowpack, early irrigation withdrawal, etc.

RESPONSE: The board is aware that eutrophied rivers and streams can manifest a period of excess algal growth in early spring prior to runoff. This has been observed in the Clark Fork River, for example. In western Montana, this algal growth is typically scoured away by the spring runoff and algal growth recommences in late June to early July. Application of nutrient standards is not necessary during the spring because spring is relatively short (typically a few weeks), generally before the main recreation season, and followed by a scouring period, flow volume is not an appropriate factor for determining the end date for application of nutrient criteria.

After runoff ends, base flow begins and can be fairly uniform into November and December. However, regional climatic influences, such as lowered temperatures and light intensity, typically cause, by early October, major reductions in aquatic plant life growth, reductions in aquatic macroinvertebrate productivity, and higher dissolved oxygen concentrations. Essentially, the productive period for the year ends in the fall and the importance of nutrient concentrations to this productivity greatly declines. The proposed end dates for the period of application of the nutrient criteria reflect this.

<u>COMMENT NO. 7:</u> Site-specific criteria or modified periods of application should be used to tighten nutrient standards where it is apparent that nuisance algae are becoming worse or not improving.

<u>RESPONSE:</u> The board agrees. These changes would be made in another rulemaking proceeding that would be initiated once it is determined that the existing standards do not protect the use.

<u>COMMENT NO. 8:</u> In ARM 17.30.715(1)(c), the words "inorganic nitrogen, or inorganic phosphorus" should be deleted. If they are not, discharges of nutrients will be subject to the non-significance of both (1)(c) and (1)(f). Based on numerous meetings of the nutrient work group and early drafts, non-significance should be determined under (1)(f) using the base numeric nutrients standards rather than under the existing narrative standard.

<u>RESPONSE:</u> The board agrees. ARM 17.30.715(1)(c) has been modified so that non-significance will be determined under ARM 17.30.715(1)(f) for all discharges that require a surface or groundwater permit. Additionally, for discharges that do not require one of those permits, but which undergo non-degradation review nonetheless, a new subsection, ARM 17.30.715(1)(g), has been added. This subsection retains the trigger value requirement as the initial criterion for significance. Retention of the trigger value will allow the department's subdivisions program, which has stringent deadlines for reviewing subdivisions, to continue to use an expeditious means of determining significance for small subdivisions. Failing this first test, the next test for non-significance will be the same as is found in ARM 17.30.715(1)(f); that is, a change is non-significant if the change is < 10% of the numeric nutrient standards and existing water quality is currently < 40% of those standards.

<u>COMMENT NO. 9:</u> Movement from the 7Q10 to the seasonal 14Q5 for nutrients is a poor policy choice. The board should stick with the 7Q10 for permitting base numeric nutrient standards.

<u>RESPONSE:</u> The low-flow design flow is chosen to ensure compliance with the water quality criteria. Given that the proposed criteria are to be permitted based on a 30-day average concentration, and have an allowable excursion frequency of once in five years, it is overly restrictive to consider the 7Q10. When establishing permit limits, the 14Q5 ensures protection of the proposed criteria at a level that corresponds to the averaging period and allowable excursion frequency of the underlying criteria, while simultaneously providing a margin of safety because the low-flow averaging period is somewhat shorter (14 days instead of 30).

COMMENT NO. 10: A commenter pointed out that the nonseverability provision in 17.30.619 is triggered by the expiration of the general variance rules, while the nonseverability provision in 17.30.715 is not. The commenter stated that this should be a trigger in both rules. Several other commenters stated that the board should modify the non-severability clauses to be triggered if EPA objects to or vetoes a permit based on the inclusion of a variance.

RESPONSE: The board agrees that the triggers for both provisions should be the same and has added the expiration of the variance rules to the nonseverability provision in ARM 17.30.715.

The board has not included the permit trigger. If EPA disapproves the general variance, the nonseverability provisions would apply. Once EPA approves the general variance, EPA would not have authority to object to or veto a permit for an existing discharger based on inclusion of general variance limits in the permit. In written comments on this rulemaking, EPA has indicated that variances may be available to new dischargers as long as existing uses are protected. EPA personnel have indicated to department personnel that inclusion of the permit trigger for nonseverability would likely result in EPA rule disapproval. Should this occur, the narrative standards would be reinstated. However, narrative standards are implemented in a permit by imposition of numeric limits, and it is possible that a court would hold that the proper application of the narrative standards results in the same numeric permit limits as the numeric standards would require. However, because the variance provisions of 75-5-313, MCA, take effect only upon adoption of numeric standards, that statute would not be available to new or existing dischargers in those instances.

COMMENT NO. 11: The rules should clarify implementation of nondegradation for existing and future permits. The board should recognize the seasonal nature of the nutrient standards when implemented in permits and nondegradation provisions.

RESPONSE: Nondegradation requirements do not apply to existing permittees unless they become increased sources, as defined in ARM 17.30.702(18). For new or increased sources, as defined in ARM 17.30.702(18), nondegradation for base numeric nutrient standards will be applied following ARM 17.30.715.(1)(f), (1)(g), (2), and (3). If this process results in a finding that degradation will occur, the applicant can apply for an authorization to degrade.

Circular 12A clearly provides that the standards are seasonal in nature. The department would therefore be legally bound to recognize this seasonal nature in permitting, including application of nondegradation.

COMMENT NO. 12: Applying criteria "as an average, not to be exceeded more than once in any three year period, on average" needs clarification.

RESPONSE: The allowable excursion frequency (once in five years in this case) is referred to "on average" in order to accommodate datasets longer or shorter than the specified five-year period. For example, if the dataset were ten years long, standards exceedences would be allowed twice in that period (2/10 years, equal to 1/5), but not three times.

<u>COMMENT NO. 13:</u> The rules should be clarified to show that these seasonal nutrient standards will not be applied to storm water discharges.

<u>RESPONSE:</u> All discharges, including storm water discharges, are subject to water quality standards, whether those standards are narrative, as the nutrient standards are currently structured, or numeric, as is proposed for nutrients.

<u>COMMENT NO. 14:</u> The overall nutrient standards package (including variances) cannot result in a regulatory moratorium on new business in Montana.

<u>RESPONSE:</u> The purpose of the variance processes is to assure that the economic effects of nutrient standards will not cause a regulatory moratorium on new business in Montana. In turn, the rules that have been developed to implement the statute reflect this intent. Variances can be granted to new businesses so long as the new dischargers show that the variance protects the existing use.

<u>COMMENT NO. 15:</u> Is a nuisance threshold for algae, as determined by public opinion polling in Montana, an appropriate standard to determine impact to beneficial use?

<u>RESPONSE:</u> A scientific poll of Montanans' opinions regarding what constitutes a nuisance algae level is appropriate for establishing a water-quality standard. All Montana surface waters have bathing, swimming, and recreation as part of their legally-defined beneficial uses. To determine when this interrelated set of uses was harmed, it was necessary to identify at what point the Montana public felt that their recreation was impaired by excess attached algae. Increased growth of attached algae is one of the most common manifestations of excess nutrients in regional streams, and was therefore appropriate to consider. Attached algae is very commonly measured via its chlorophyll a content and the department has standard operating procedures to do so. The public-opinion survey showed that there was a clear threshold at 150 mg chlorophyll a per square meter of stream bottom; algae levels above this impacted peoples' desire to fish, wade, swim, and boat (page 135, Suplee et al., 2009) which are all common recreational activities in Montana.

Montana is not alone in using this approach. In 2010/2011, the state of Utah carried out its own recreational survey to determine the opinion of the Utah public regarding algae levels in streams. They arrived at identical conclusions and thresholds as were found in Montana. The state of Vermont is planning to carry out its own algae recreation-impact public survey starting this summer; the focus will be the recreational use of Vermont streams. Similar approaches have also been used to establish phosphorus standards to protect water clarity and recreational use in lakes (Lake Champlain, for example).

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collection of nutrient samples along with algae samples and other biological measurements. All the data are considered together, and a few high nutrient samples do not necessarily mean the stream will be found to be impaired by nutrients; it depends on the degree to which the biological measurements show impairment as well. The standard operating procedure (Nutrient Assessment Method) for this process may be found on the department's website at: http://deq.mt.gov/wqinfo/qaprogram/sops.mcpx.

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RESPONSE: Under both state and federal law, water quality standards must protect the uses of the water. The Legislature anticipated that nutrient standards that protect the use of the waters would not be immediately achievable. By providing for nutrient standards variances, the Legislature provided a process that allows adoption of standards that meet legal requirements and a process that alleviates negative impacts on dischargers by providing variances for up to 20 years to achieve compliance with those standards.

4. No other comments or testimony were received.

Reviewed by: BOARD OF ENVIRONMENTAL REVIEW

________________________ By: ________________________

JOHN F. NORTH
Rule Reviewer

ROBIN SHROPSHIRE
Chairman

Certified to the Secretary of State, ____________, 2014.

Wittenberg, Joyce

From:	Anne Kania <floatingisland@me.com>
Sent:	Tuesday, April 01, 2014 12:47 PM
To:	DEQ WQP Admin
Subject:	Comments on Circular DEQ-12B
Attachments:	DEQ Response-BioHaven.docx; Dodkins report swansea.pdf; Summaries sorted by type and date 4-01-14.pdf

Dear Carrie,

Please find attached a word document containing comments on the Variances Circular, and two supporting pdf documents.

Thank you,
Anne Kania

Anne Kania

BioHaven, Inc.
10052 Floating Island Way,
Shepherd, MT 59079

1-800 450 1088
PH: +1 406 373 5200
Cell: +1 406 208 5520
Skype name: LAMLOW

Montana Department of
ENVIRONMENTAL QUALITY MEMO

TO: Board of Environmental Review
Tracy Stone-Manning, Director

FROM: John F. North, Chief Legal Counsel

DATE: March 24, 2014

SUBJECT: HB 521 Stringency and SB 311Takings Analyses for MAR Notices No. 17-355 and 356

HB 521, which is codified at 75-5-203, MCA, requires that the Department make certain findings before it may adopt water quality rules that are more stringent than comparable federal regulations that address the same circumstances. Section 75-5-309, MCA, contains a similar requirement.

In MAR Notice No. 17-355, the Board is proposing to adopt numeric nutrient standards. In MAR Notice No. 17-356, the Department is proposing to adopt procedures for variances from the numberic nutrient standards and to place in rule numeric nutrient general variance effluent limits that, purusnat to 75-5-313, MCA, will go into effect upon adoption of the numeric nutreint standards. Because the general variances and numeric nutrient standards will substantively take effect at the same time, I am analyzing them together in this memorandum.

The EPA has not adopted regulations imposing numeric nutrient standards. However, it has adopted guidance for states to use to set numeric nutrient standards. Using the formulas in the guidance, the Department has calculated numeric nutrient standards for Montana. Those calculations result in standards that are more stringent than standards proposed in MAR Notice 17-356, except for the proposed standards for Flathead Lake, which are more stringent than EPA guidance for phosphorus, chlorophyl A, and secchi depth. Findings pursuant to 75-5-203 and 308 would be necessary for the Flathead Lake. EPA has no regulations setting criteria for variances. The varienacc creiteria and procedures proposed in MAR Notice No. 17-356 are no more stringent than EPA.

SB 311 is codified as Title 2, Chapter 10, MCA. That chapter requires an agency to conduct a takings impact assessment for actions, including adoption of rules, with taking or damaging implications. It directs that the Attorney General provide a checklist for agencies to use in determining whether actions have taking or damaging implications. Enclosed is a checklist for these rule amendments. It indicates that adoption of these rule amendments does not have taking or damaging implications.

Enclosure

Attachment

PRIVATE PROPERTY ASSESSMENT ACT CHECKLIST: MAR Notices No. 17-355 and 17-356

DOES THE PROPOSED AGENCY ACTION HAVE TAKINGS IMPLICATIONS UNDER THE PRIVATE PROPERTY ASSESSMENT ACT?

Yes	No	
X		1. Does the action pertain to land or water management or environmental regulation affecting private real property or water rights?
	X	2. Does the action result in either a permanent or indefinite physical occupation of private property?
	X	3. Does the action deprive the owner of all economically viable uses of the property?
	X	4. Does the action deny a fundamental attribute of ownership?
	X	5. Does the action require a property owner to dedicate a portion of property or to grant an easement? [If the answer is NO, skip questions 5a and 5b and continue with question 6.]
		5a. Is there a reasonable, specific connection between the government requirement and legitimate state interests?
		5b. Is the government requirement roughly proportional to the impact of the proposed use of the property?
	X	6. Does the action have a severe impact on the value of the property?
	X	7. Does the action damage the property by causing some physical disturbance with respect to the property in excess of that sustained by the public generally? [If the

answer is NO, do not answer questions 7a through 7c.]

_______ _______ 7a. Is the impact of government action direct, peculiar, and significant?

_______ _______ 7b. Has government action resulted in the property becoming practically inaccessible, waterlogged, or flooded?

_______ _______ 7c. Has government action diminished property values by more than 30% and necessitated the physical taking of adjacent property or property across a public way from the property in question?

Taking or damaging implication exist if YES is checked in response to question 1 and also to any one or more of the following questions: 2, 3, 4, 6, 7a, 7b, 7c; or if NO is checked in response to questions 5a or 5b.

[signature] March 24, 2014

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
OF THE STATE OF MONTANA

In the matter of the amendment of ARM 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.619, 17.30.622, 17.30.623, 17.30.624, 17.30.625, 17.30.626, 17.30.627, 17.30.628, 17.30.629, 17.30.635, 17.30.702, and 17.30.715 pertaining to permit application, degradation authorization, and annual permit fees, specific restrictions for surface water mixing zones, standard mixing zones for surface water, definitions, incorporations by reference, A-1 classification standards, B-1 classification standards, B-2 classification standards, B-3 classification standards, C-1 classification standards, C-2 classification standards, I classification standards, C-3 classification standards, general treatment standards, definitions, and criteria for determining nonsignificant changes in water quality	NOTICE OF AMENDMENT (WATER QUALITY)

TO: All Concerned Persons

1. On February 13, 2014, the Board of Environmental Review published MAR Notice No. 17-356 regarding a notice of proposed amendment of the above-stated rules at page 280, 2014 Montana Administrative Register, Issue Number 3.

2. The board has amended 17.30.201, 17.30.507, 17.30.516, 17.30.602, 17.30.619, 17.30.622, 17.30.623, 17.30.624, 17.30.625, 17.30.626, 17.30.627, 17.30.628, 17.30.629, 17.30.635, 17.30.702 exactly as proposed. It has amended ARM 17.30.619 as proposed, except that the reference to "Circular DEQ-12A, entitled 'Montana Base Numeric Nutrient Standards' (December 2013 Edition)" has been changed to "Circular DEQ-12A, entitled 'Montana Base Numeric Nutrient Standards' (July 2014 Edition)" to reflect the date of adoption of the circular and has amended ARM 17.30.715 as proposed, but with the following changes, stricken matter interlined, new matter underlined:

17.30.619 INCORPORATIONS BY REFERENCE (1) The board adopts and incorporates by reference the following state and federal requirements and procedures as part of Montana's surface water quality standards:

(a) through (d) remain as proposed.

(e) Department Circular DEQ-12A, entitled "Montana Base Numeric Nutrient Standards" (~~December 2013~~ July 2014 edition), which establishes numeric water quality standards for total nitrogen and total phosphorus in surface waters.

(2) and (3) remain as proposed.

17.30.715 CRITERIA FOR DETERMINING NONSIGNIFICANT CHANGES IN WATER QUALITY (1) The following criteria will be used to determine whether certain activities or classes of activities will result in nonsignificant changes in existing water quality due to their low potential to affect human health or the environment. These criteria consider the quantity and strength of the pollutant, the length of time the changes will occur, and the character of the pollutant. Except as provided in (2), changes in existing surface or ground water quality resulting from the activities that meet all the criteria listed below are nonsignificant, and are not required to undergo review under 75-5-303, MCA:

(a) and (b) remain as proposed.

(c) discharges containing toxic parameters, ~~inorganic nitrogen, or inorganic phosphorus, except as specified in (1)(d) and (e),~~ which will not cause changes that equal or exceed the trigger values in Department Circular DEQ-7. Whenever the change exceeds the trigger value, the change is not significant if the resulting concentration outside of a mixing zone designated by the department does not exceed 15% of the lowest applicable standard;

(d) through (e) remain as proposed.

(f) changes in the quality of water for any harmful parameter, ~~including~~ and parameters listed in Department Circular DEQ-12A, except as specified in (1)(g), for which water quality standards have been adopted other than carcinogenic, bioconcentrating, or toxic parameters, in either surface or ground water, if the changes outside of a mixing zone designated by the department are less than 10% of the applicable standard and the existing water quality level is less than 40% of the standard;

(g) for nutrients in domestic sewage effluent discharged from a septic system that does not require an MPDES or MGWPCS permit, except as specified in (1)(d) and (e), which will not cause changes that equal or exceed the trigger values in Department Circular DEQ-7. Whenever the change exceeds the trigger value, the change is not significant if the changes outside of a mixing zone designated by the department are less than 10% of the applicable standard and the existing water quality level is less than 40% of the standard;

(g) remain as proposed, but is renumbered (h).

(2) and (3) remains as proposed.

(4) If a court of competent jurisdiction declares 75-5-313, MCA, or any portion of that statute invalid, ~~or~~ if the United States Environmental Protection Agency disapproves 75-5-313, MCA, or any portion of that statute under 30 CFR 131.21, or if rules adopted pursuant to 75-5-313(6) or (7), MCA, expire and general variances are not available, then the significance criteria contained in (1)(g) are the significance criteria for total nitrogen and total phosphorus in surface water.

3. The following comments were received and appear with the board's responses:

COMMENT NO 1: The rule proposes uniform, relaxed mixing zone standards. In contrast, EPA policy recommends that mixing zone characteristics be defined on a case-by-case basis after it has been determined that the assimilative capacity of the receiving system can safely accommodate the discharge. EPA also states that the assessment should take into consideration the physical, chemical, and biological characteristics of the discharge and the receiving system; the life history and behavior of organisms in the receiving system; and the desired uses of the waters.

RESPONSE: The proposed rules provide for the use of the entire 14Q5 flow in dilution calculations involving a standard mixing zone. The issue of appropriate mixing zones for base numeric nutrient standards was discussed during Nutrient Work Group meetings and EPA was involved in the discussions. Although EPA's national policy is for case-by-case analysis, as noted by the commenter, much of the concern regarding such case-by-case analysis revolves around toxic compounds, which have both chronic and acute impact levels. Mixing zones are designed to make sure acute levels are not exceeded in the mixing zone because this would harm any aquatic life present. In contrast, nutrients at the concentrations of the base numeric nutrient standards behave like chronic criteria, and the changes to the mixing zone rules reflect this reduced potential for direct aquatic life impact.

COMMENT NO. 2: Endnote 4 in Circular DEQ-12A should be rephrased to clarify whether it applies to the development of permit limits or stream assessment for listing/delisting on Montana's 303(d) list. Further, it should be clarified that it is a monthly average, not a 30-day rolling average.

RESPONSE: The board agrees that this should be clarified. Section 2.2 of the circular states that permit limits for nutrient discharges are to be developed using the Average Monthly Limit (AML). Thus, the circular already provides-for permitting purposes-that the averaging timeframe (duration) for permitting nutrient discharges is 30 days. The board also agrees that the 30-day period should not be a rolling average. Section 2.2 has also been modified to provide that this is a calendar month.

In contrast to setting permit limits, when assessing a stream's ambient condition for 303(d) listing purposes, the department's monitoring and assessment unit collects nutrient samples throughout the growing season (a three-month period each year) and evaluates all data using statistical testing procedures. It does not restrict the evaluation to a calendar month. Footnote 4 has been updated to better reflect the monitoring and assessment process. It now reads: "The average concentration during a period when the standards apply may not exceed the standards more than once in any five-year period, on average." In relation to the duration and frequency requirements of the standards, it should be noted that, because permits are written to a shorter time frame (a calendar month), they are fully protective of the standard. In addition, the monitoring and assessment unit of the department evaluates biological data in concert with the nutrient concentrations to make a final assessment.

COMMENT NO. 3: Some commenters supported adoption of Flathead Lake TMDL Phase I targets as Flathead Lake water-quality standards, but stated that a document describing the technical and scientific support for the standards is needed first. Other commenters asked for postponement of the standards adoption for Flathead Lake pending a more thorough technical review and local stakeholder involvement.

RESPONSE: The board agrees that the standards for Flathead Lake should be postponed and that more details on the scientific and technical basis of the standards should be prepared. The standards for Flathead Lake have been removed from Department Circular DEQ-12A.

COMMENT NO. 4: Nutrient standards should be adopted for the Flathead River.

RESPONSE: Numeric standards for the Flathead River were not proposed for the Flathead River in the notice of public hearing, and therefore adoption of standards for that stream is not within the scope of this rulemaking. It would require commencement of a new rulemaking proceeding.

COMMENT NO. 5: Required reporting limits in Table 12A-3 of Department Circular DEQ-12A for total kjeldahl nitrogen (TKN) are not obtainable and should not be adopted in rule.

RESPONSE: The board agrees and has increased the required reporting value (RRV) for TKN from 150 µg/L to 225 µg/L. The board is aware that the typical practical quantitation limit for total kjeldahl nitrogen (TKN) is 0.5 mg/L, or 500 µg/L. However this limit is not low enough to meet some nitrogen standards proposed in Circular DEQ-12A. If a reporting limit is greater than or equal to an applicable standard, state waters that have non-detectable levels of TKN may be unnecessarily listed as impaired and/or may have requirements for extensive sampling. This is why the board calculates an RRV for parameters which have numeric standards. The board has worked with the department and statewide analytical laboratories to derive the updated RRV. The board understands that this number is still very low and that conditions must be optimal in order to achieve this number. Reporting value concentrations higher than 225 µg/L may also be acceptable on a case-by-case basis, as indicated in Table 12A-3 of Department Circular DEQ-12A. Additionally, if a sample must be diluted, it is understood that the reporting limit will be raised.

COMMENT NO. 6: The board should consider expanding the period of application of the standards, where needed, because nutrient problems can manifest at times other than in summer/early fall due to low snowpack, early irrigation withdrawal, etc.

RESPONSE: The board is aware that eutrophied rivers and streams can manifest a period of excess algal growth in early spring prior to runoff. This has been observed in the Clark Fork River, for example. In western Montana, this algal growth is typically scoured away by the spring runoff and algal growth recommences in late June to early July. Application of nutrient standards is not necessary during the spring because spring is relatively short (typically a few weeks), generally before the main recreation season, and followed by a scouring period, flow volume is not an appropriate factor for determining the end date for application of nutrient criteria.

After runoff ends, base flow begins and can be fairly uniform into November and December. However, regional climatic influences, such as lowered temperatures and light intensity, typically cause, by early October, major reductions in aquatic plant life growth, reductions in aquatic macroinvertebrate productivity, and higher dissolved oxygen concentrations. Essentially, the productive period for the year ends in the fall and the importance of nutrient concentrations to this productivity greatly declines. The proposed end dates for the period of application of the nutrient criteria reflect this.

COMMENT NO. 7: Site-specific criteria or modified periods of application should be used to tighten nutrient standards where it is apparent that nuisance algae are becoming worse or not improving.

RESPONSE: The board agrees. These changes would be made in another rulemaking proceeding that would be initiated once it is determined that the existing standards do not protect the use.

COMMENT NO. 8: In ARM 17.30.715(1)(c), the words "inorganic nitrogen, or inorganic phosphorus" should be deleted. If they are not, discharges of nutrients will be subject to the non-significance of both (1)(c) and (1)(f). Based on numerous meetings of the nutrient work group and early drafts, non-significance should be determined under (1)(f) using the base numeric nutrients standards rather than under the existing narrative standard.

RESPONSE: The board agrees. ARM 17.30.715(1)(c) has been modified so that non-significance will be determined under ARM 17.30.715(1)(f) for all discharges that require a surface or groundwater permit. Additionally, for discharges that do not require one of those permits, but which undergo non-degradation review nonetheless, a new subsection, ARM 17.30.715(1)(g), has been added. This subsection retains the trigger value requirement as the initial criterion for significance. Retention of the trigger value will allow the department's subdivisions program, which has stringent deadlines for reviewing subdivisions, to continue to use an expeditious means of determining significance for small subdivisions. Failing this first test, the next test for non-significance will be the same as is found in ARM 17.30.715(1)(f); that is, a change is non-significant if the change is < 10% of the numeric nutrient standards and existing water quality is currently < 40% of those standards.

COMMENT NO. 9: Movement from the 7Q10 to the seasonal 14Q5 for nutrients is a poor policy choice. The board should stick with the 7Q10 for permitting base numeric nutrient standards.

RESPONSE: The low-flow design flow is chosen to ensure compliance with the water quality criteria. Given that the proposed criteria are to be permitted based on a 30-day average concentration, and have an allowable excursion frequency of once in five years, it is overly restrictive to consider the 7Q10. When establishing permit limits, the 14Q5 ensures protection of the proposed criteria at a level that corresponds to the averaging period and allowable excursion frequency of the underlying criteria, while simultaneously providing a margin of safety because the low-flow averaging period is somewhat shorter (14 days instead of 30).

COMMENT NO. 10: A commenter pointed out that the nonseverability provision in 17.30.619 is triggered by the expiration of the general variance rules, while the nonseverability provision in 17.30.715 is not. The commenter stated that this should be a trigger in both rules. Several other commenters stated that the board should modify the non-severability clauses to be triggered if EPA objects to or vetoes a permit based on the inclusion of a variance.

RESPONSE: The board agrees that the triggers for both provisions should be the same and has added the expiration of the variance rules to the nonseverability provision in ARM 17.30.715.

The board has not included the permit trigger. If EPA disapproves the general variance, the nonseverability provisions would apply. Once EPA approves the general variance, EPA would not have authority to object to or veto a permit for an existing discharger based on inclusion of general variance limits in the permit. In written comments on this rulemaking, EPA has indicated that variances may be available to new dischargers as long as existing uses are protected. EPA personnel have indicated to department personnel that inclusion of the permit trigger for nonseverability would likely result in EPA rule disapproval. Should this occur, the narrative standards would be reinstated. However, narrative standards are implemented in a permit by imposition of numeric limits, and it is possible that a court would hold that the proper application of the narrative standards results in the same numeric permit limits as the numeric standards would require. However, because the variance provisions of 75-5-313, MCA, take effect only upon adoption of numeric standards, that statute would not be available to new or existing dischargers in those instances.

COMMENT NO. 11: The rules should clarify implementation of nondegradation for existing and future permits. The board should recognize the seasonal nature of the nutrient standards when implemented in permits and nondegradation provisions.

RESPONSE: Nondegradation requirements do not apply to existing permittees unless they become increased sources, as defined in ARM 17.30.702(18). For new or increased sources, as defined in ARM 17.30.702(18), nondegradation for base numeric nutrient standards will be applied following ARM 17.30.715.(1)(f), (1)(g), (2), and (3). If this process results in a finding that degradation will occur, the applicant can apply for an authorization to degrade.

Circular 12A clearly provides that the standards are seasonal in nature. The department would therefore be legally bound to recognize this seasonal nature in permitting, including application of nondegradation.

COMMENT NO. 12: Applying criteria "as an average, not to be exceeded more than once in any three year period, on average" needs clarification.

RESPONSE: The allowable excursion frequency (once in five years in this case) is referred to "on average" in order to accommodate datasets longer or shorter than the specified five-year period. For example, if the dataset were ten years long, standards exceedences would be allowed twice in that period (2/10 years, equal to 1/5), but not three times.

COMMENT NO. 13: The rules should be clarified to show that these seasonal nutrient standards will not be applied to storm water discharges.

RESPONSE: All discharges, including storm water discharges, are subject to water quality standards, whether those standards are narrative, as the nutrient standards are currently structured, or numeric, as is proposed for nutrients.

COMMENT NO. 14: The overall nutrient standards package (including variances) cannot result in a regulatory moratorium on new business in Montana.

RESPONSE: The purpose of the variance processes is to assure that the economic effects of nutrient standards will not cause a regulatory moratorium on new business in Montana. In turn, the rules that have been developed to implement the statute reflect this intent. Variances can be granted to new businesses so long as the new dischargers show that the variance protects the existing use.

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Reviewed by: BOARD OF ENVIRONMENTAL REVIEW

__________________________ By: ________________________________

JOHN F. NORTH
Rule Reviewer

ROBIN SHROPSHIRE
Chairman

Certified to the Secretary of State, ______________, 2014.

FRED VAN VALKENBURG
MISSOULA COUNTY ATTORNEY
D. James McCubbin
Deputy County Attorney
200 West Broadway
Missoula, Montana 59802
(406) 258-4737
jmccubbin@co.missoula.mt.us

Attorneys for Missoula Valley
Water Quality District

BEFORE THE BOARD OF ENVIRONMENTAL REVIEW
OF THE STATE OF MONTANA

IN THE MATTER OF: THE NOTICE OF APPEAL AND REQUEST FOR HEARING BY MISSOULA COUNTY REGARDING DEQ'S ISSUANCE OF MPDES PERMIT NO. MT0000035 ISSUED TO M2GREEN REDEVELOPMENT'S SITE IN FRENCHTOWN, MT	Case Nos. : BER 2014-02 WQ & BER 2014-03 WQ STIPULATION FOR DISMISSAL OF ADMINISTRATIVE APPEAL

COME NOW the Montana Department of Environmental Quality ("DEQ"), Missoula Valley Water Quality District, Clark Fork Coalition, and the Confederated Salish and Kootenai Tribes, each by and through counsel, and hereby stipulate to dismissal without prejudice of this appeal, subject to the following agreed terms:

1. Dismissal of Administrative Appeal:

a. The parties stipulate that Case Nos.: BER 2014-02 WQ & BER 2014-03 WQ should be dismissed without prejudice upon approval of the terms of this Stipulation by the Hearings Examiner.

b. The parties agree that this administrative appeal may be re-opened if any Montana district court or the Montana Supreme Court determine that original jurisdiction for appellants' and intervenor's challenges to Montana Pollutant Discharge Elimination System Permit MT0000035 properly lies with the Board of Environmental Review. DEQ will not object if the other parties to this Stipulation seek to re-open the administrative appeal within 30 days following any such order.

2. District Court proceedings: DEQ will not challenge jurisdiction of the Montana district court in a petition for judicial review to challenge DEQ's issuance of Montana Pollutant Discharge Elimination System Permit MT0000035. Subject to Mont. Code Ann. § 75-1-107, the scope of this stipulation is for claims or challenges to the MPDES Permit that are equivalent to the claims/challenges raised in BER 2014-02 WQ and/or BER 2014-03 WQ. DEQ stipulates that each of the other parties to this Stipulation has standing to bring such a Petition in district court.

3. Statute of Limitations: DEQ stipulates that it will raise no statute of limitations defense that would bar the appellants or the intervenor from challenging the issuance of Montana Pollutant Discharge Elimination System Permit MT0000035 through a petition for judicial review filed in district court. Subject to Mont. Code Ann. § 75-1-107, the scope of this stipulation is for claims or challenges to the MPDES Permit that are

equivalent to the claims/challenges raised in BER 2014-02 WQ and/or BER 2014-03 WQ.

4. No Admissions: The parties stipulate that their agreement to this Stipulation is not intended to, and shall not be taken as, any admission regarding the merit of claims and legal arguments made in this administrative appeal.
5. Multiple parts and copies: This Stipulation may be executed in multiple parts and copies, which taken together shall be equivalent to a single original.

DATED this 24th day of June, 2014.

D. James McCubbin
Legal Counsel for Missoula Valley Water Quality District

DATED this ____ day of __________________, 2014.

____
Jack Tuholske
Legal Counsel for Clark Fork Coalition

DATED this ____ day of __________________, 2014.

____
John T. Harrison
Legal Counsel for Confederated Salish and Kootenai Tribes

equivalent to the claims/challenges raised in BER 2014-02 WQ and/or BER 2014-03 WQ.

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DATED this 24 day of June, 2014.

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D. James McCubbin
Legal Counsel for Missoula Valley Water Quality District

DATED this ____ day of ____________________, 2014.

____
Jack Tuholske
Legal Counsel for Clark Fork Coalition

DATED this ____ day of ____________________, 2014.

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John T. Harrison
Legal Counsel for Confederated Salish and Kootenai Tribes

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DATED this ____ day of ____________________, 2014.

____
D. James McCubbin
Legal Counsel for Missoula Valley Water Quality District

DATED this ____ day of ____________________, 2014.

____
Jack Tuholske
Legal Counsel for Clark Fork Coalition

DATED this 25th day of June, 2014.

____
John T. Harrison
Legal Counsel for Confederated Salish and Kootenai Tribes

DATED this 25 day of June, 2014.

Kurt Moser
Legal Counsel for Department of Environmental Quality

CERTIFICATE OF SERVICE

I hereby certify that this 1st day July, 2014, I caused to be served a true and correct copy of the foregoing documents to all parties or their counsel of record as set forth below:

Martha E. McClain
D. James McCubbin
Deputy County Attorneys
Missoula County Courthouse
200 W. Broadway
Missoula, MT 59802
(406) 258-4737
Email: jmccubbin@co.missoula.mt.us
Attorneys for Missoula County/Missoula County Water Quality District

[✓] U.S. Mail, postage prepaid
[ ] Electronic Mail
[ ] Facsimile Transmission
[ ] Personal Delivery

Jack Tuholske, Esq.
Tuholske Law Office, P.C.
234 E. Pine Street
P.O. Box 7458
Missoula, MT 59802
Phone: (406) 396-6415
Email: tuholske@gmail.com
Attorney for Clark Fork Coalition

[✓] U.S. Mail, postage prepaid
[ ] Electronic Mail
[ ] Facsimile Transmission
[ ] Personal Delivery

John T. Harrison
Rhonda R. Swaney
Confederated Salish and Kootenai Tribes
P.O. Box 278
Pablo, MT 59855
(406) 675-2700
Email: johnh@cskt.org
Email: rhondas@cskt.org
Attorneys for Intervenor Confederated Salish and Kootenai Tribes

[✓] U.S. Mail, postage prepaid
[ ] Electronic Mail
[ ] Facsimile Transmission
[ ] Personal Delivery

Ben Reed, Esq.
Hearing Examiner
Agency Legal Services Bureau
1712 Ninth Avenue
P.O. Box 201440
Helena, MT 59620-1440
Phone: (406) 444-0160
Email: BenReed@mt.gov

[✓] U.S. Mail, postage prepaid
[ ] Electronic Mail
[ ] Facsimile Transmission
[ ] Personal Delivery

Debra Sutliff, Administrative Assistant
MT-Dept. of Environmental Quality

Filed with the
MONTANA BOARD OF
ENVIRONMENTAL REVIEW
This 14 day of July, 2014
at 1:40 o'clock p.m.
By: [signature]

Tyler Myrstol
Myrstol Logging, Inc.
P.O. Box 189
Clyde Park MT 59018
Phone: 406-220-0020

Appellant

IN THE MATTER OF: VIOLATIONS OF THE CLEAN AIR ACT OF MONTANA BY MYRSTOL LOGGING, INC., CLYDE PARK, PARK COUNTY, MONTANA [FID 2295, DOCKET NO. AQ–14–02]	Case No. BER 2014-04 AQ **WITHDRAWAL OF REQUEST FOR CONTESTED CASE**

Myrstol Logging, Inc. hereby withdraws its request for a contested case hearing in this case and dismisses this case.

7-10-14
Date

[signature]
Tyler Myrstol
Myrstol Logging Inc.

CERTIFICATE OF SERVICE

I hereby certify that on 7 10, 2014, I caused a complete and accurate copy of the foregoing document to be served on the following person by first class U.S. mail, postage prepaid, addressed to:

Department of Environmental Quality
Legal Unit- Attn Norman J. Mullen
Po Box 200901
Helena MT 59620-0901