Using a Computer Water Quality Model to Derive Numeric Nutrient Criteria

LOWER YELLOWSTONE RIVER, MT

Photo: Montana DEQ; Mark Greytak, MontanaPictures.Net

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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:
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- 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 7^{th} 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.

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- 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 Agriculture 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.

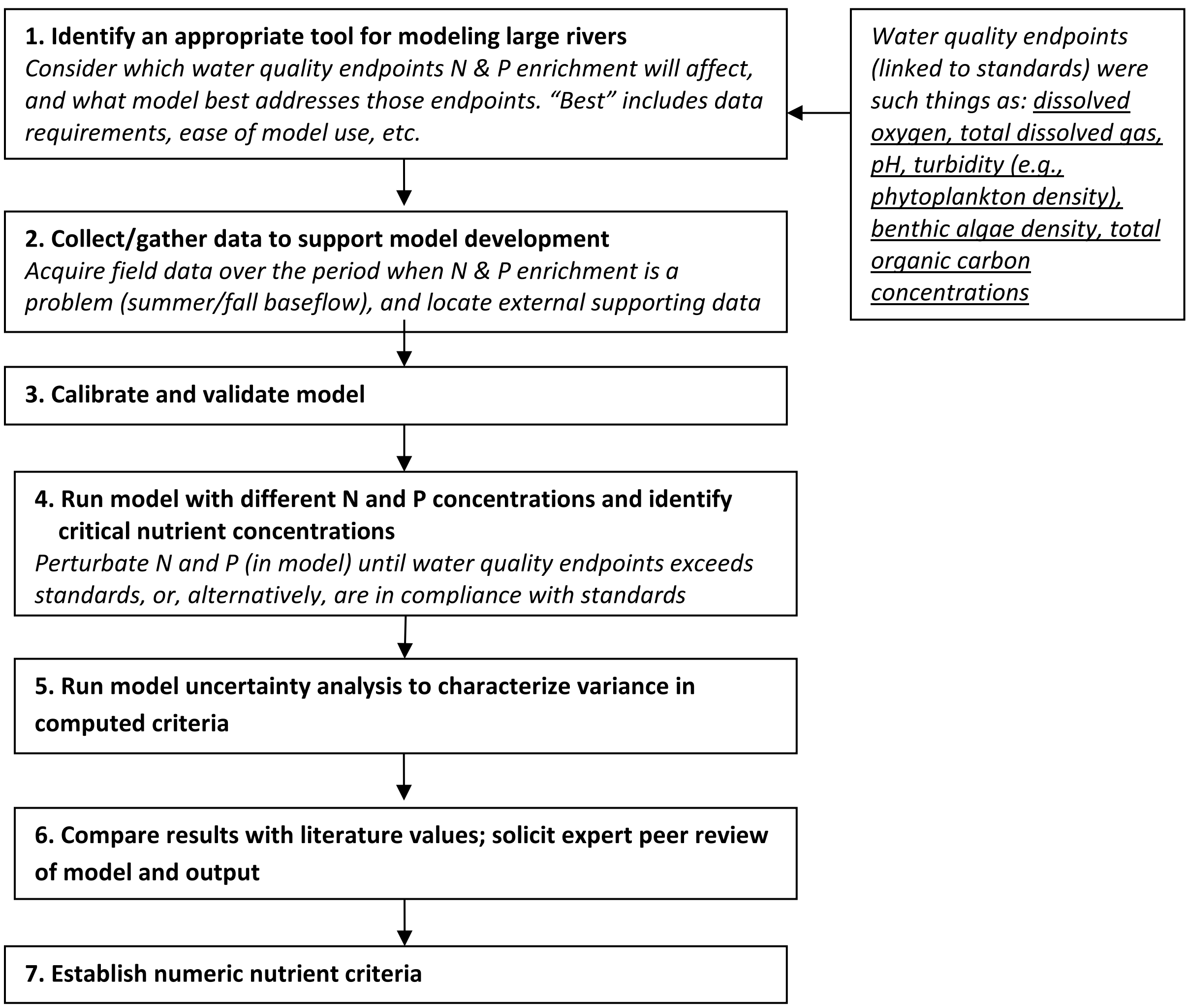

Figure 1-1. Montana's proposed approach for large river numeric nutrient criteria development.

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.

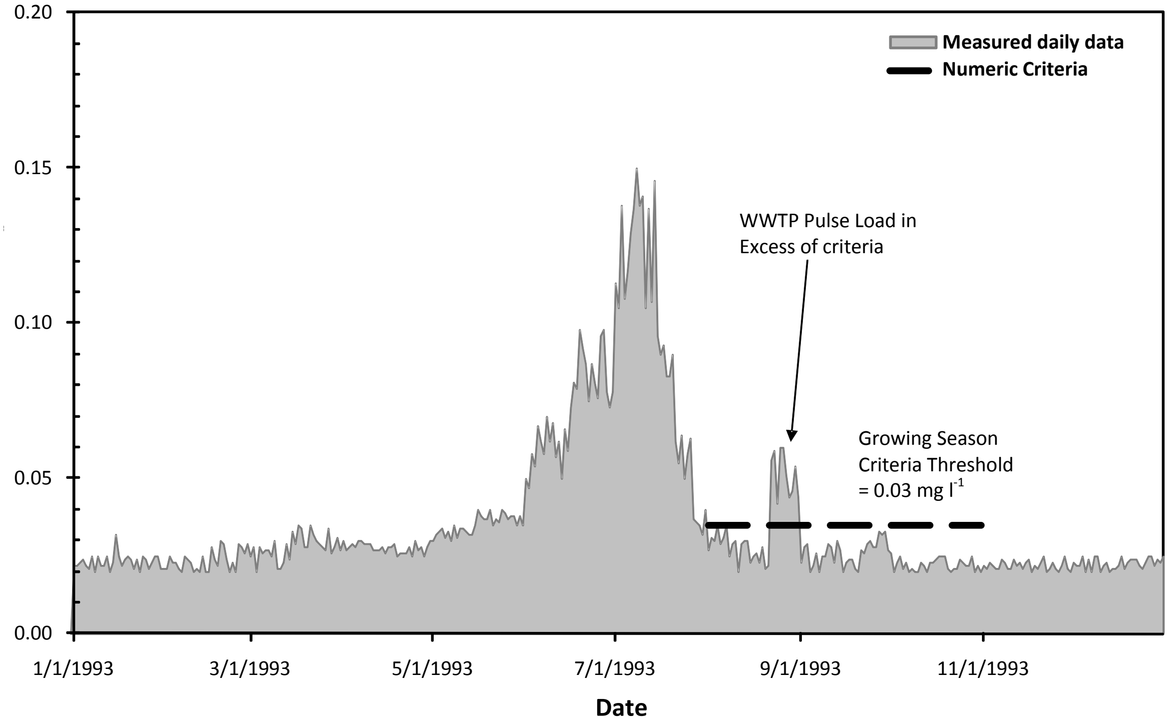

Figure 1-2. Hypothetical example of numeric nutrient criteria for total P in a Montana river. Nutrient levels in excess of the proposed criteria are indicative of beneficial use impairment. Concentrations below the criteria would support their intended uses. This hypothetical example illustrates probable 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.

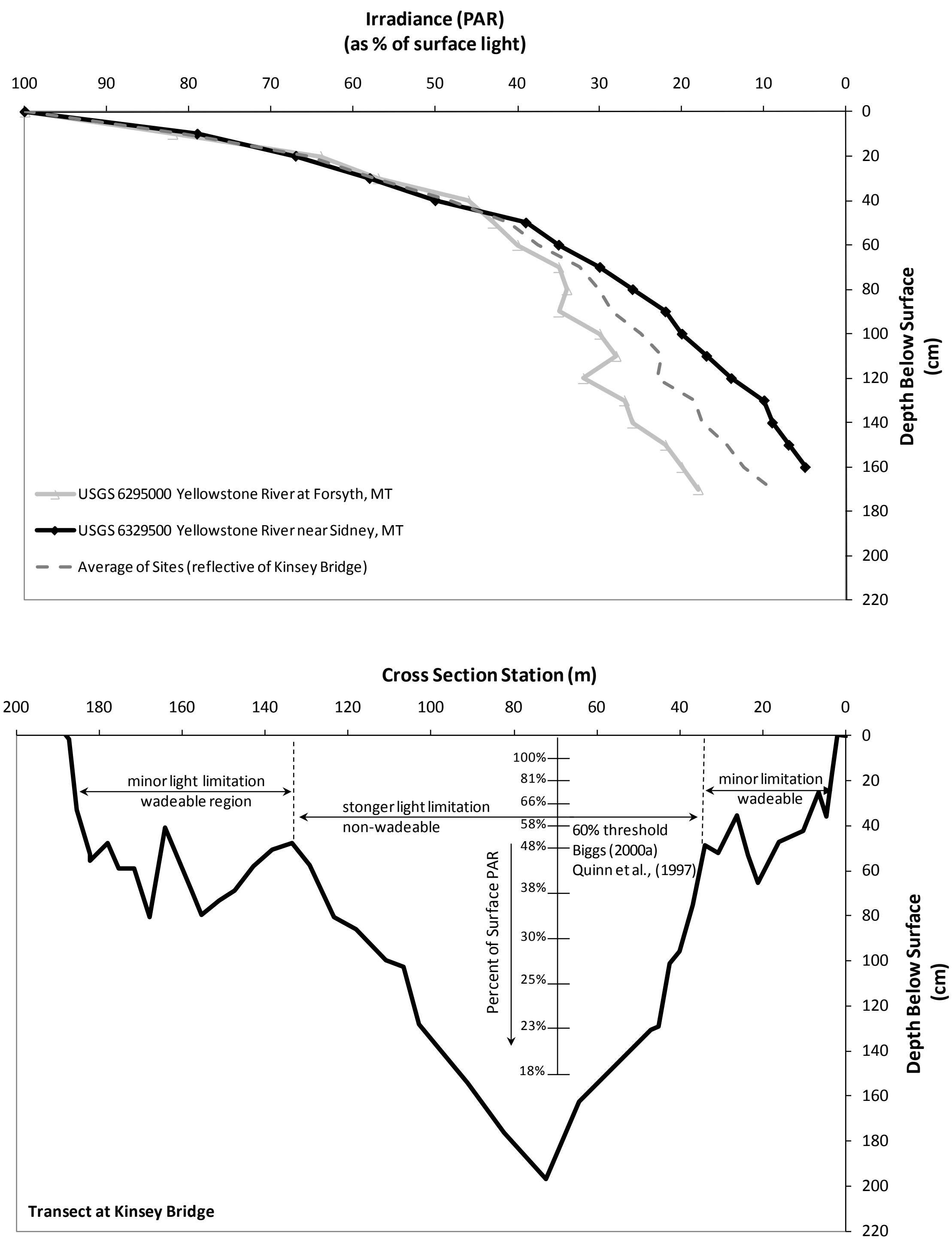

Figure 2-2. Light extinction in a Montana river and its lateral extent.
(Top panel) Light is quickly attenuated vertically in a Montana large river. In this example, only 50% of the surface light is available at 40 cm (15 inches) and drops to 25% at 120 cm (4 ft). Data from U.S. Geological survey (Peterson, 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).

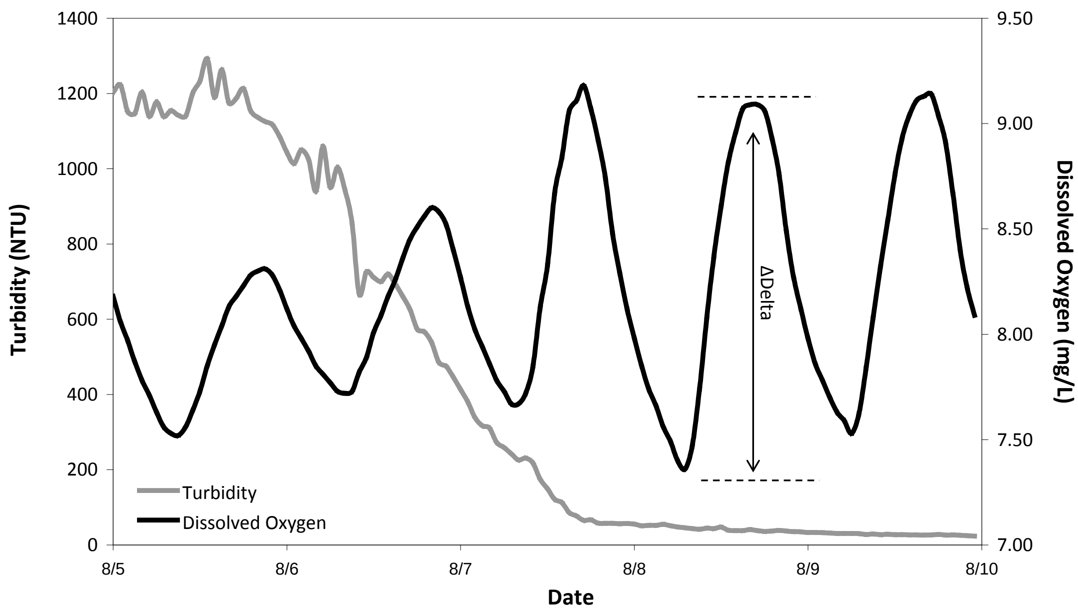

Figure 2-3. Influence of light attenuation on productivity in a Montana River.

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)].

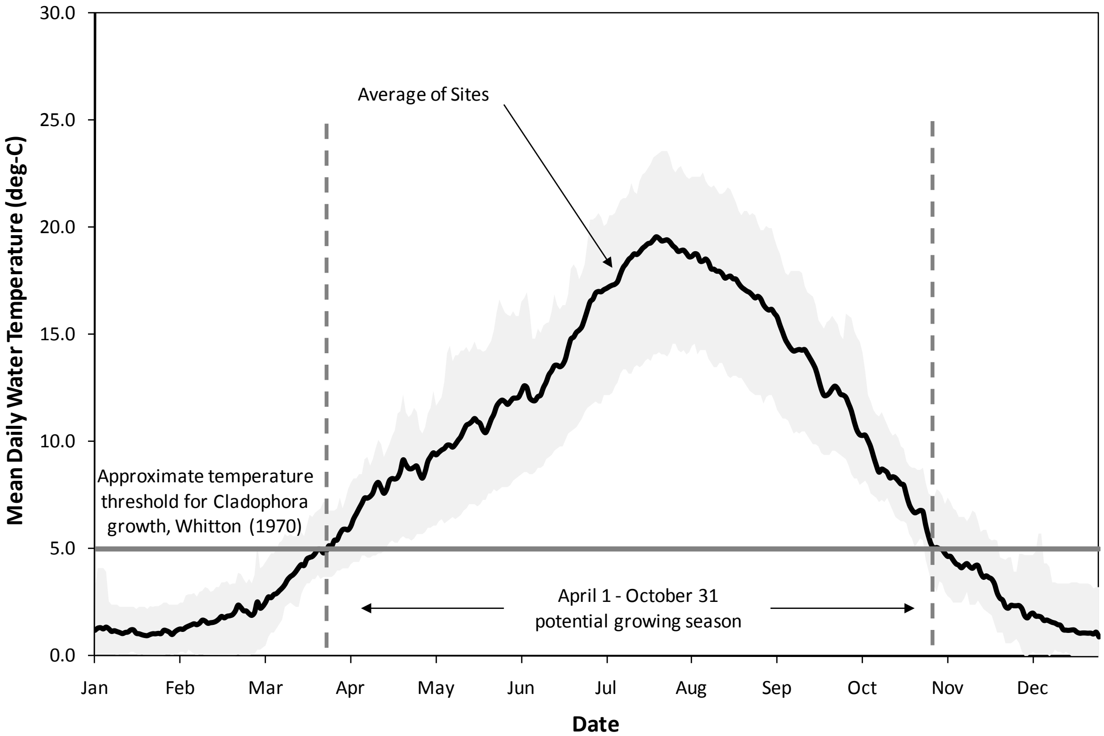

Figure 3-1. Plot of mean daily water temperature against algal growth limiting threshold.
Data includes a compilation of nine rivers in Montana with sufficient record to characterize long-term variation in water temperature. Sites include the Bitterroot, Blackfoot, Clark Fork, Dearborn, Flathead (2 sites), 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].

[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}.

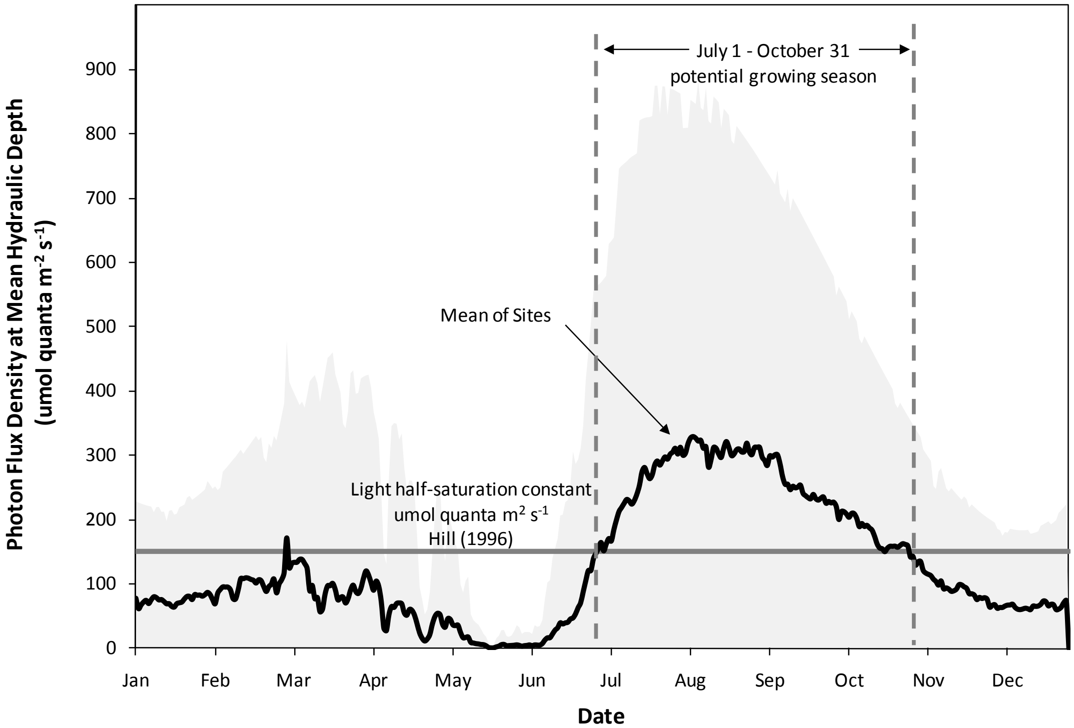

Figure 3-2. Plots of daily photon flux density against algal growth limiting threshold.
Based on a compilation of Montana Rivers[7].

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.

[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).

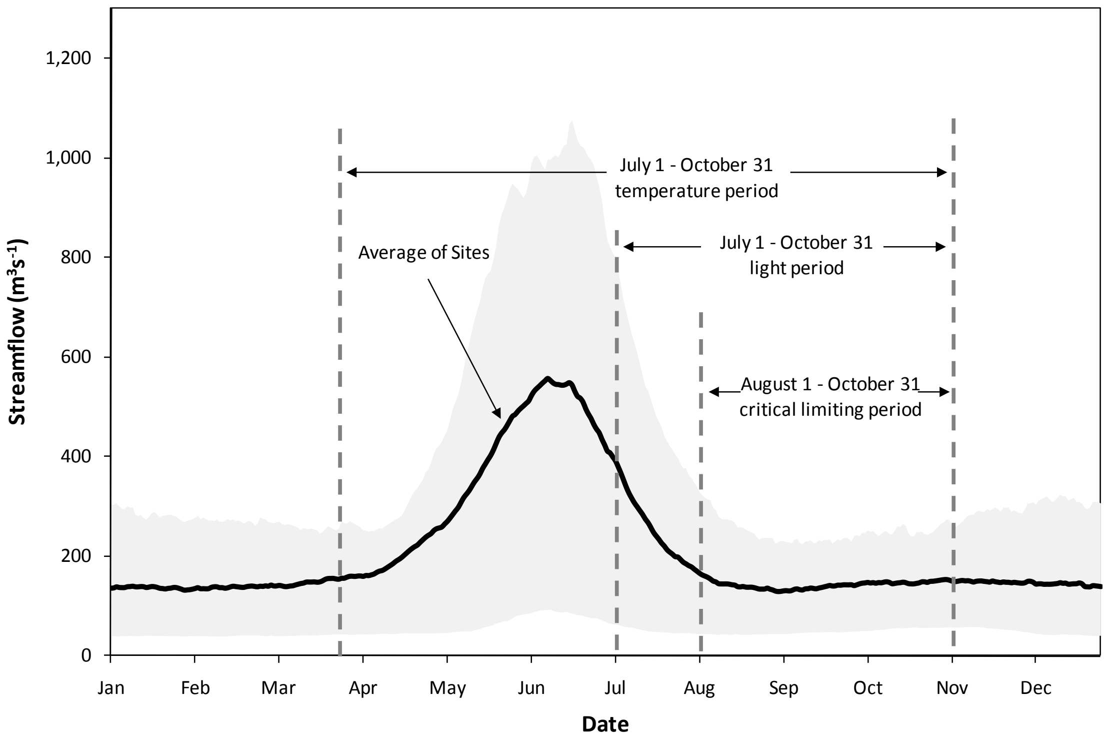

Figure 3-3. Streamflow hydrology and the critical period for large river criteria development. The most restrictive period relative to temperature, light, and streamflow is from August 1-October 31. Monitoring and assessment activities should target this timeframe for large river criteria development.

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.

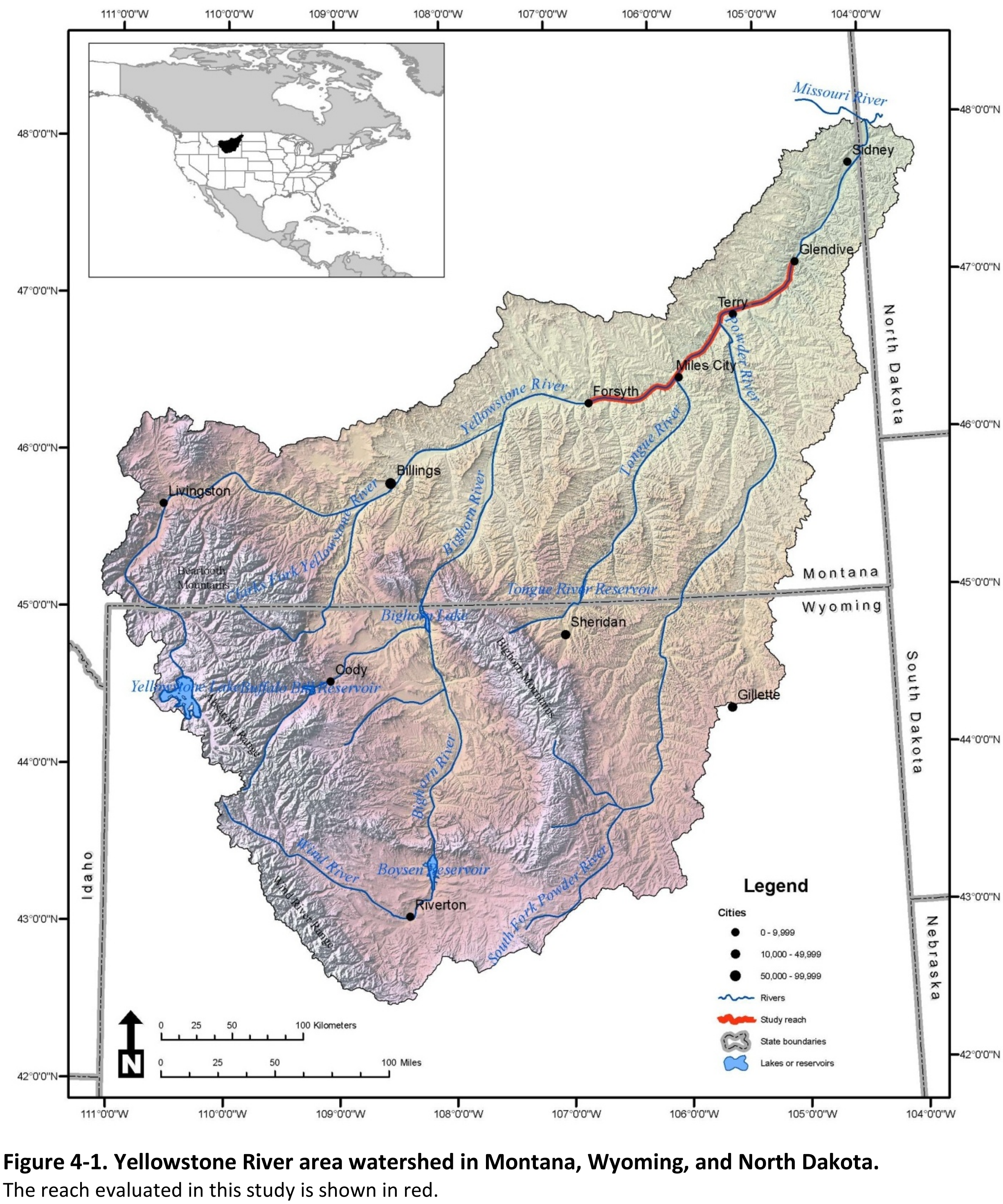

Figure 4-1. Yellowstone River area watershed in Montana, Wyoming, and North Dakota.
The reach evaluated in this study is shown in red.

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.

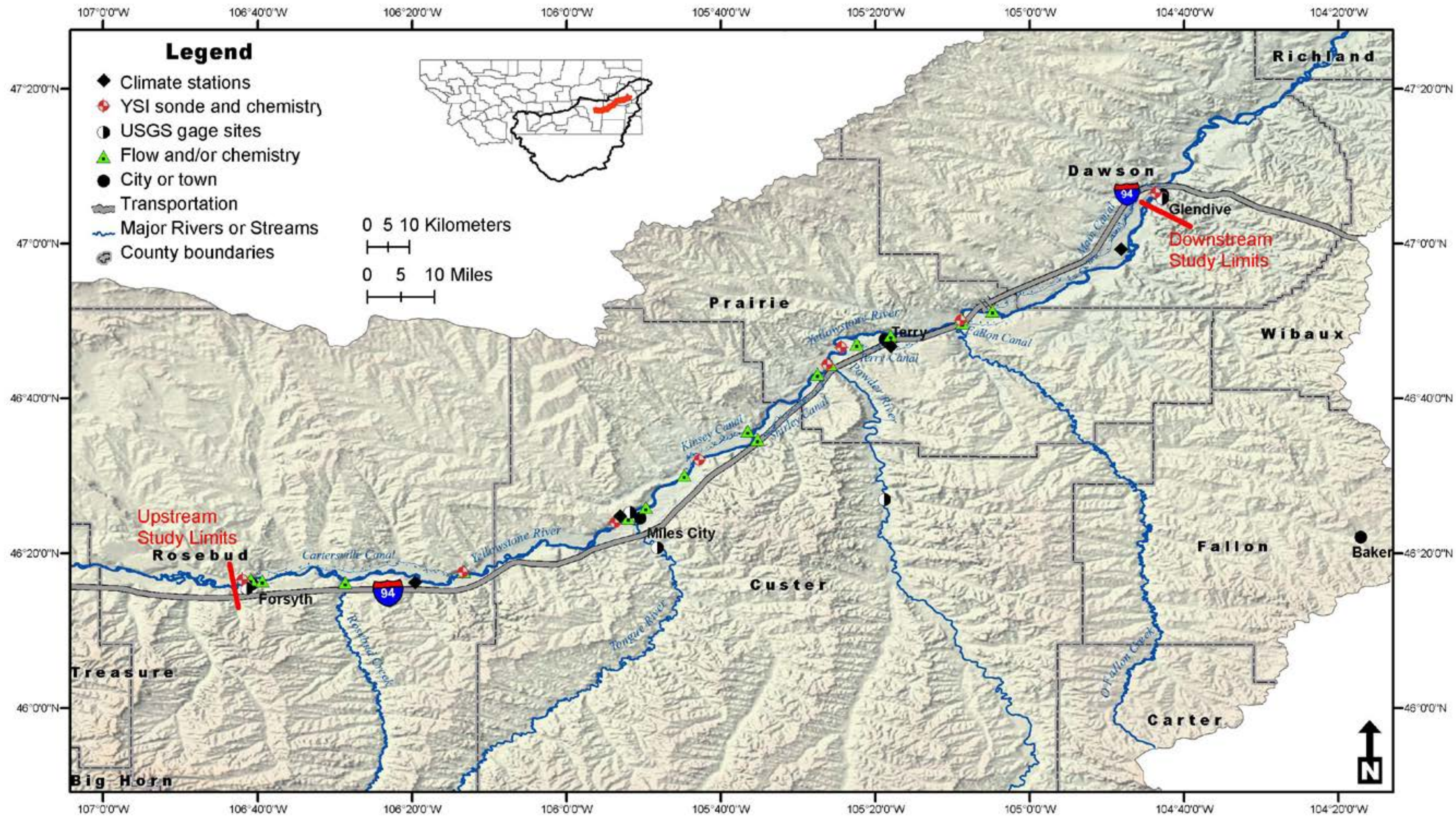

Figure 4-2. Lower Yellowstone River study area showing monitoring locations and other features.

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

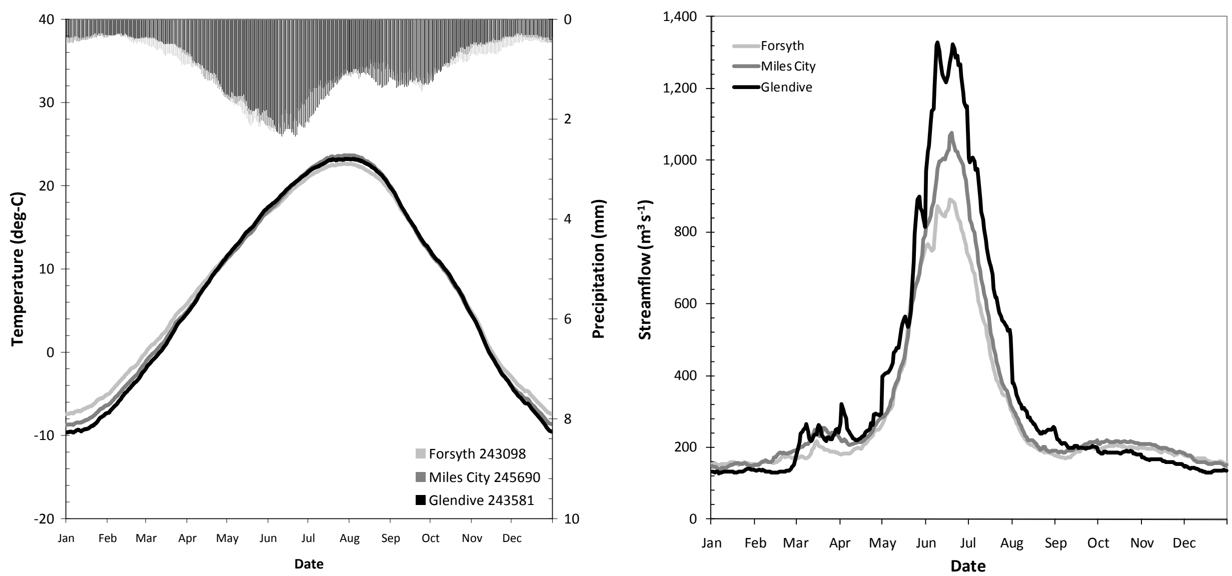

Figure 4-4. Mean daily normals for the lower Yellowstone River.
(Left panel) 1971-2000 precipitation and temperature for Forsyth (243098), Miles City Airport (245690), and Glendive, MT (243581), (Right panel) 2000-2008 mean daily streamflow at USGS Yellowstone River at Forsyth (06295000), Miles City (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.

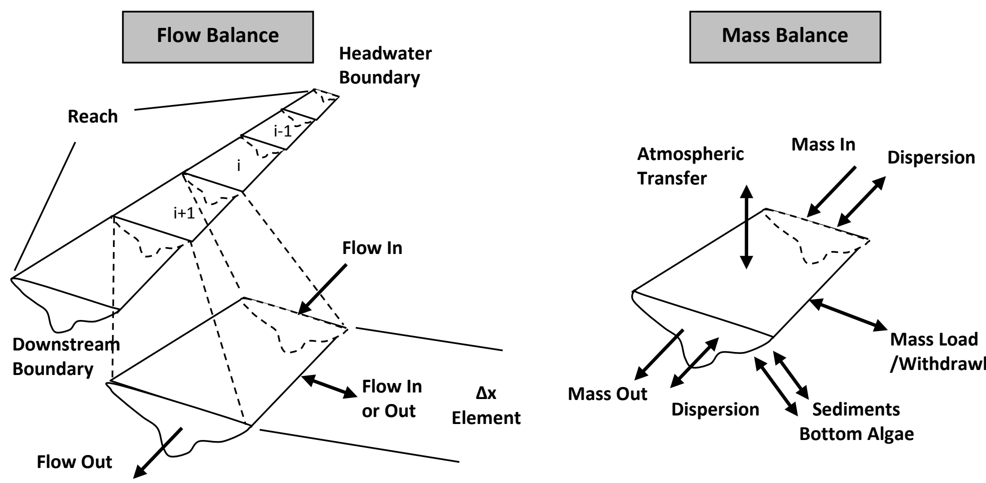

Figure 5-1. Conceptual representation of Q2K (redrawn from Brown and Barnwell, 2004).
(Left panel) Flow balance. (Right panel) Mass balance.

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 [^oC], 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 $^oC)^{-1}$], $J_{a,i}$ = 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}

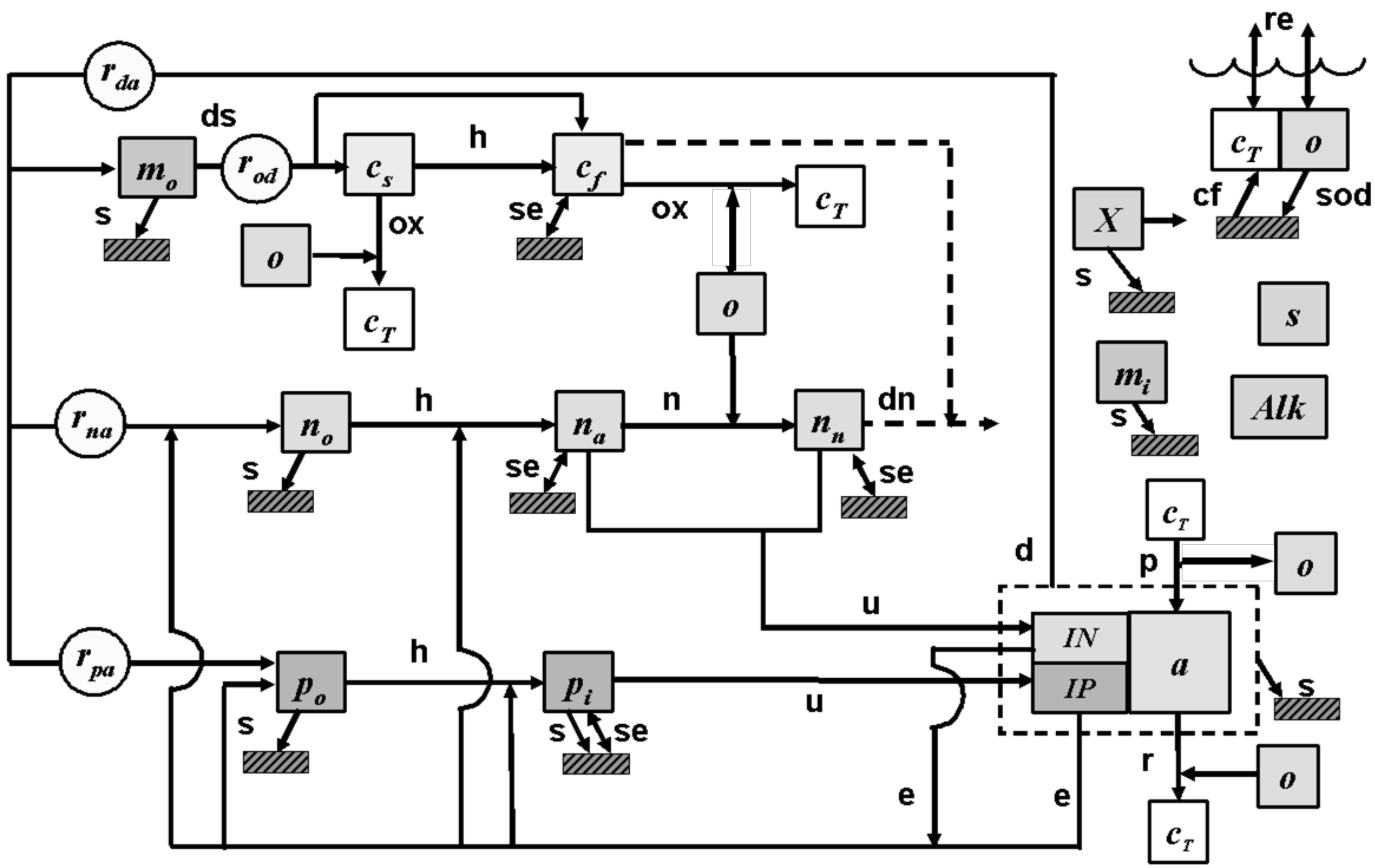

Figure 5-2. Diagram of model kinetics and mass transport processes in Q2K.
Redrawn from Chapra et al., 2008 (with permission). Kinetic processes are as follows: ds = dissolution, h = hydrolysis, ox = oxidation, n = nitrification, dn = denitrification, p = photosynthesis, r = respiration, e = excretion, 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 $(m^3d)^{-1}$ or mg $(m^3d)^{-1}$]. 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

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[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.

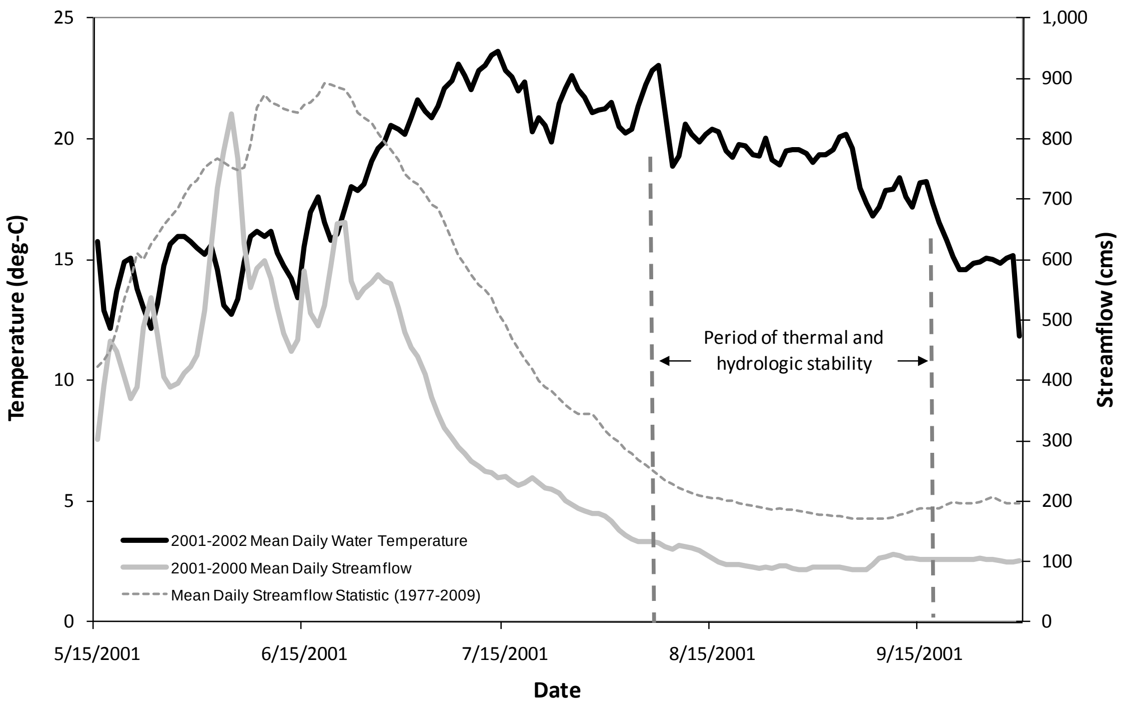

Figure 5-3. Typical occurrence of thermal and hydrologic stability in the Yellowstone River.
The onset of thermal and hydrologic stability begins approximately August 1 and continues into late September. Temperature data from 2001-2002 were obtained by taking the flow-weighted average of USGS 06214500 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.

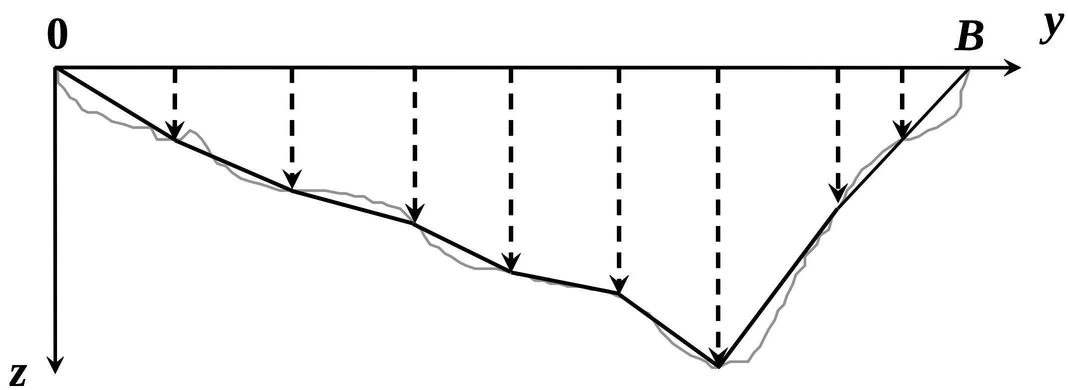

Figure 5-4. Conceptual representation of the AlgaeTransect2K (AT2K) model.
The model represents a river transect as a single Q2K element with variable depth to evaluate the effect of lateral light attenuation on algal growth. The primary consideration in the development of AT2K was to make it consistent 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 Climate/Air Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	daily hourly 184 188 134 214 136 127 13 yes	1946-2008 1936-2008 1974-2007 1974-2007 1971-2007 1974-2007 1971-2007 1965-2007 1975-2007 2001,2007
Terry	06326530	4086YE01, Y23YELLR02, Y23YELLR03	Flow Climate/Air Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	16 hourly 109 112 19 122 20 103 14 yes	1974-1979 1998-2008 1974-2007 1974-2007 1974-2007 1974-2007 1974-2007 1975-2007 1975-2007 2001,2007
Glendive	06327500	4490YE01, Y23YELLR04	Flow Climate/Air Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	daily hourly 2 2 16 14 17 9 4 yes	2002-2008 1973-2008 2007 2007 1976-2007 1976-2007 1973-2007 1975-2007 2007 2001,2004
Sidney	06329500	NA	Flow Climate/Air Total N Ammonia ($NH_{3/4}$) NO_2+NO_3 Total P SRP Solids Algae (either) Feature	daily hourly 333 468 281 426 272 427 10 yes	1933-2008 1973-2008 1970-2007 1969-2007 1971-2007 1969-2007 1971-2007 1965-2008 1975-2005 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.

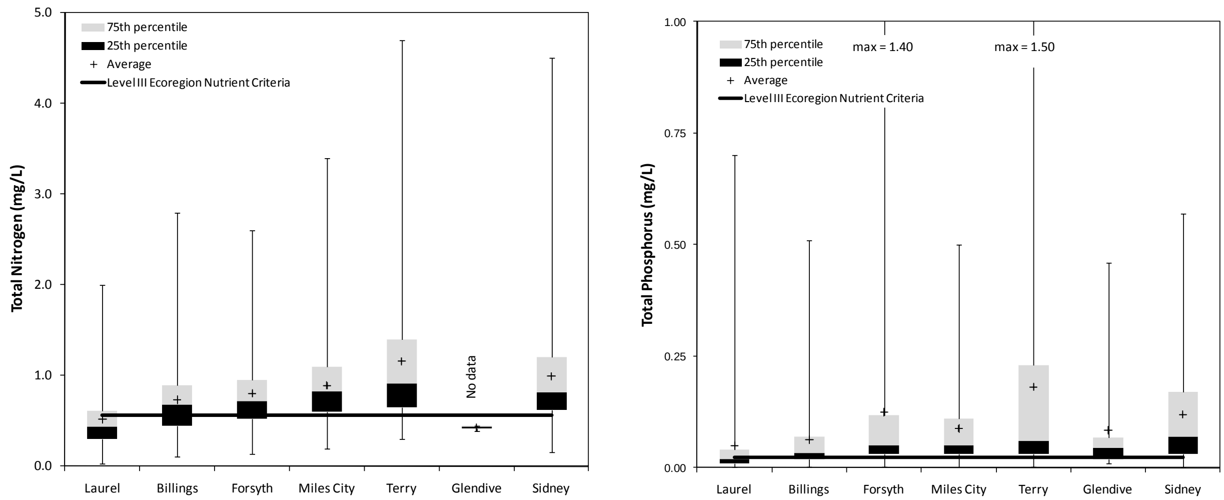

Figure 6-1. Historical nutrient concentrations in the lower Yellowstone River.
(Left panel) Historical TN data on the Yellowstone River. (Right panel). Same but for TP. Data are shown over the period of record for each site (1969-2007), and in both instances are well above the Level III ecoregional criteria (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.

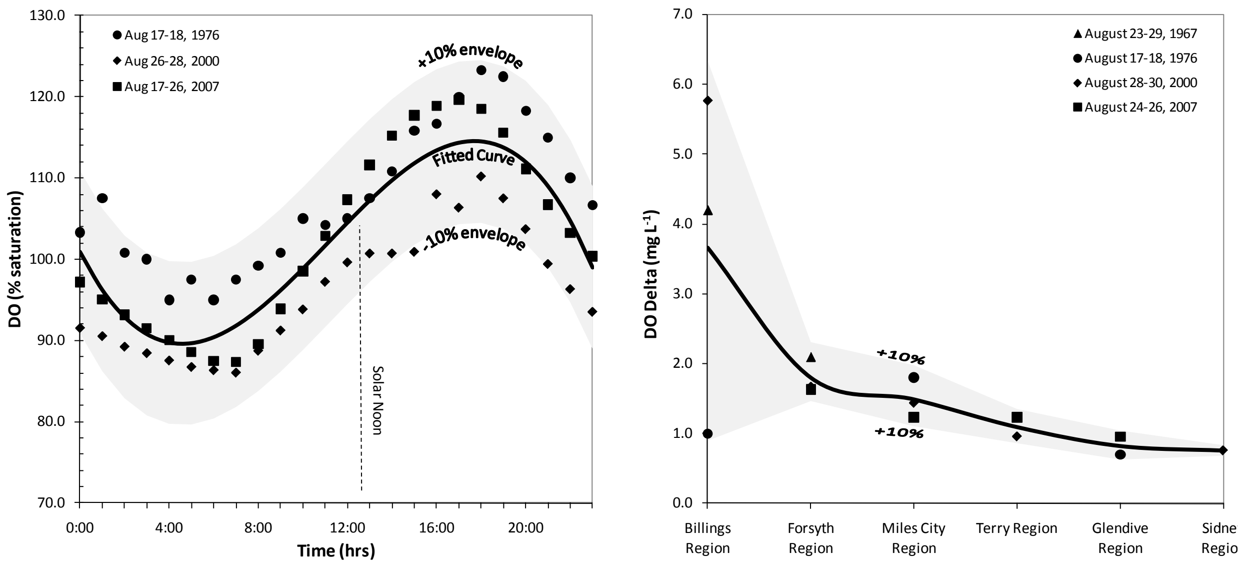

Figure 6-2. Dissolved oxygen data on the Yellowstone River for August 1967, 1976, 2000, and 2007. (Left panel) Typical diurnal pattern at Forsyth, MT over the August 1976, 2000, and 2007 period. A fitted curve is shown along with as ±10% saturation envelope. (Right panel) Longitudinal diurnal fluctuation 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).

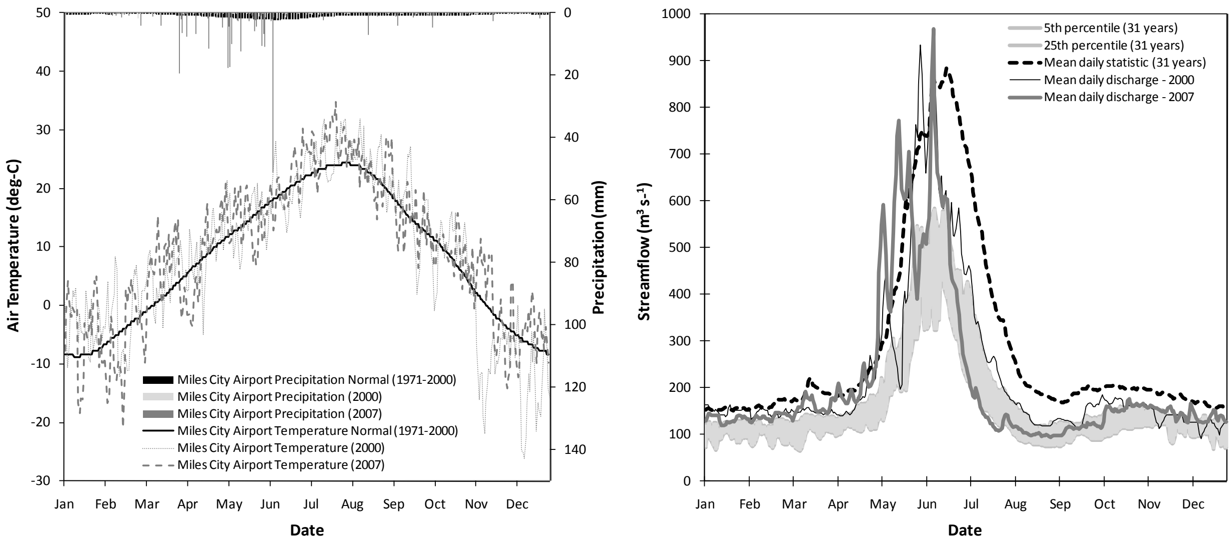

Figure 6-3. Conditions encountered during the 2000 and 2007 field data collection efforts.
(Left panel) Climatological data. (Right panel) Streamflow hydrology.

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**).

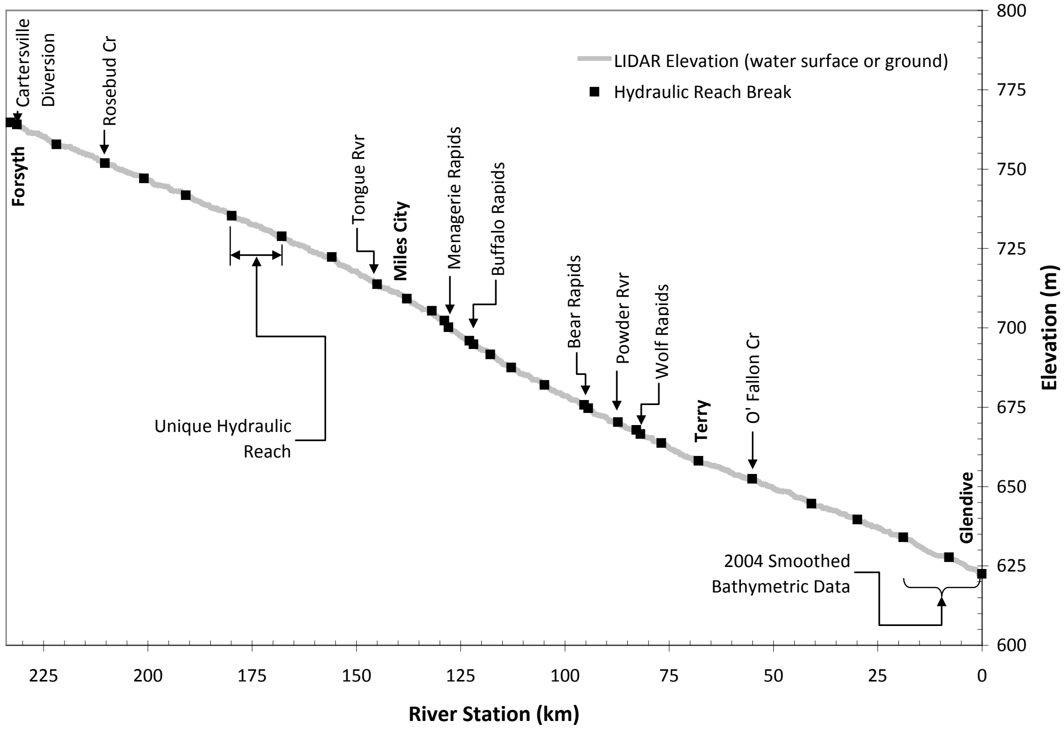

Figure 7-1. Longitudinal profile of the Yellowstone River.
Estimated from 2.5 meter DEM of the project site (see previous footnote for details on the DEM). Thirty-one hydraulic reaches were defined based on subtle changes in gradient. This included identification of several rapids in the project site.

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).

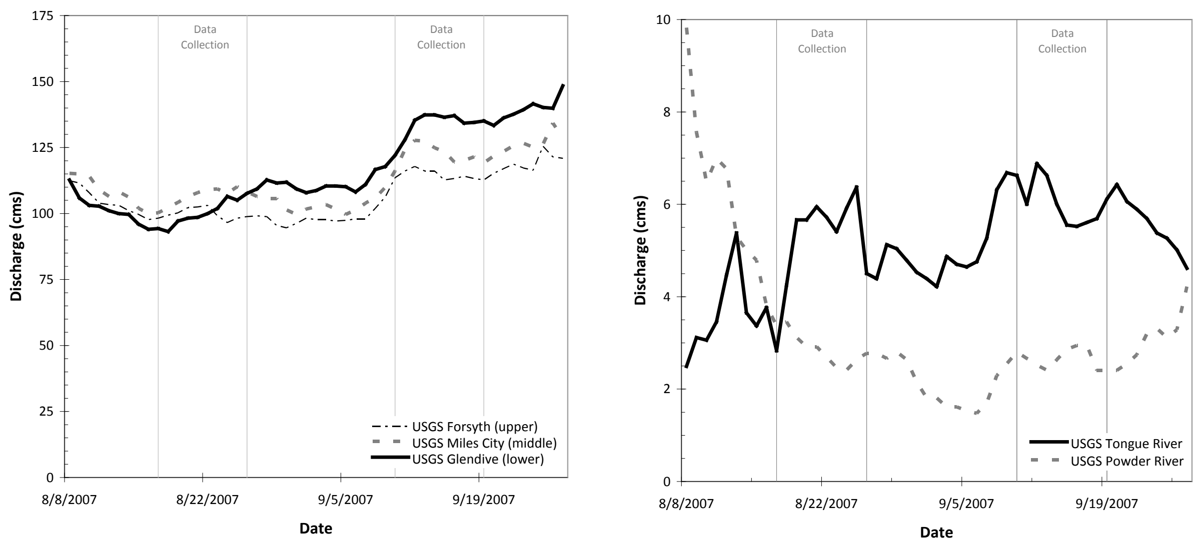

Figure 7-2. Surface water summary for the lower Yellowstone River during 2007.
(Left panel) Streamflow on the mainstem river for USGS 06295000 Yellowstone River at Forsyth (upper reach), USGS 06309000 Yellowstone River at Miles City (middle reach), and USGS 06327500 Yellowstone River at Glendive (lower 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

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[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.

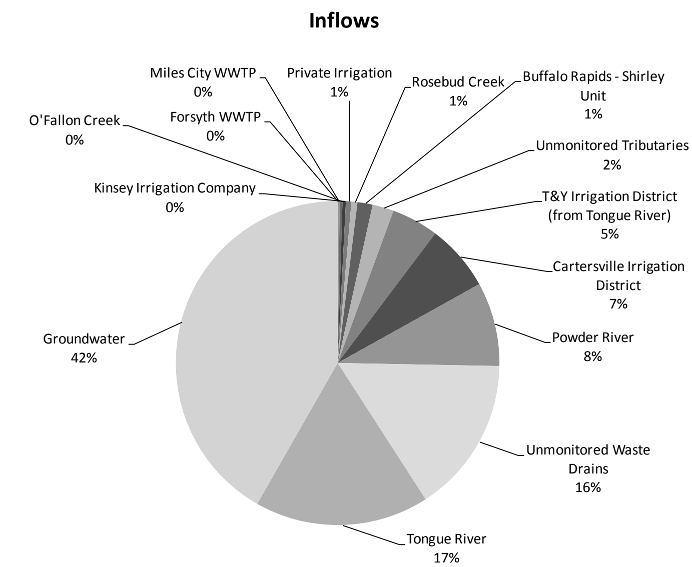
[21] A pan coefficient of 0.72 was used which compares reasonably with most work in the United States (Linsley et al., 1982). It does not compare that well with reported values for Fort Peck Reservoir (0.64 and 1.21 each month) (Army Corps of Engineers, 2003). Given the inability of DEQ to verify the 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).



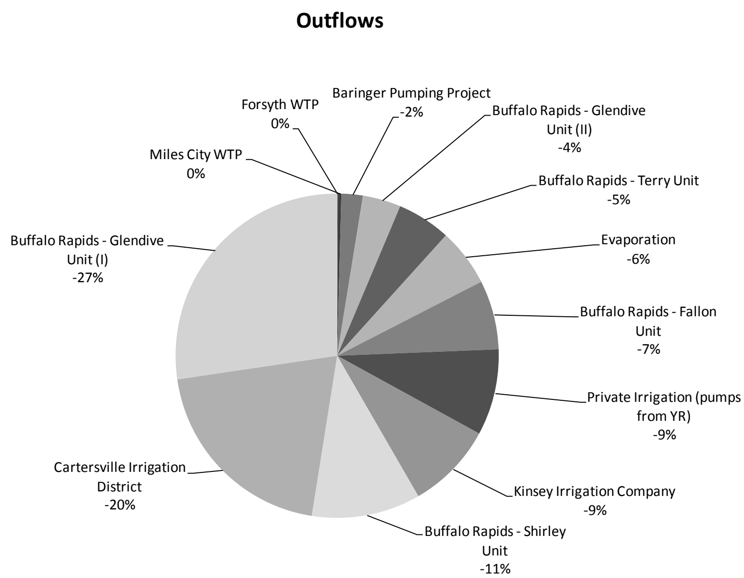


Figure 7-3. Graphical summary of water exchanges in the Yellowstone River during August 2007. (Top panel) Summary of inflows to the river during August of 2007. Note that the values shown are relative to one another; river flow at the upstream boundary (at Forsyth) accounted for 70% of the total inflow (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.

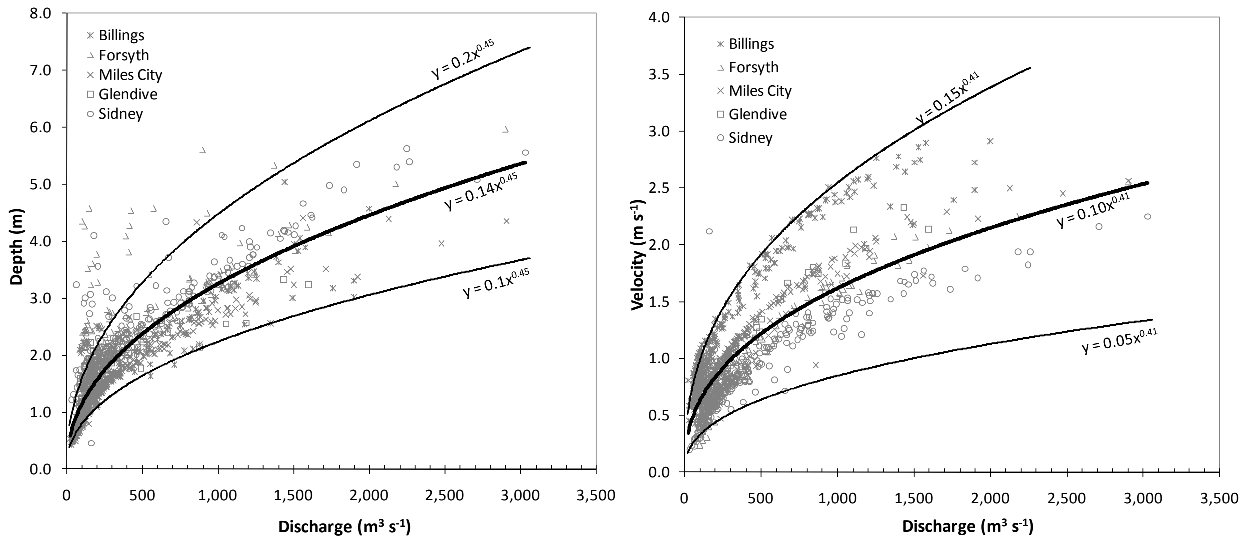


Figure 7-4. Depth and velocity rating curves derived for the lower Yellowstone River.
(Left panel). Depth vs. discharge. (Right panel) Velocity vs. discharge.

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.

[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.

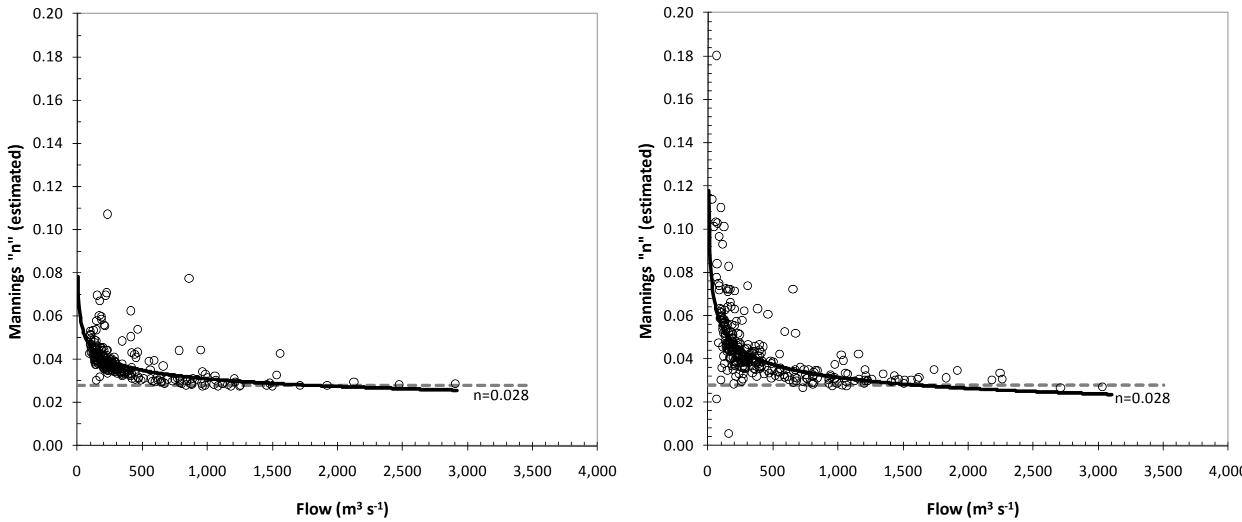


Figure 7-5. Estimated variation of Manning's n with flow for the Yellowstone River.
(Left panel) USGS 06309000 Yellowstone River at Miles City; (Right panel) 06329500 Yellowstone River near Sidney.

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.

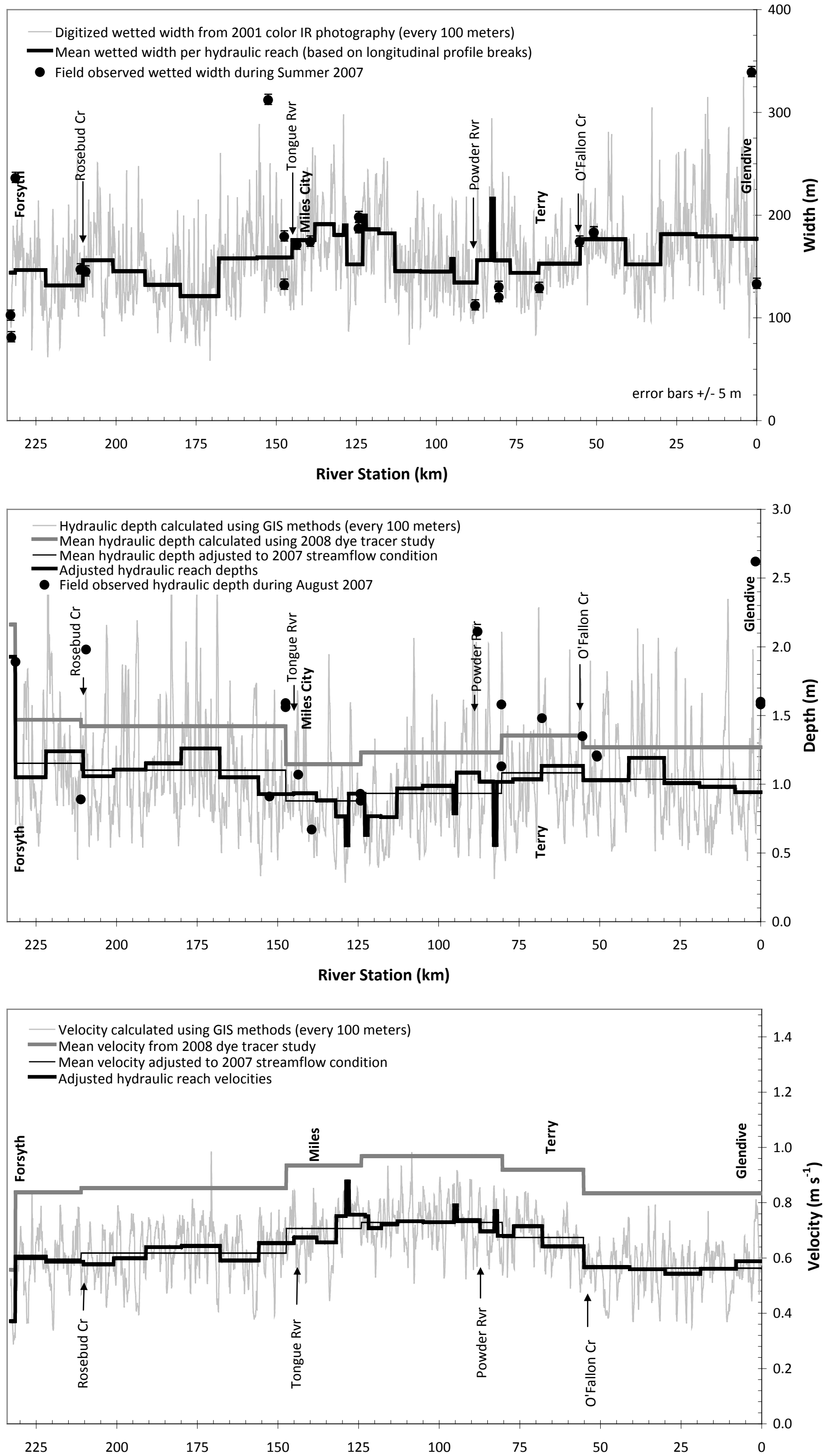


Figure 7-6. Estimated width, depth, and velocity over 1 km increments in the Yellowstone River. Data shown for the August 2007 flow condition.

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.

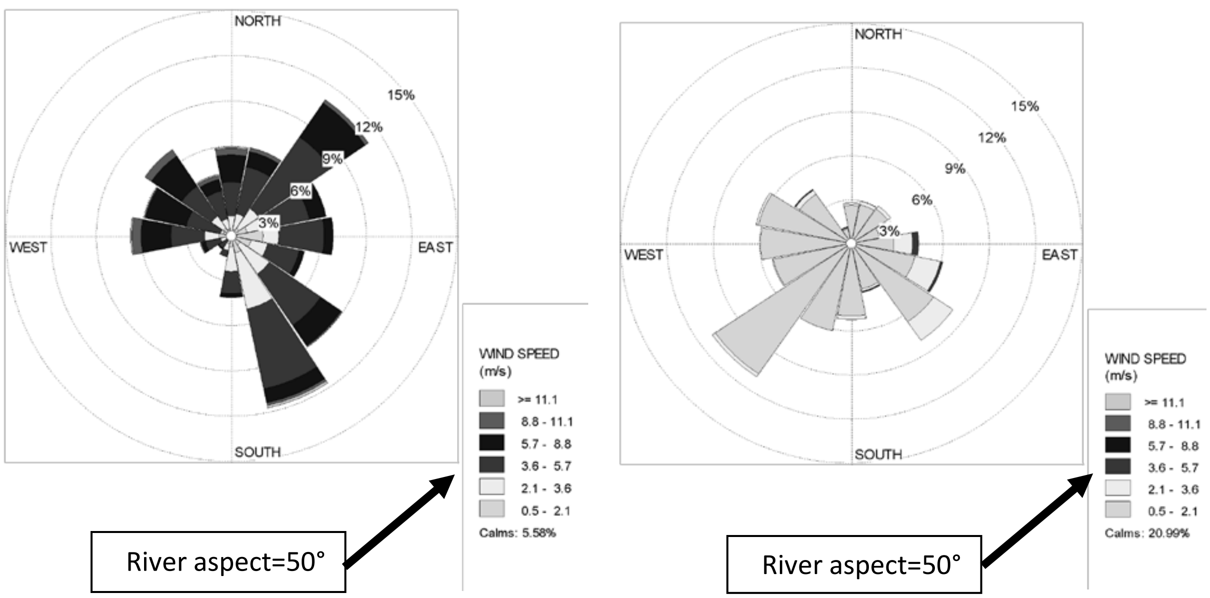


Figure 7-7. Paired wind rose data for the lower Yellowstone River during 2007.
(Left panel) Wind magnitude and direction at the Miles City Municipal Airport. (Right panel) Same, but for the DEQ station on Roche Juan Island. The reversal in direction and decline in magnitude from turbulence was used to 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.

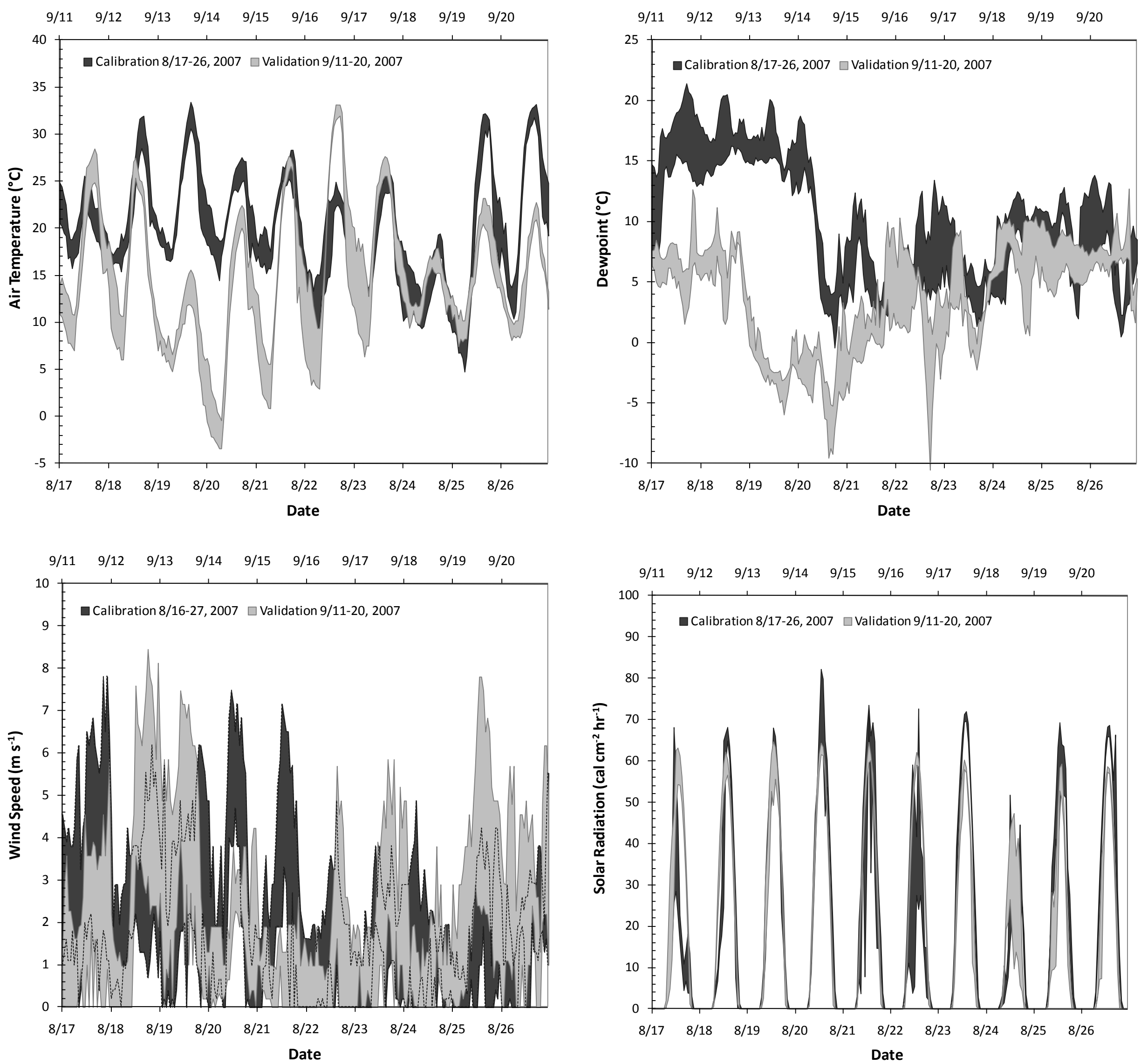


Figure 7-8. Hourly meteorological data summary for lower Yellowstone River in 2007.
(Top left/right panel) Air temperature and dew point for the four climatic zones in the lower Yellowstone River during the August 17-26 (calibration) and September 11-20, 2007 (validation) period (the shaded ribbon represents 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).

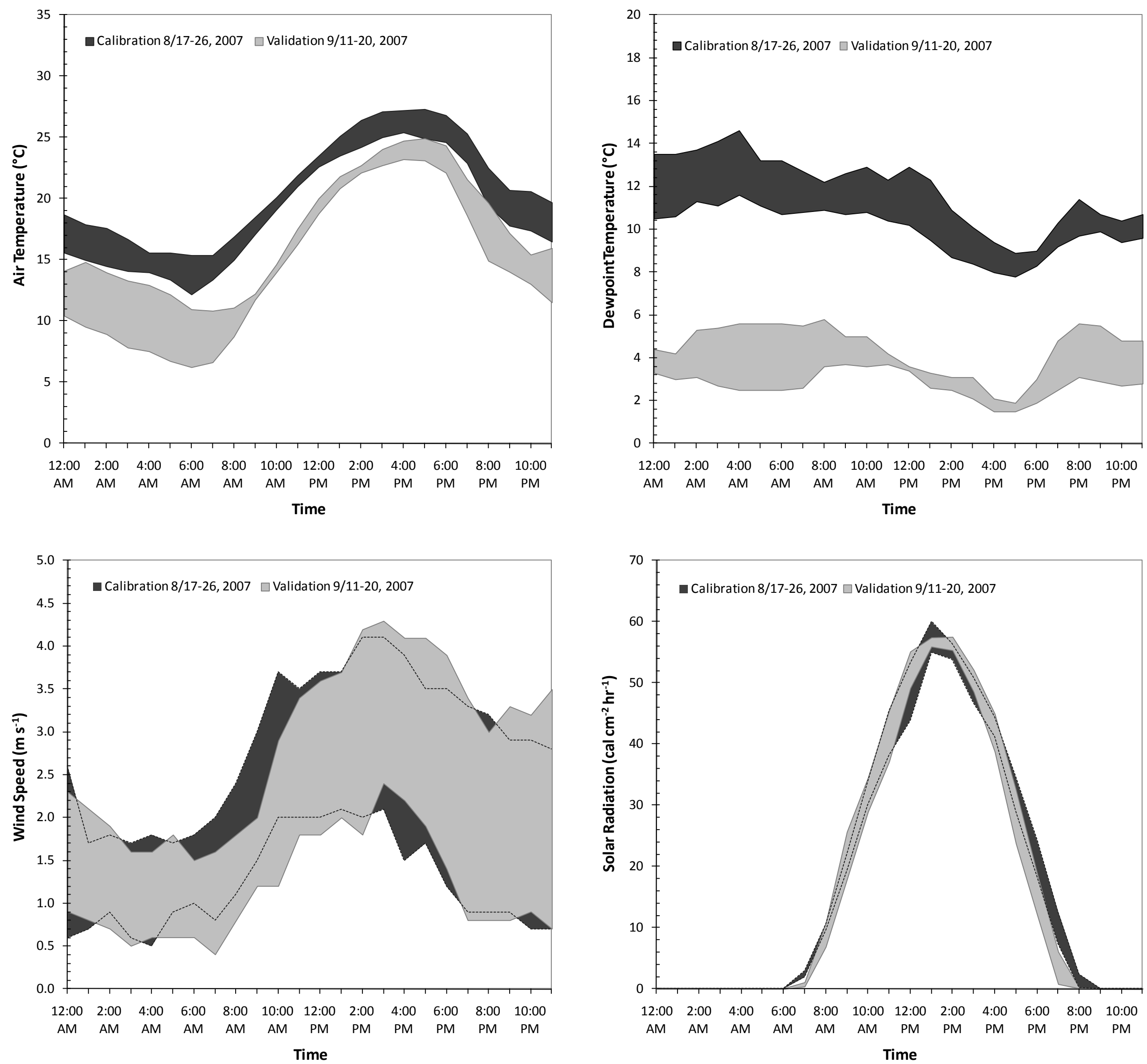


Figure 7-9. Mean repeating day climatic inputs for lower Yellowstone River Q2K model.
(Top left/right panel) Air temperature and dew point for the four climatic zones for the August 17-26 (calibration) and September 11-20, 2007 (validation) period. (Bottom left/right panel) Same but for wind speed and 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

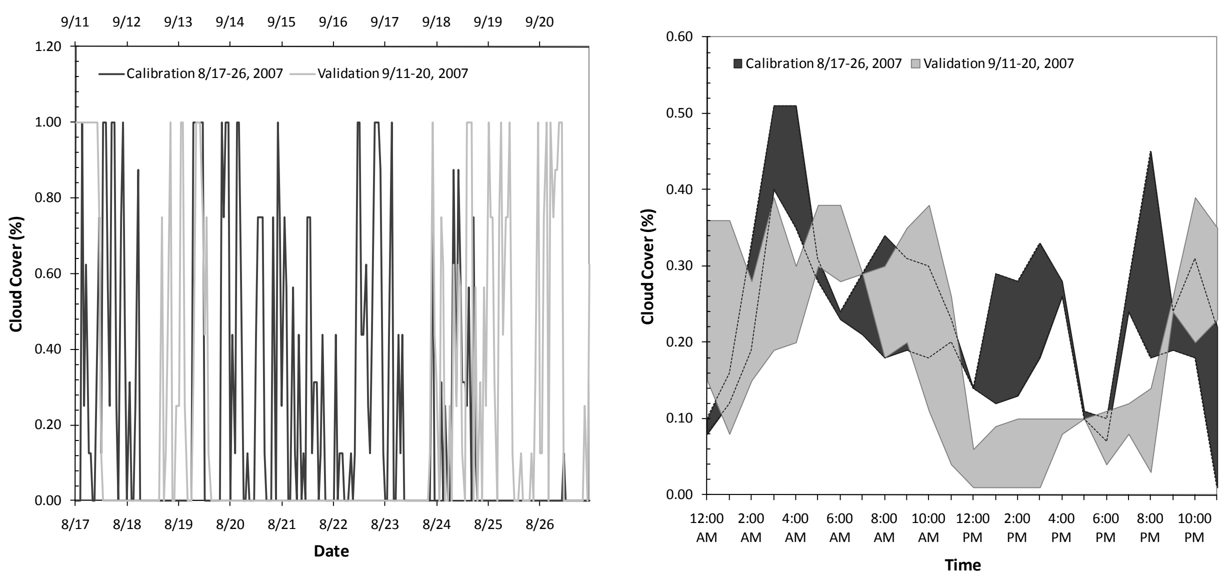


Figure 7-10. Hourly and repeating day cloud cover data for the lower Yellowstone River.
(Left panel) Cloud cover data for the 2007 period. (Right panel) Same but in a mean repeating day format. The range of the climatic stations used in the modeling reflects the shaded ribbon.

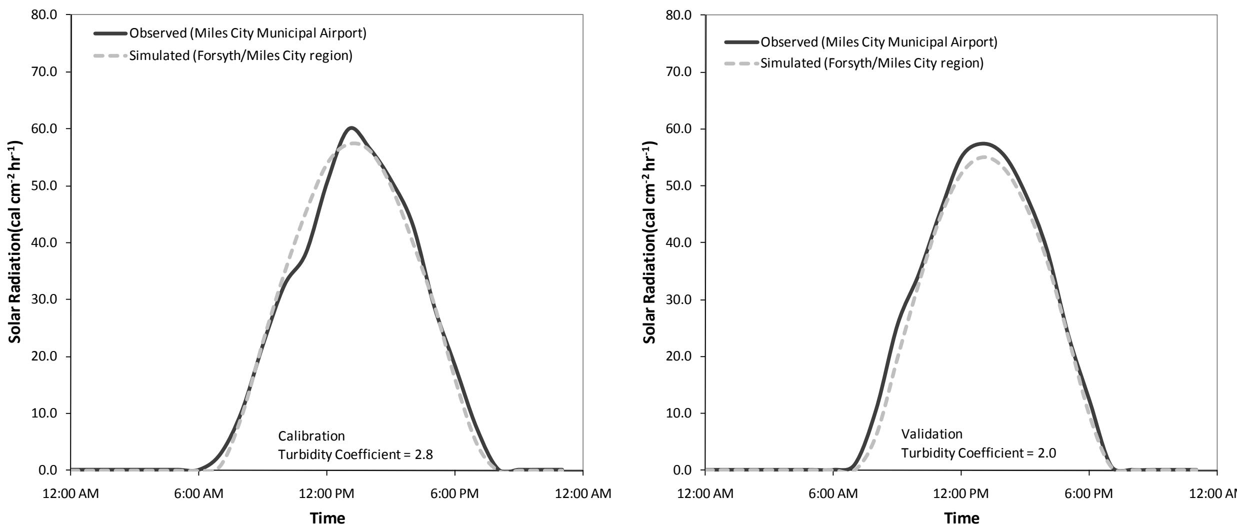


Figure 7-11. Simulated and observed solar radiation for the lower Yellowstone River.
(Left panel) Simulated and observed solar radiation for August 17-26, 2007 at Miles City Airport. (Right panel) Same but for September 11-20, 2007.

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).

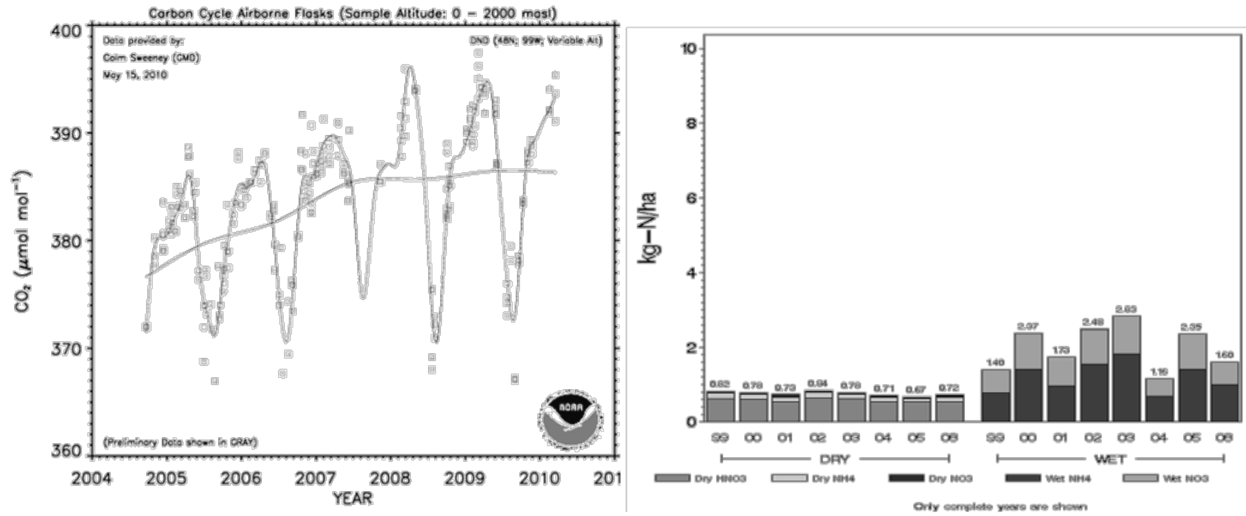


Figure 7-12. CO_2 data and nitrogen dry deposition by species for the Yellowstone River.
(Left panel) CO_2 data from the Dahlen, ND GLOBALVIEW-CO2 monitoring site (2004-2010). (Right panel) Nitrogen deposition data from Theodore Roosevelt, National Park EPA CASTNET site (1999-2008). Both figures taken 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.

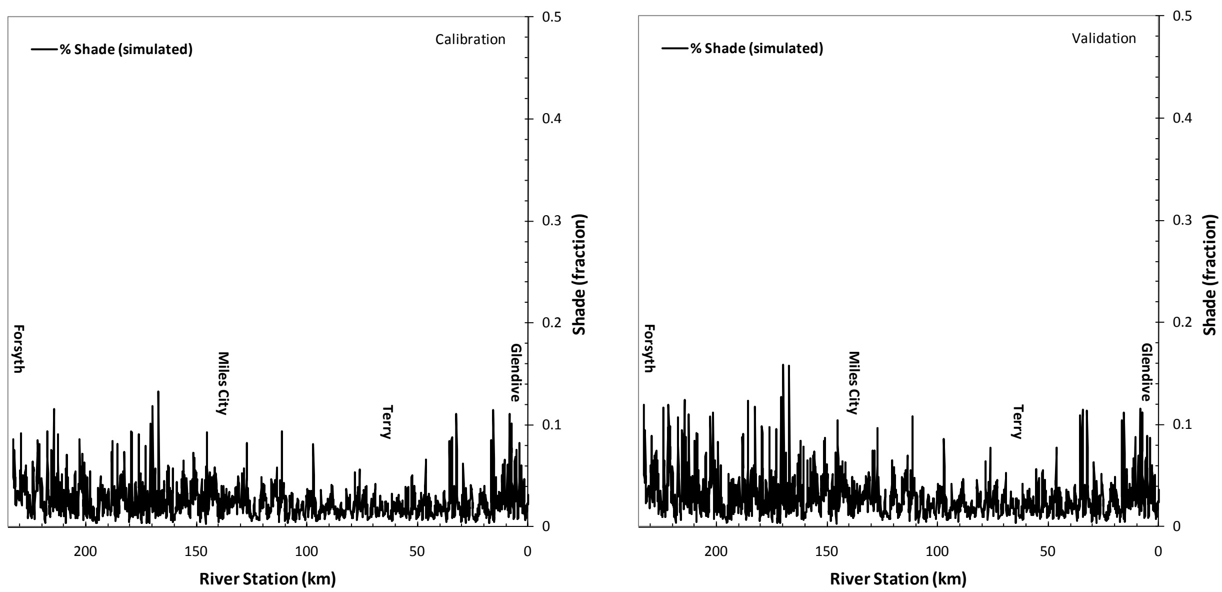


Figure 7-13. Simulated mean daily shade for the Yellowstone River.
(Left panel) Simulated shade for the August 17-26, 2007 period. (Right panel) Same but for September 11-20. No field data were available to verify the simulations. In both cases, shade is a minor component as indicated by mean daily shading 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.

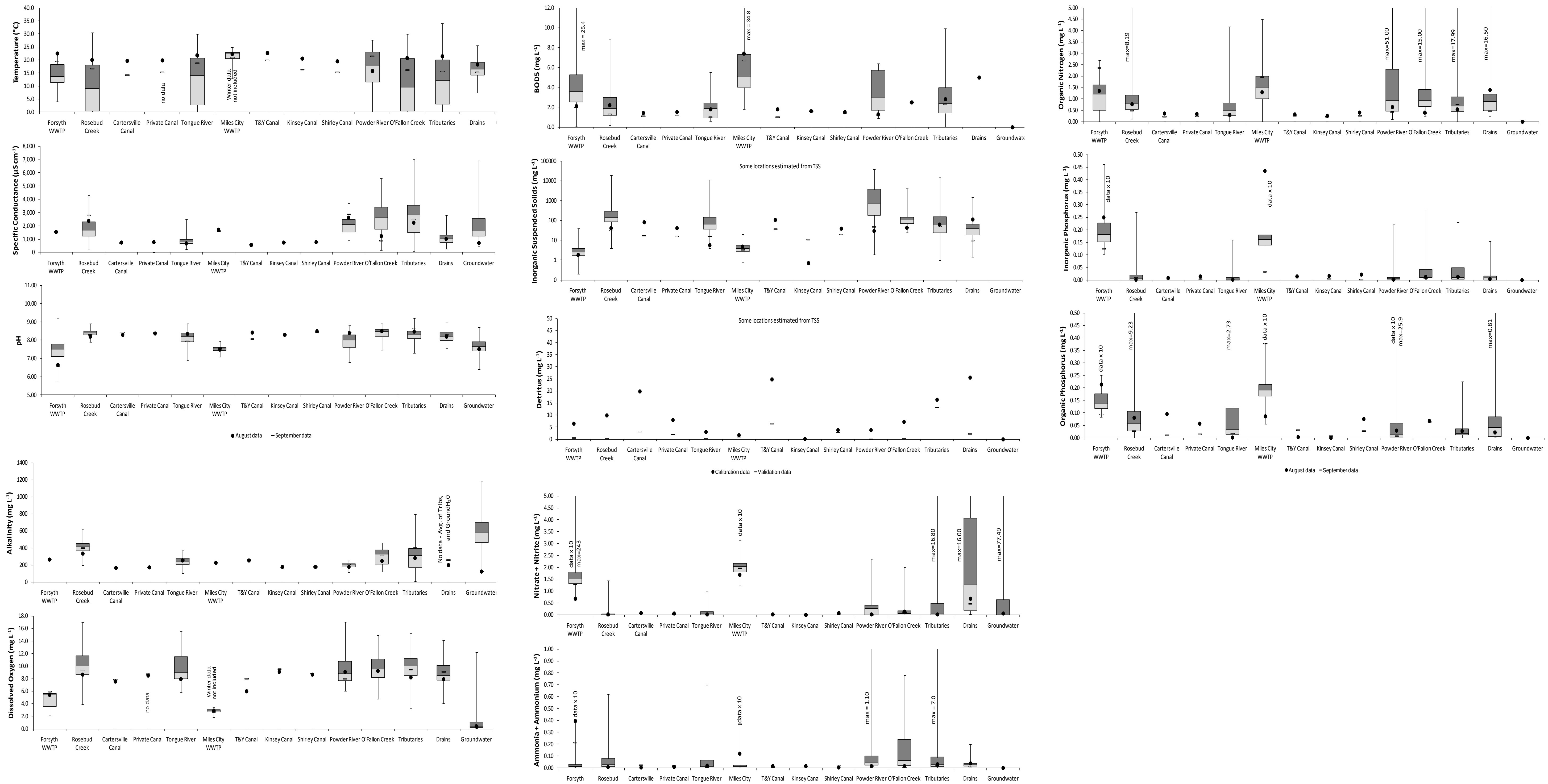


Figure 7-14. Comparative water quality inflow plots for the Yellowstone River.

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.

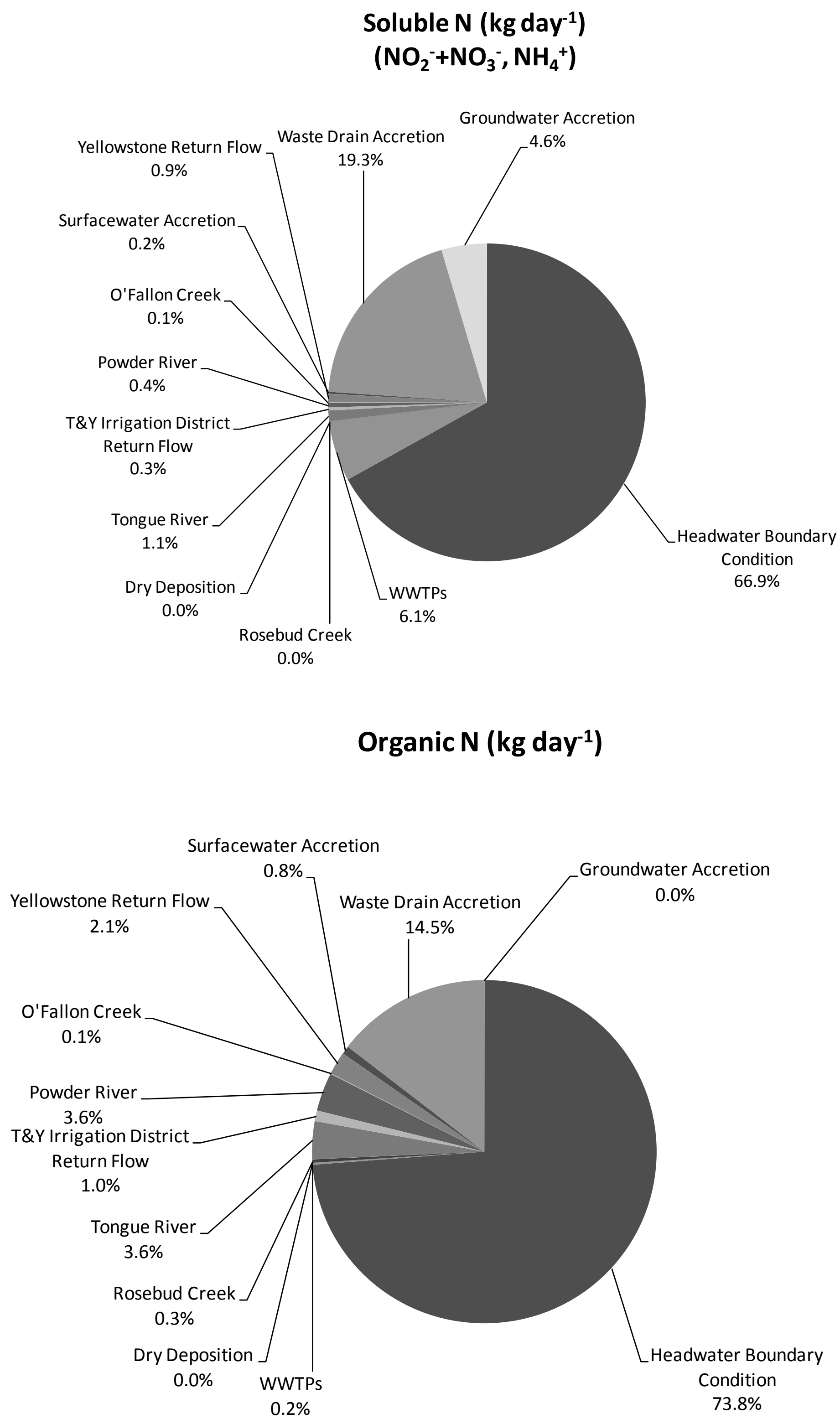
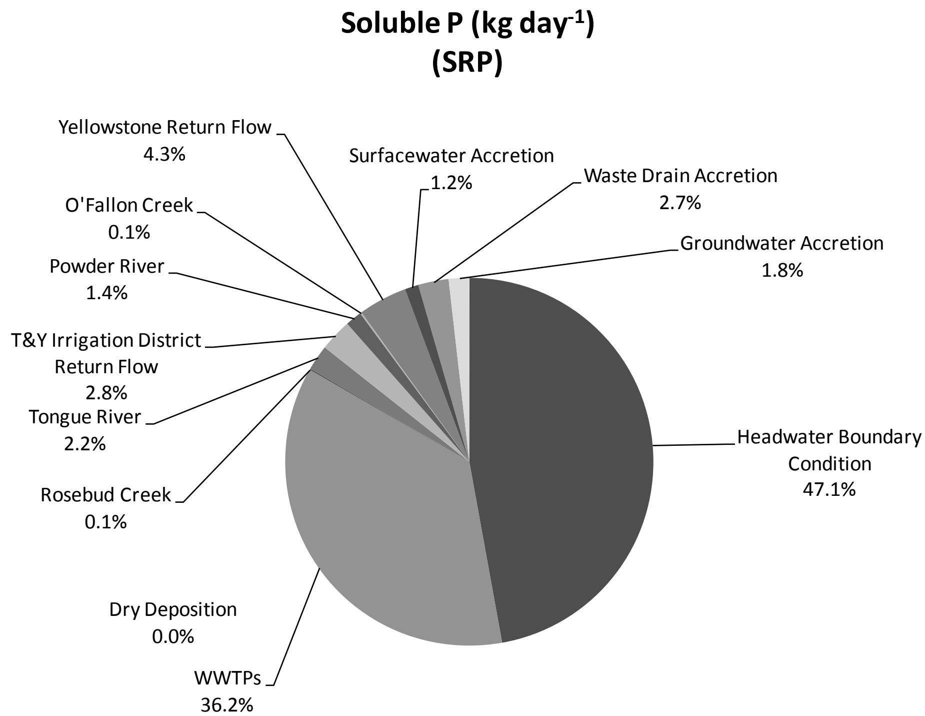


Figure 7-15. Estimated nitrogen contributions to the lower Yellowstone River during 2007.



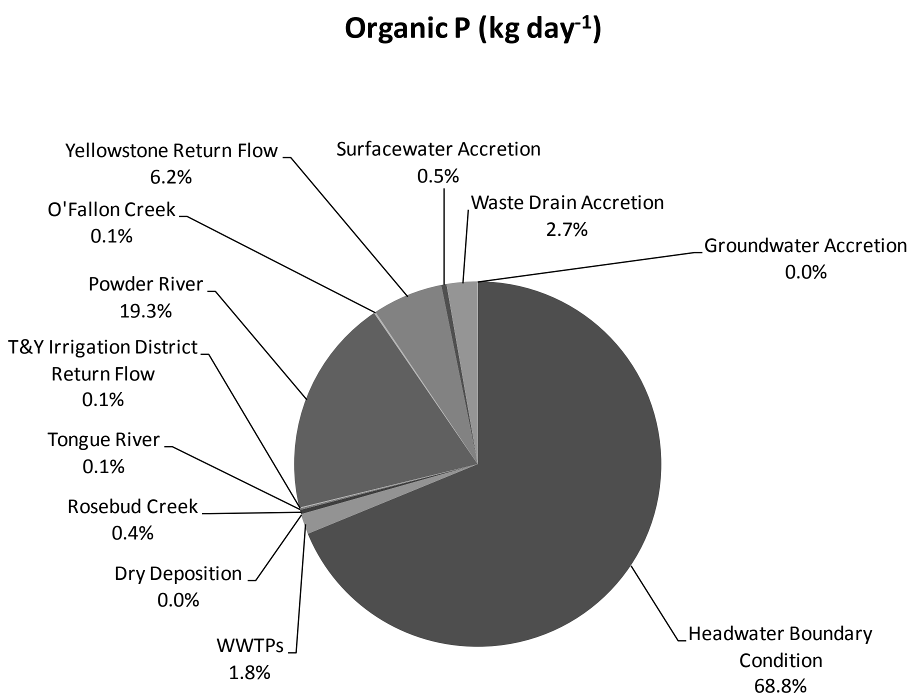


Figure 7-16. Estimated phosphorus contributions to the lower Yellowstone River during 2007.

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.

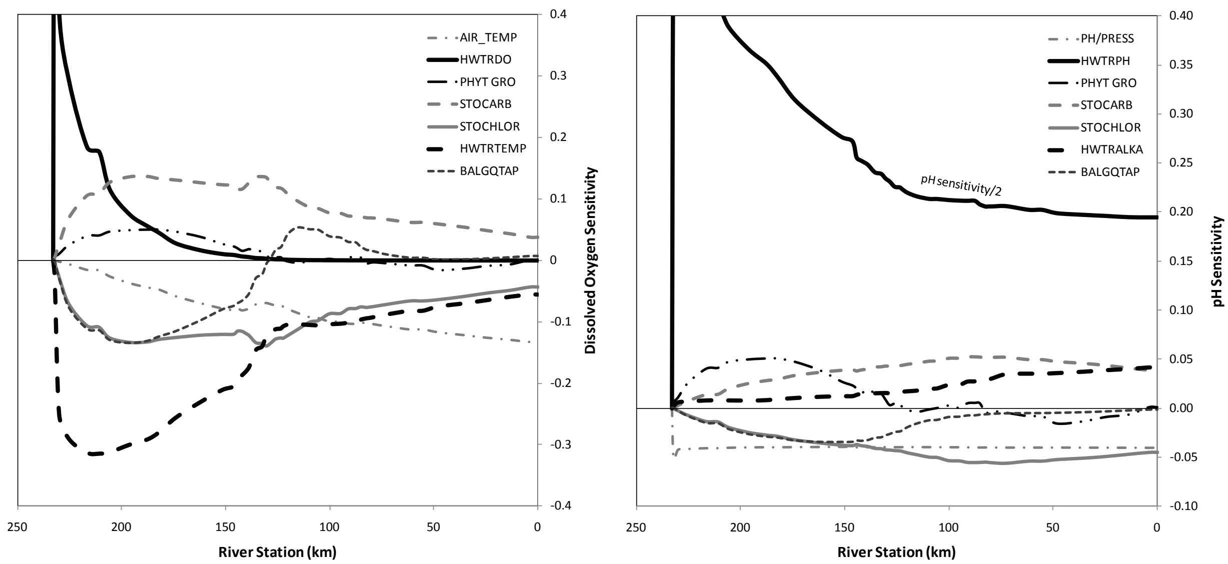


Figure 8-1. Longitudinal sensitivities of selected model rates and forcings for DO and pH.
(Left) Model sensitivities in relation to DO. (Right) Same but for pH.

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.

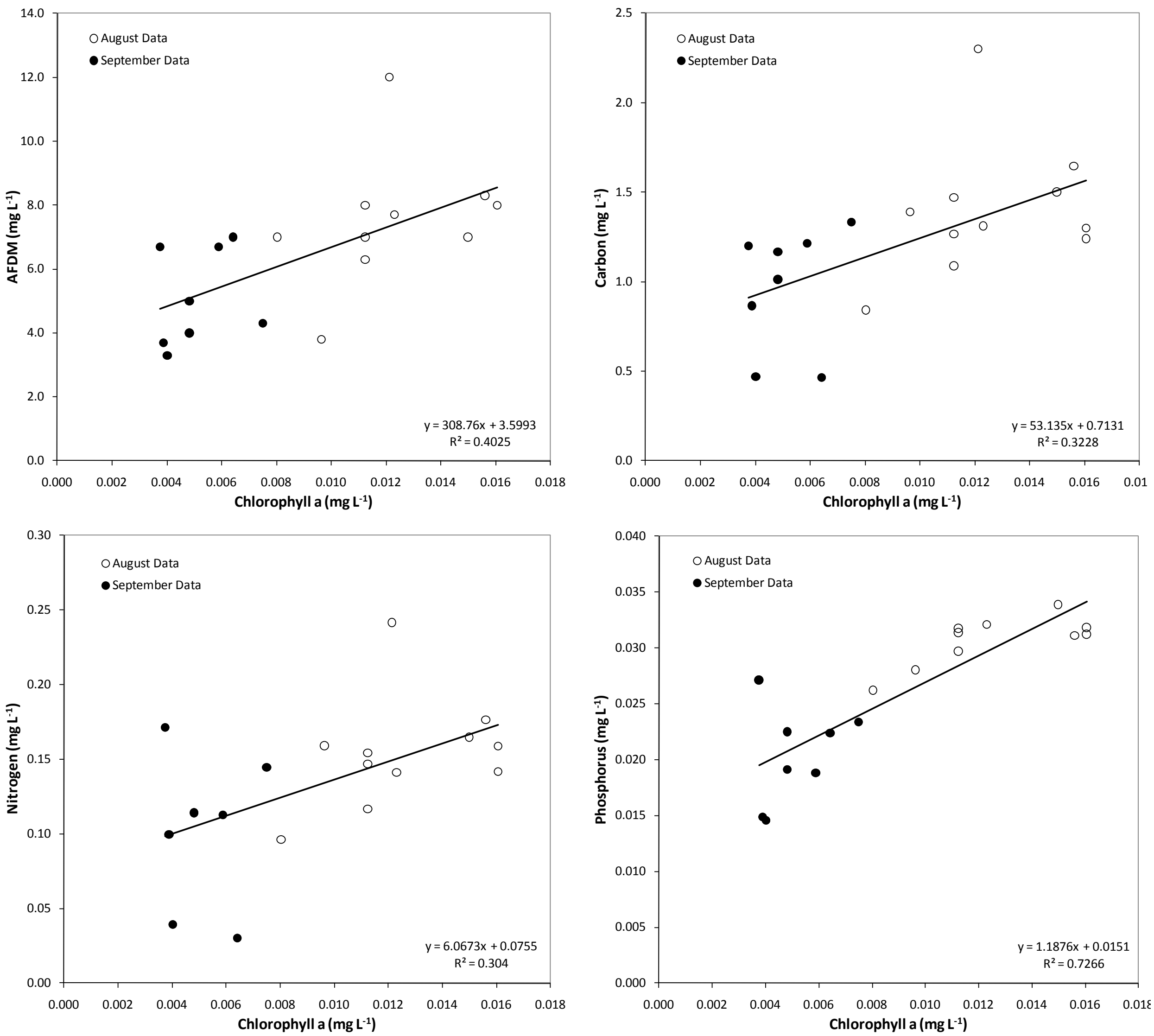


Figure 8-2. Stoichiometric C:N:P regression relationships for the Yellowstone River.
Note: One September data point was adjusted and one was removed due to inconsistent results. This was done as the carbon and AFDM values for this sample were anomalous compared to all of the other observations for the 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}.

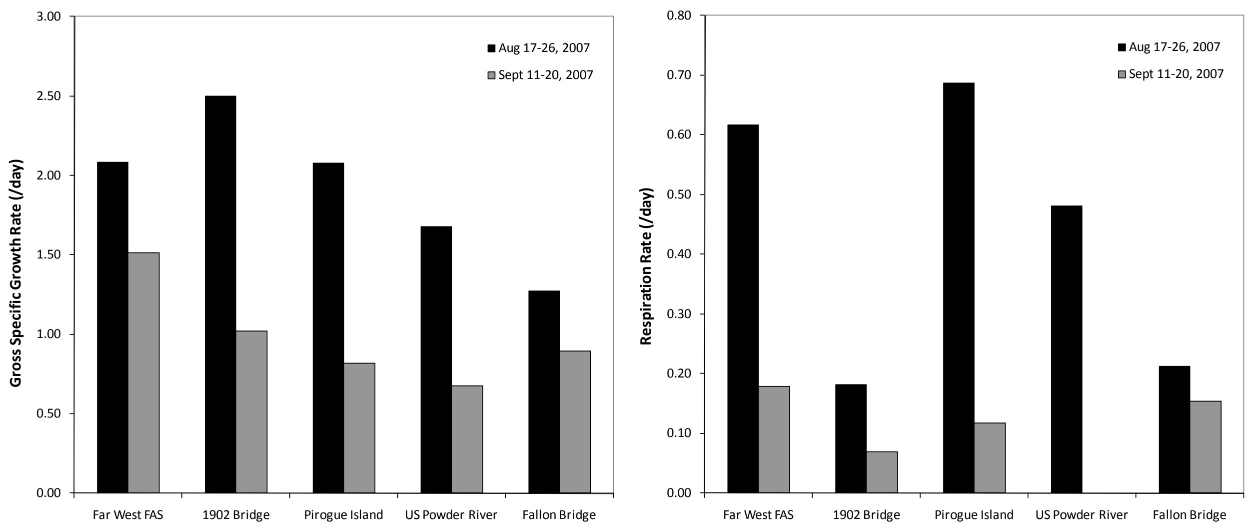


Figure 8-3. Primary productivity and respiration measurements on the Yellowstone River in 2007. There is a strong downstream decrease in productivity which is linked directly with river turbidity (and perhaps soluble nutrients to a lesser extent given the fact that nitrogen happened to be limiting phytoplankton). 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.

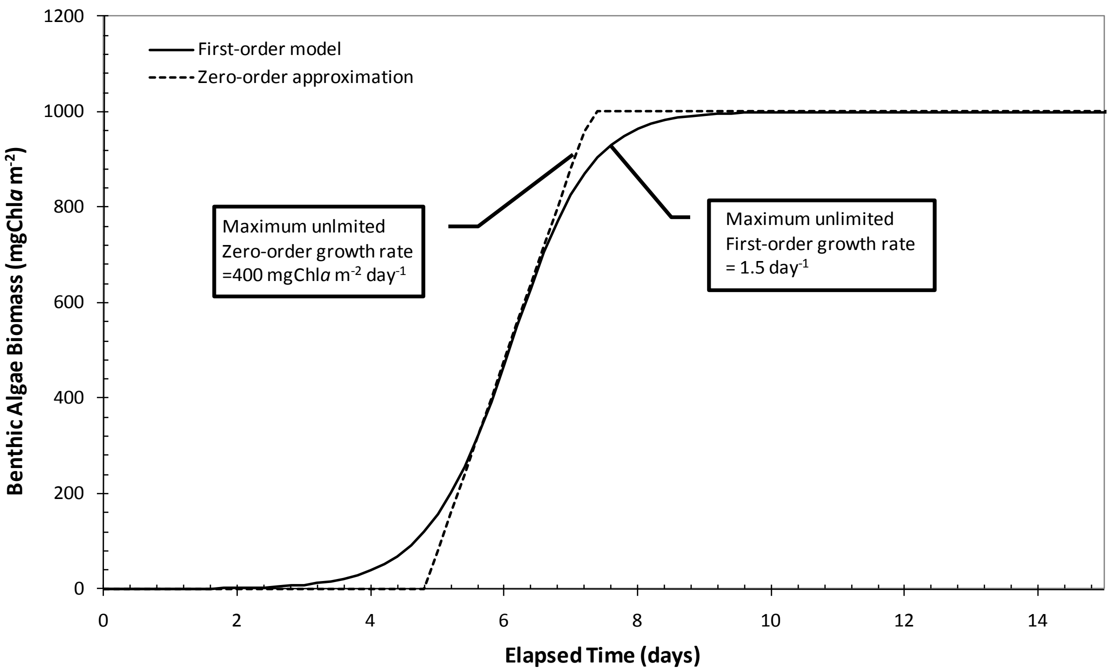


Figure 8-4. Comparison between zero- and first-order maximum unlimited growth estimates.
First-order growth modeled with an initial condition of 0.1 mg Chl*a* m^{-2} (**Equation 12-1**). The slowing of biomass accumulation over elapsed time is represented as a logistic function for implied space limitation (**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).

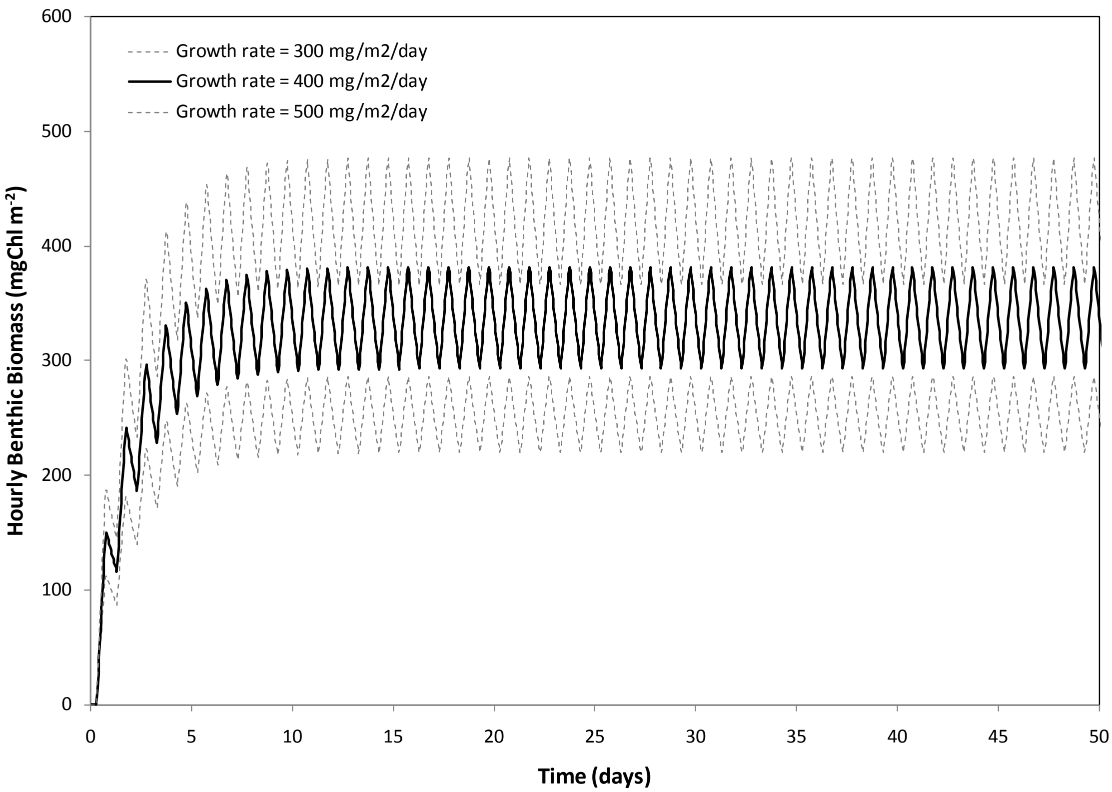


Figure 8-5. Estimated maximum biomasses with losses but no nutrient or light limitation. Simulations show that a growth rate of approximately 400 mg Chl*a* m^{-2} day^{-1} is required to meet expected peak diatom biomass under unlimited growth conditions. The daily oscillations reflect the disparity 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 $k_{a20}\pm$	September k_{a20} (day^{-1})	95% CI $k_{a20}\pm$
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 Duplicate	0.66 0.35	0.51	43	0.2[1] 0.1[2]	1.0 10
Fallon Bridge Duplicate	0.78 0.57	0.67	22		
Richland Park Duplicate	0.43 0.06	0.24	105		

[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] $\ln(PAR_z) = -k_e z + \ln(PAR_{surface})$

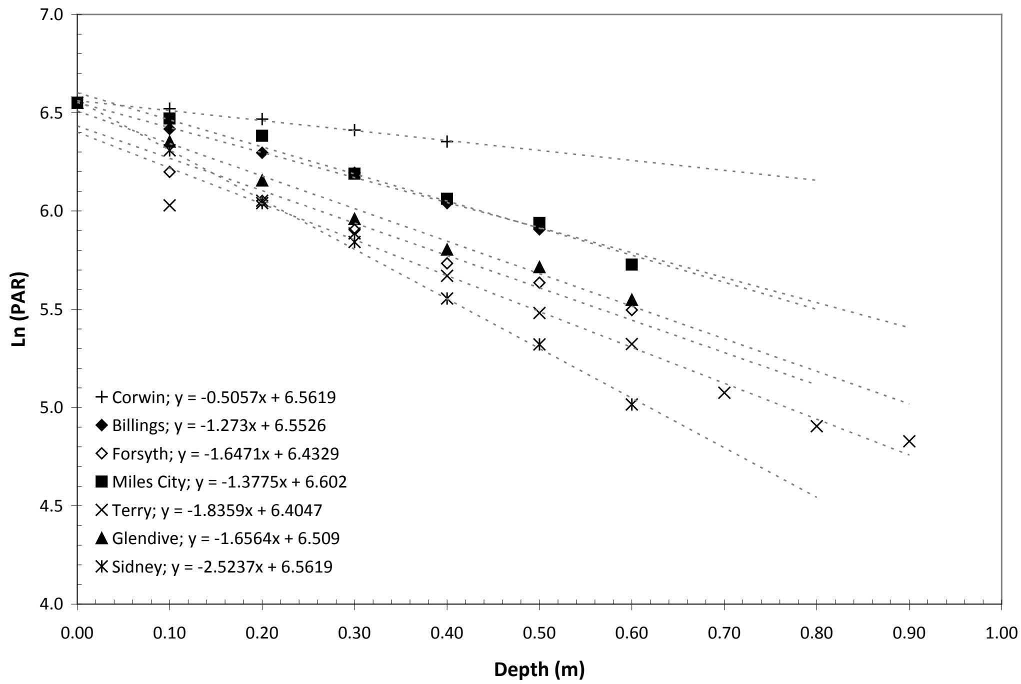


Figure 8-6. Light extinction coefficients calculated for the Yellowstone River.
Extinction values range from 1.3-2.5 m^{-1} in the lower river (Billings to Sidney). Measurements taken in late August or early September by Peterson (2009).

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.

[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).

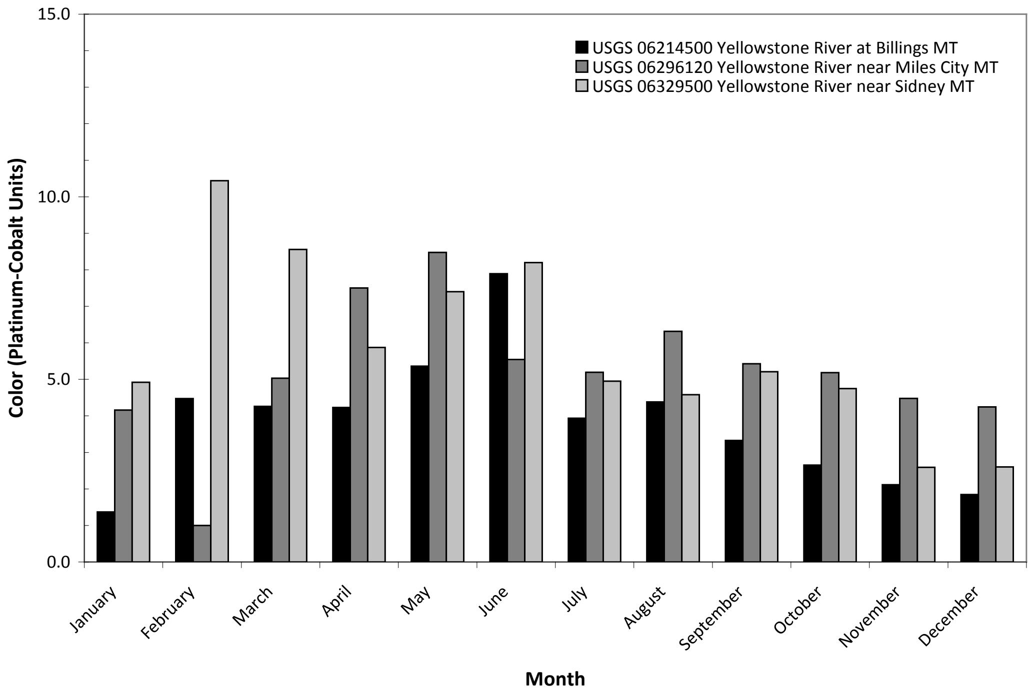


Figure 8-7. Monthly mean true color measurements for the lower Yellowstone River (1963-1970). All months had at least 2+ observations. Observations vary from year to year, however, during low-flow conditions remain relatively consistent. Billings gage shown for reference only.

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).

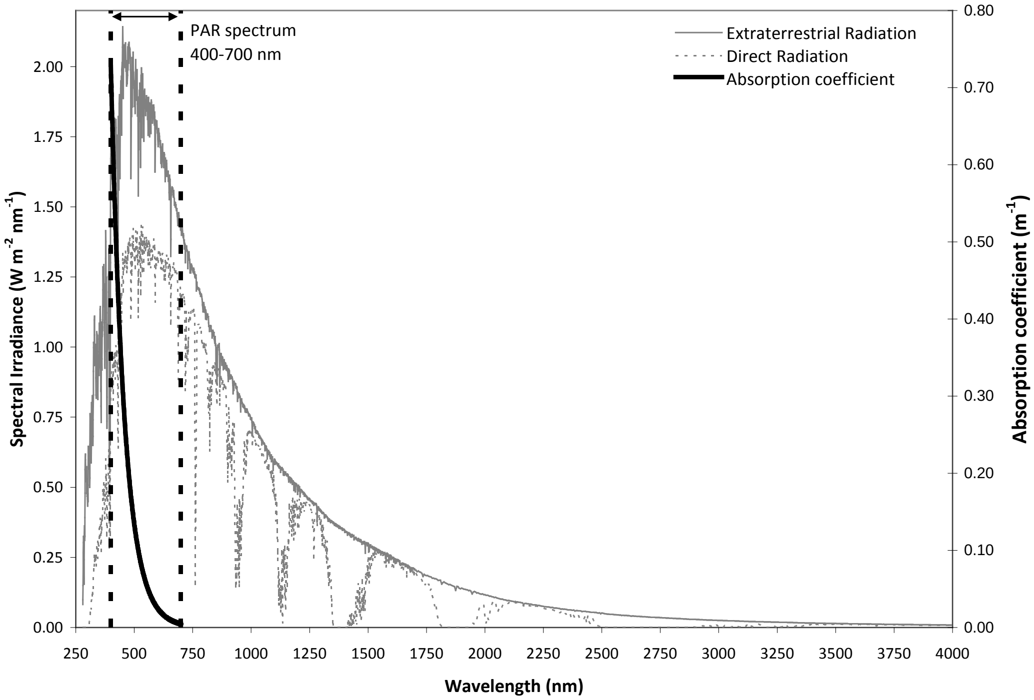


Figure 8-8. ASTM reference spectra used to evaluate the net absorption coefficient for color.

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).

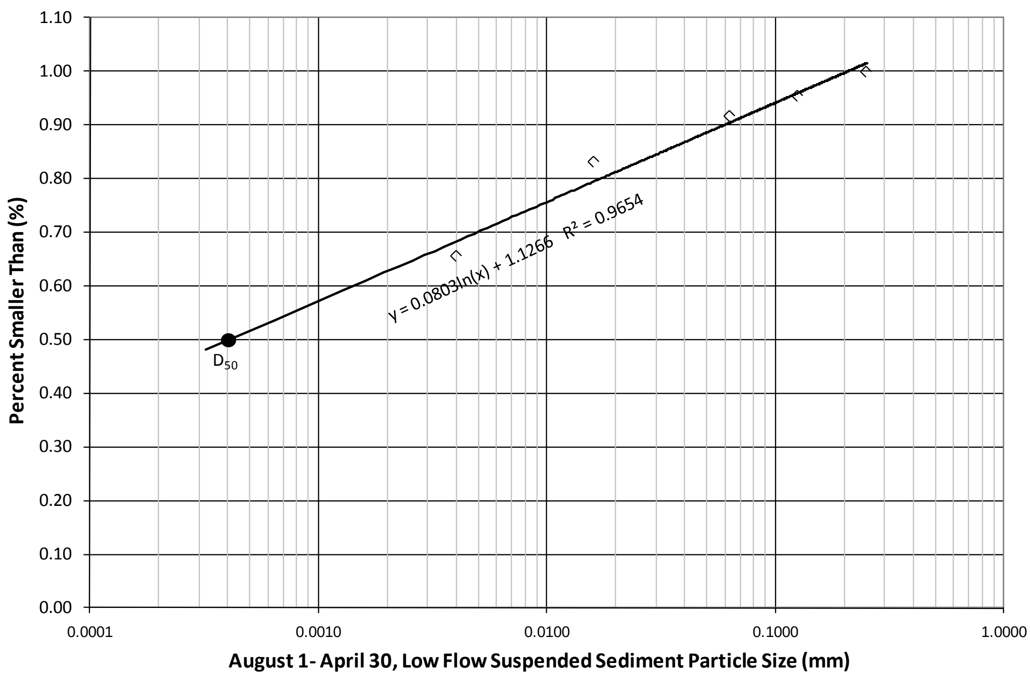


Figure 8-9. Suspended sediment particle size (mm) in Yellowstone River during low-flow conditions. Data taken as average of fall diameter measurements at USGS 06295000 Yellowstone River at Forsyth MT and USGS 06329500 Yellowstone River near Sidney MT.

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).

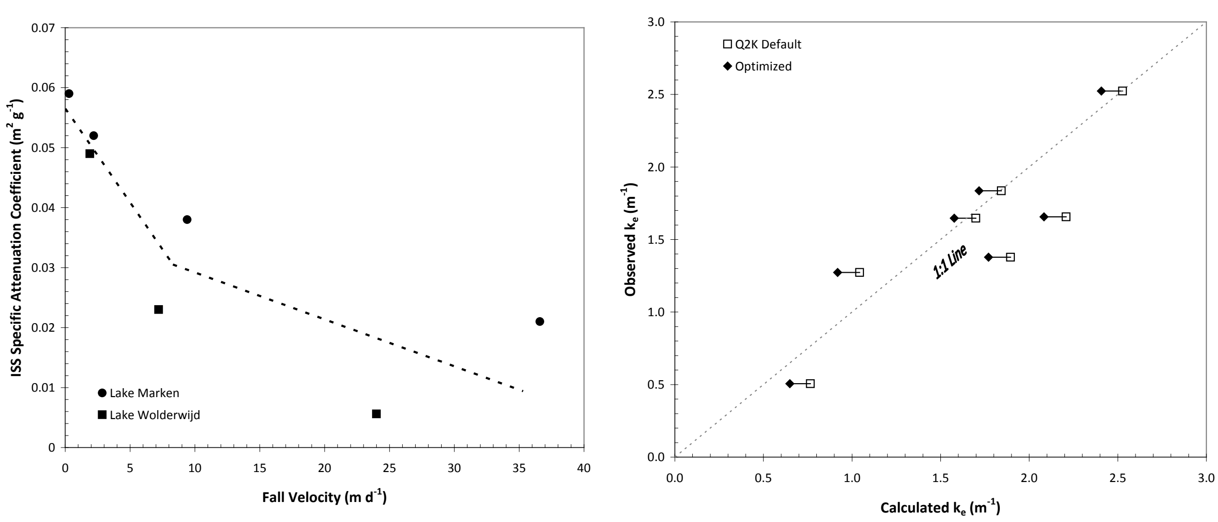


Figure 8-10. Optimized partial extinction coefficients for the remaining particles (αi, αo, αp & αpn). (Left) Relationship between fall velocity and inorganic suspended solids partial attenuation coefficient ($m^2\ g^{-1}$) based on data from Blom et al., (1994). (Right) Optimization of remaining partial 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].

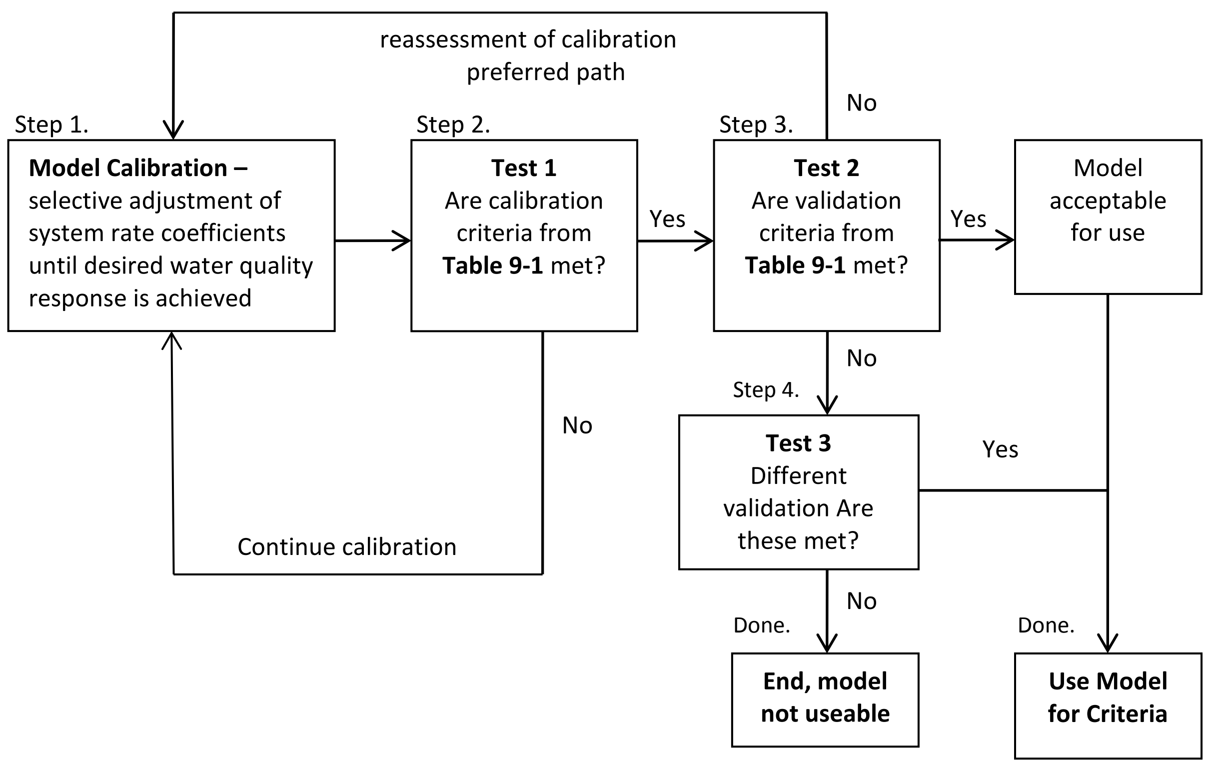

Figure 9-1. Model calibration and validation approach for the Yellowstone River.
Calibration was completed iteratively until acceptability criteria in **Table 9-1** were achieved. Validation or confirmation tests were then performed to identify whether the model was acceptable for use.

[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.

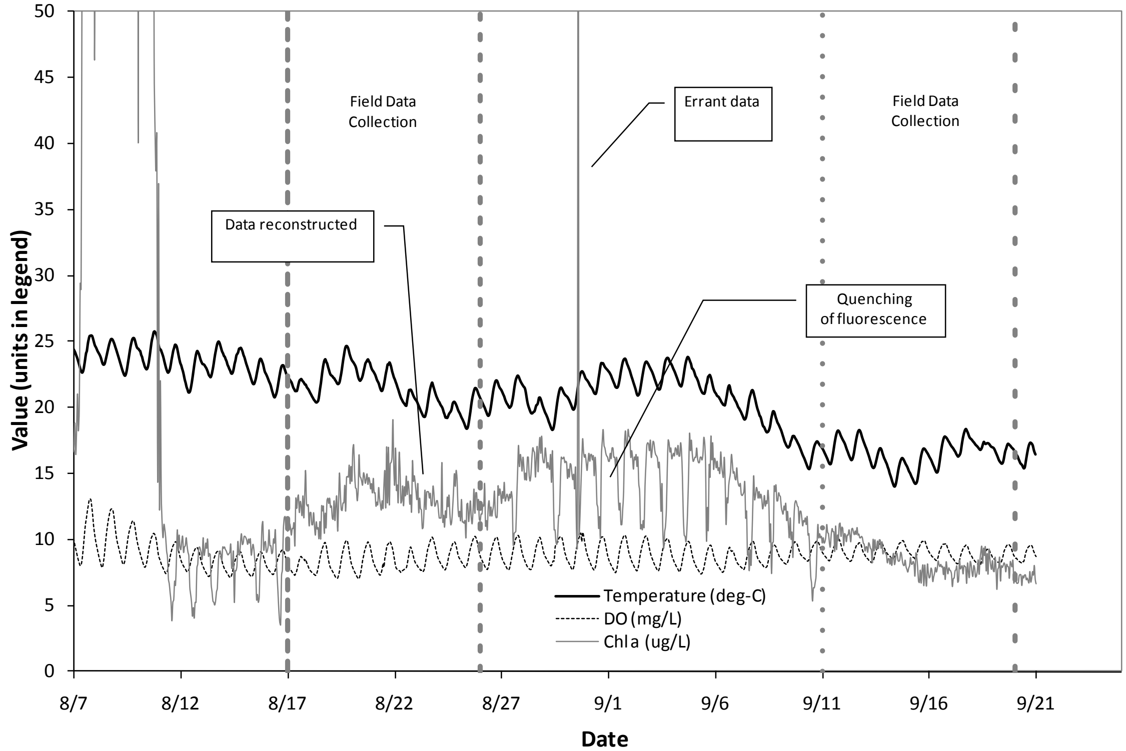

Figure 9-2. Example of YSI sonde data from the Yellowstone River in 2007.
Temperature, DO, and Chl*a* shown for YSI-20, upstream of Cartersville canal.

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.

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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**.

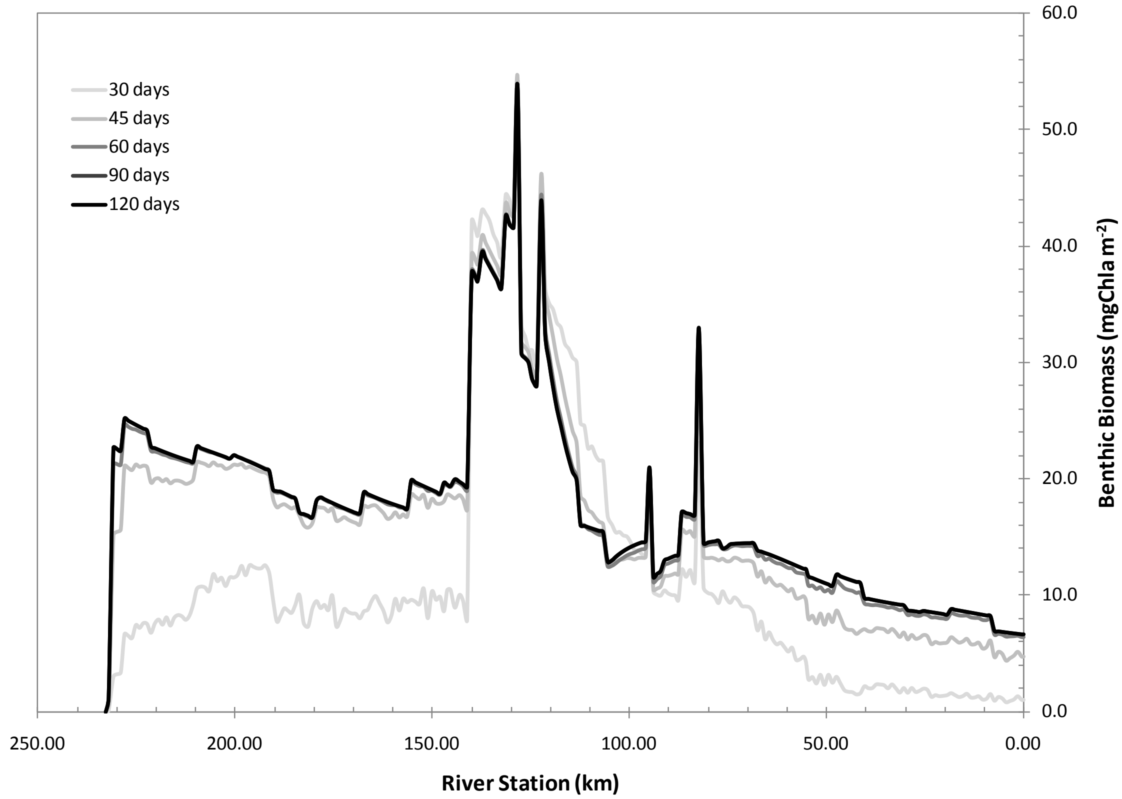

Figure 9-3. Evaluating model runtime requirements for the Yellowstone River.
Existing condition model run reflecting time required to reach steady state algal biomass. A run of approximately 90 days is required to reach convergence.

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	(µg A L^{-1}) m^{-1} *	0.031	0.009	0.031	0.031[1]
Nonlinear Chl*a* extinction	α_{pn}	(µg 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].

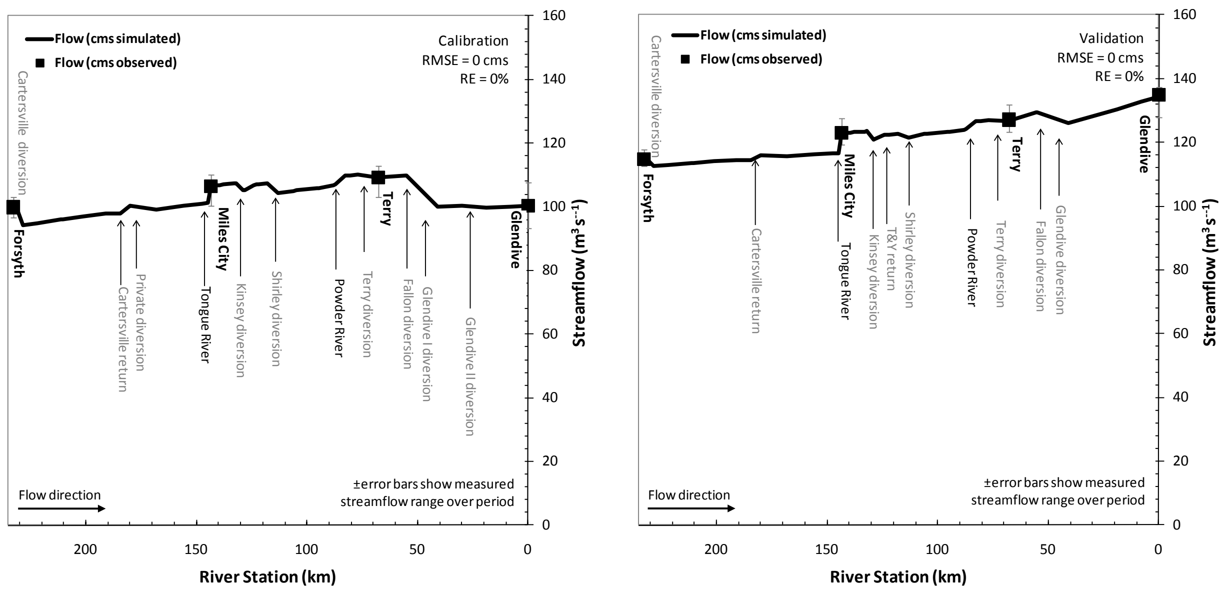

Figure 10-1. Simulated and observed streamflow for the Yellowstone River during 2007. (Left panel) August calibration. (Right panel) September validation. Flows during August remained relatively constant due to irrigation depletion whereas in September the streamflow profile shows a longitudinal increase 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.

[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.

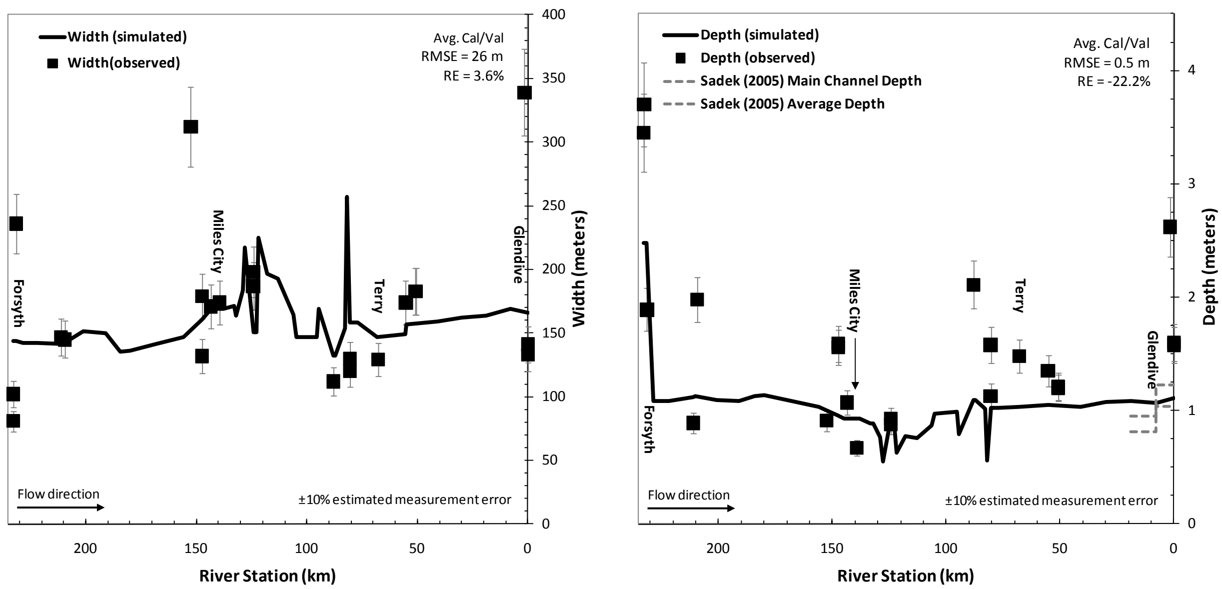

Figure 10-2. Simulated and observed river widths and depths for the Yellowstone River during 2007. (Left panel) Simulated river width with associated error statistics. (Right panel) Same but for river depth. Comparisons shown for the average of the calibration and validation periods as very little change 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).

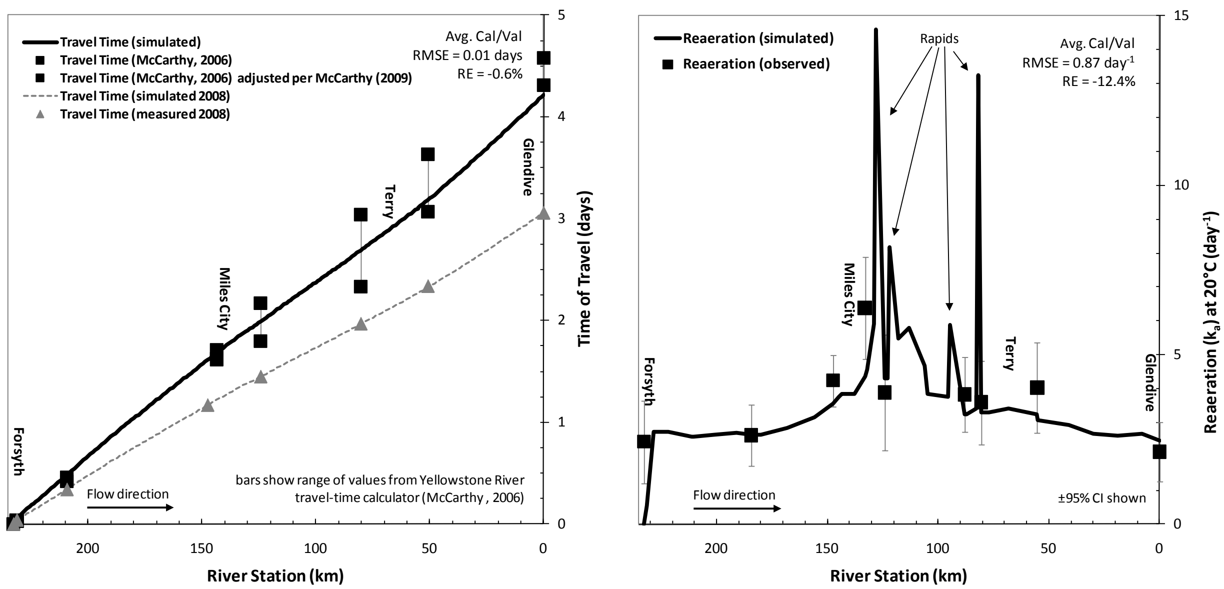

Figure 10-3. Verification of travel-times for the Yellowstone River during 2007.
(Left panel). Simulated and observed travel-time for 2007 and 2008 flow conditions. (Right panel) Simulated and observed reaeration for the Yellowstone River during 2007. It should be noted that reaeration from the Cartersville 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.

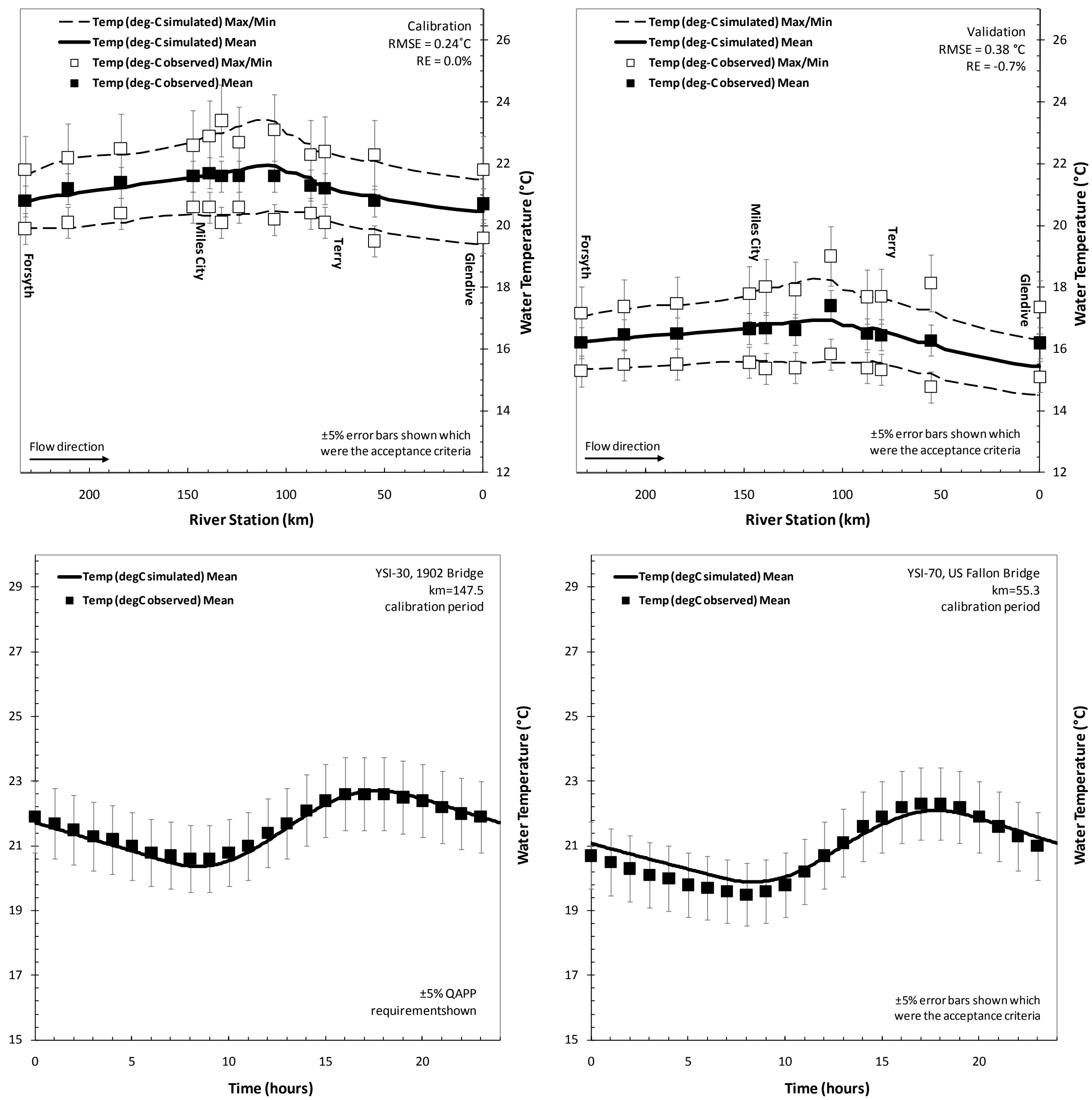

Figure 10-4. Water temperature simulation for the lower Yellowstone River during 2007.
(Top left/right panel) Simulated and observed water temperature for the August calibration and September validation periods. (Bottom left/right panel) Diurnal simulations for km 147.5 (1902 Bridge) and km 55.3 (upstream 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.

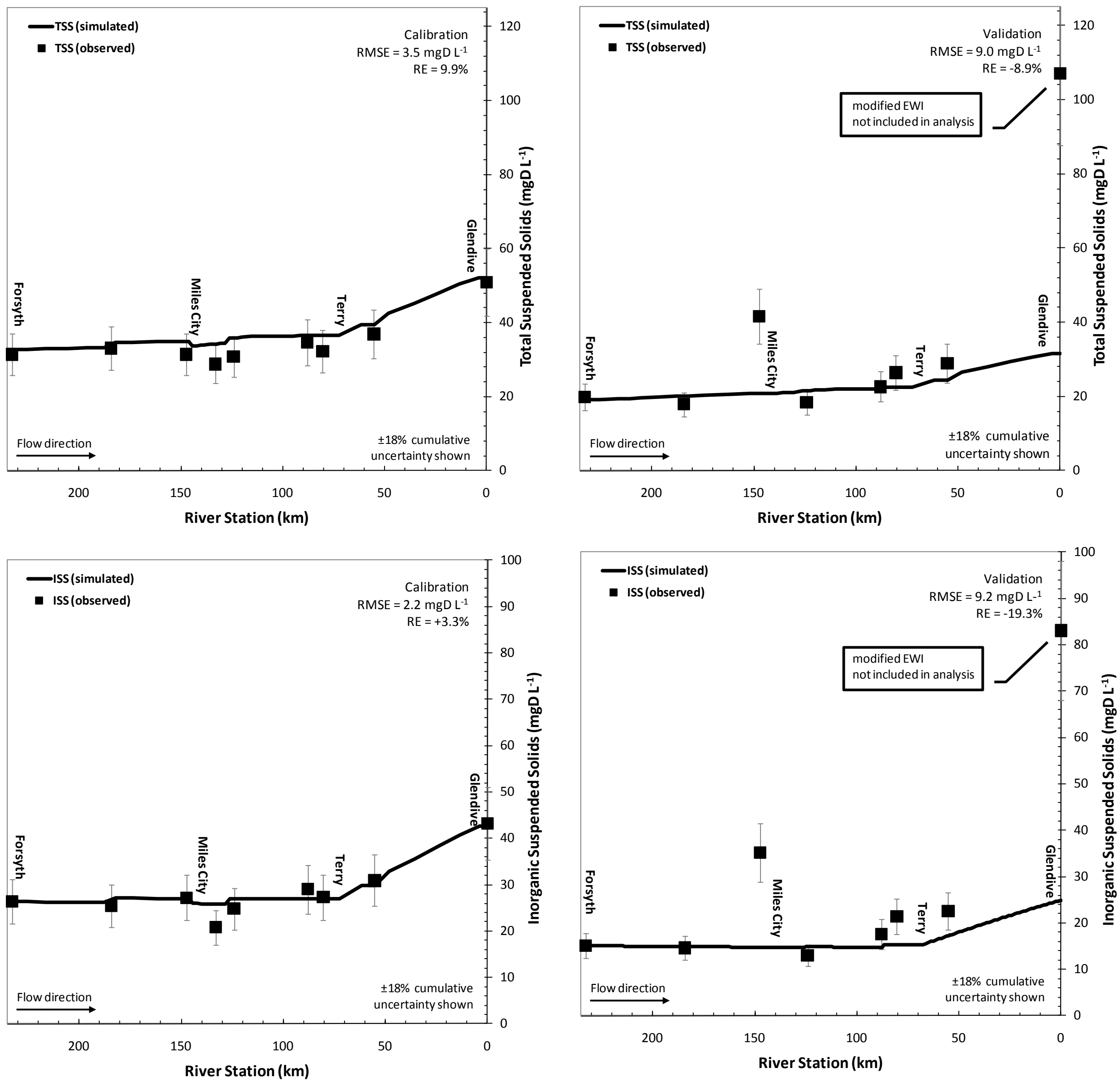

Figure 10-5. TSS and ISS simulations for the lower Yellowstone River during 2007.
(Top left/right panel) Simulated and observed TSS values for the August and September evaluation period. (Bottom left/right panel) Same but for ISS. The ±18% cumulative uncertainty is based on Harmel, et al., (2006).

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.

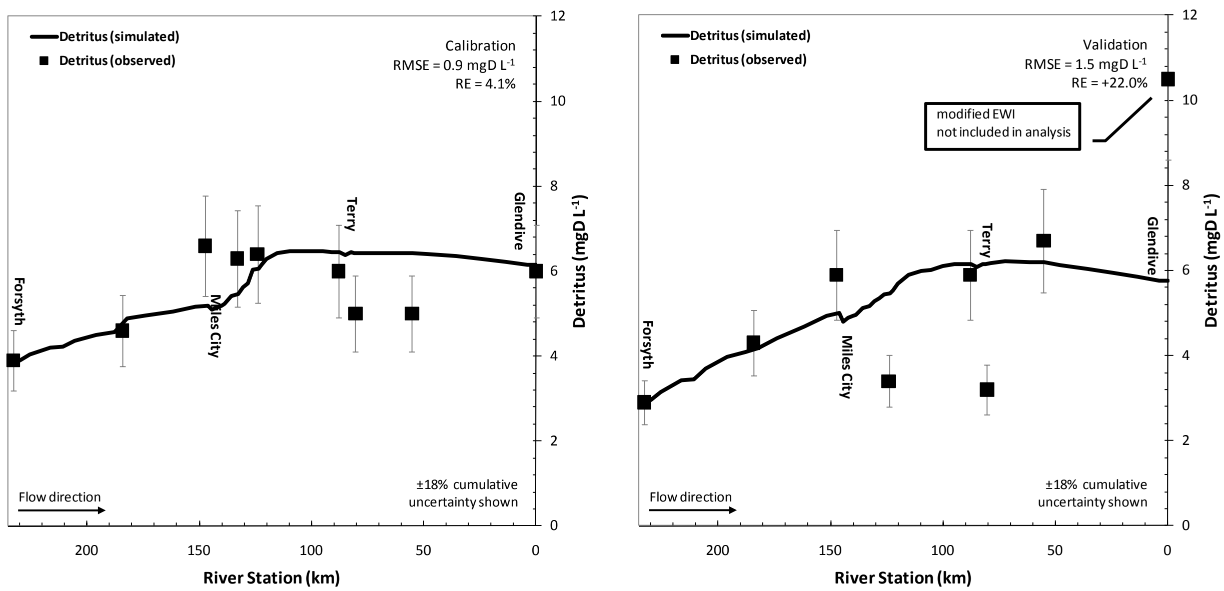

Figure 10-6. Detritus simulation for the lower Yellowstone River during 2007.
(Left panel) Simulated and observed detrital values for the August calibration period. (Right panel) Same but for the validation. The ±18% cumulative uncertainty is from Harmel, et al., (2006) values for TSS.

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].

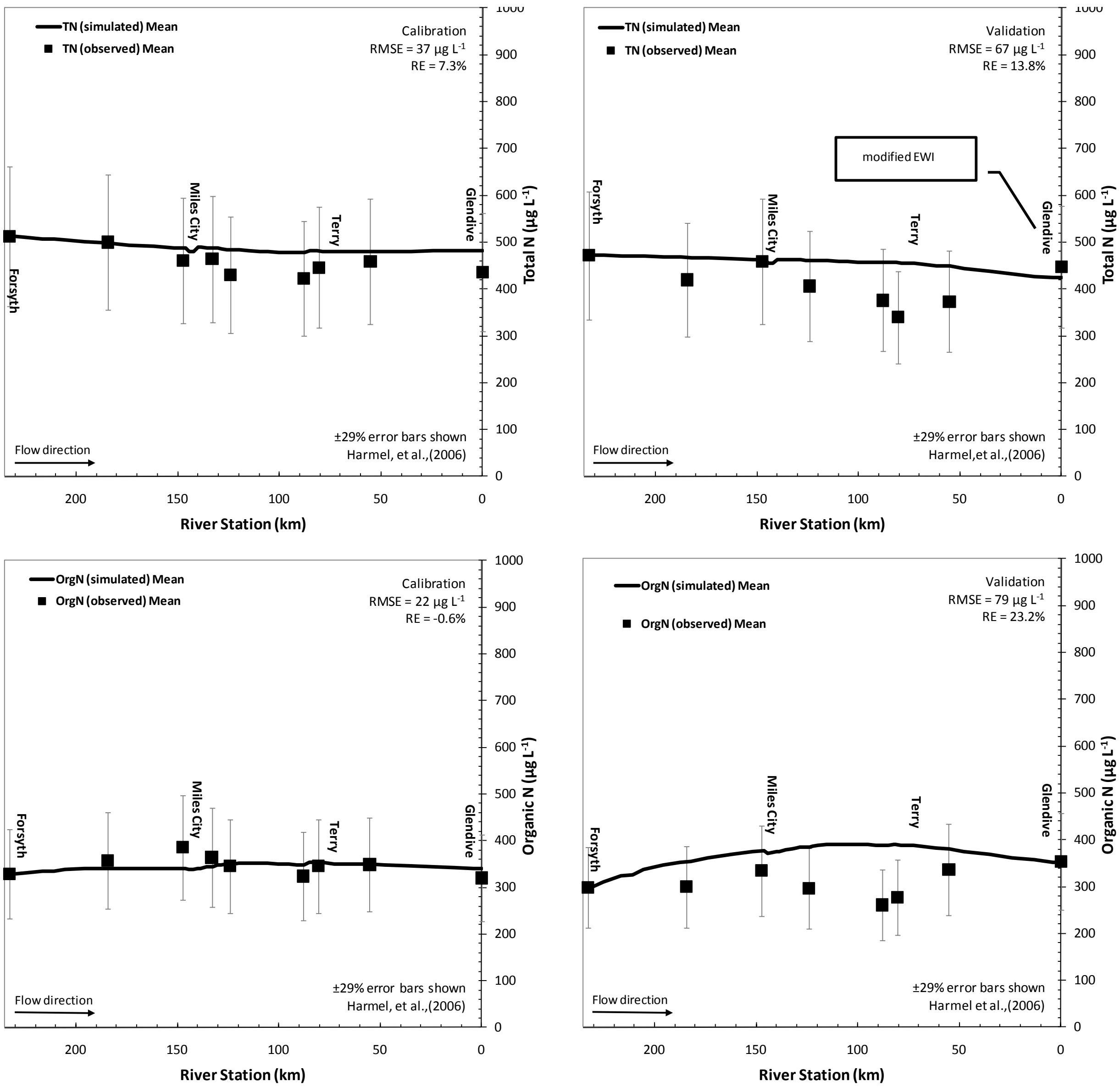

Figure 10-7. Total and organic N simulations for the Yellowstone River during 2007.
(Top left/right panel). Simulated TN for the August and September 2007 calibration and validation periods. (Bottom left/right panel). Same but for OrgN. The system appears to perhaps behave differently than the model 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).

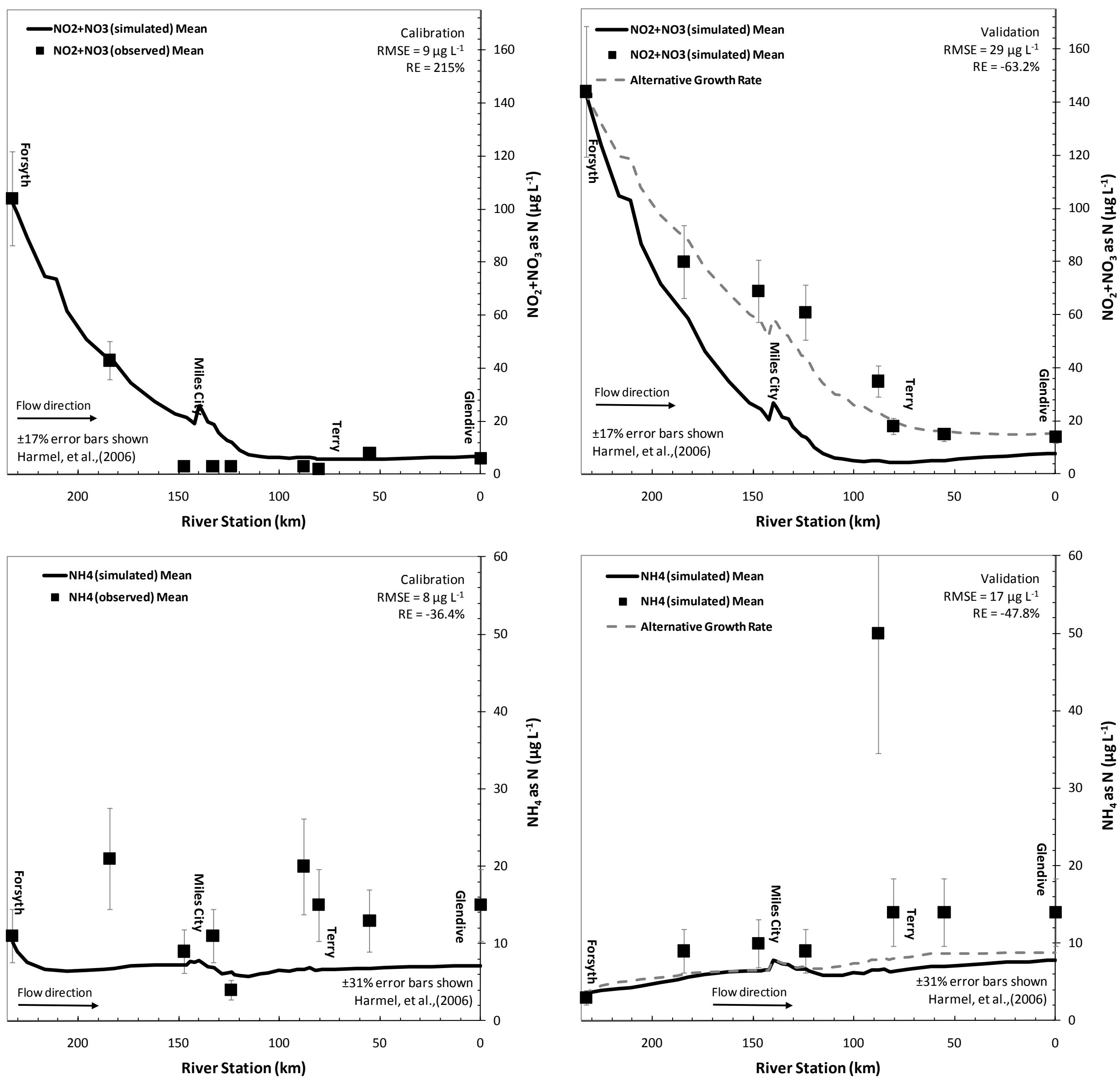

Figure 10-8. Dissolved nitrogen simulations for the Yellowstone River during 2007.
(Top left/right panel) Nitrate simulations for the August calibration and September period. (Bottom left/right panel) Same but for ammonium. The alternative growth rate illustrates the change in benthic algal growth rate 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.

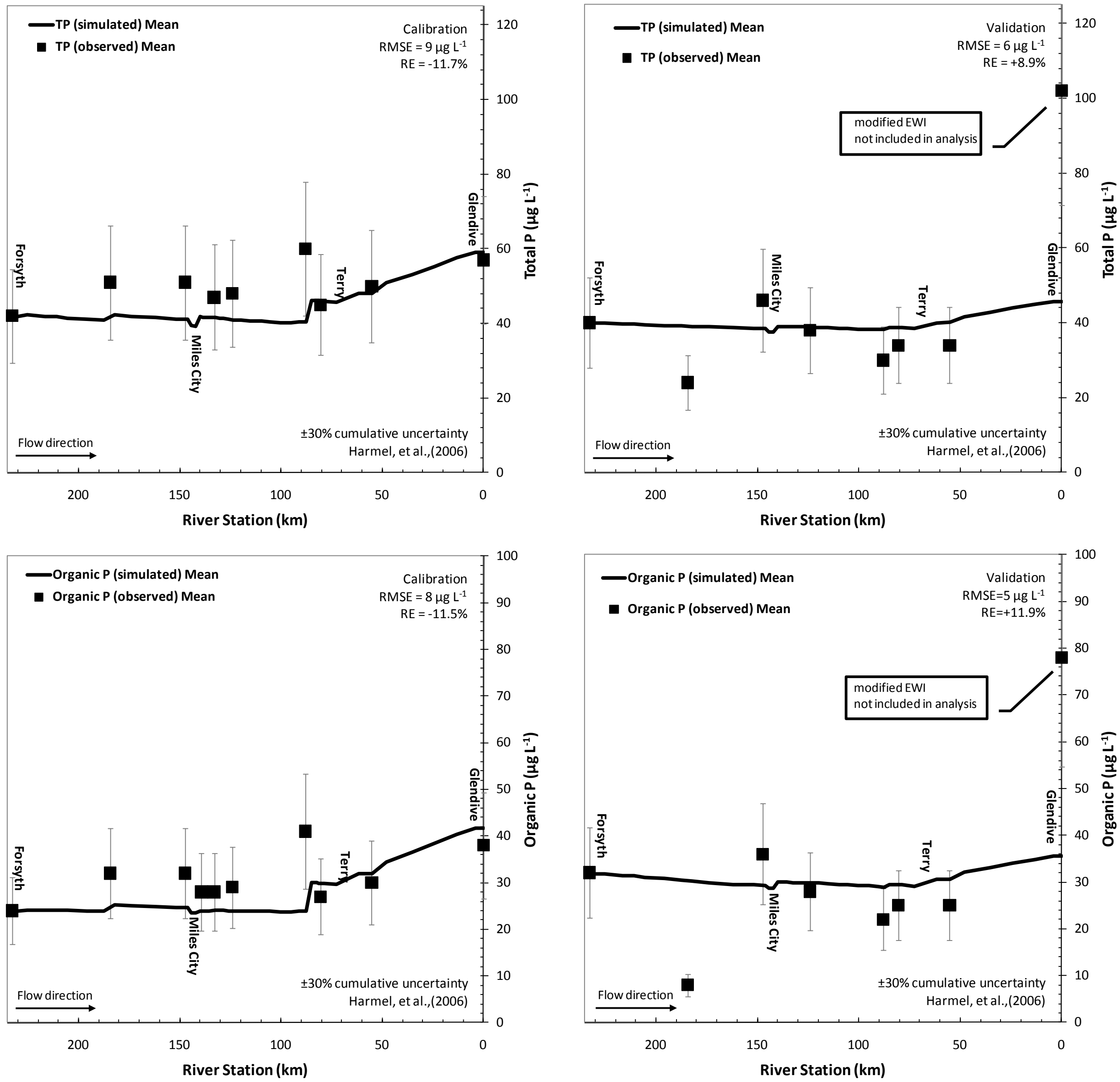

Figure 10-9. Total and organic phosphorus simulations for the Yellowstone River during 2007. (Top left/right panel) Simulated TP for the August and September 2007 calibration and validation periods. (Bottom left/right panel) Same but for OrgP. The problems shown in previous validation plots (i.e., for 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.

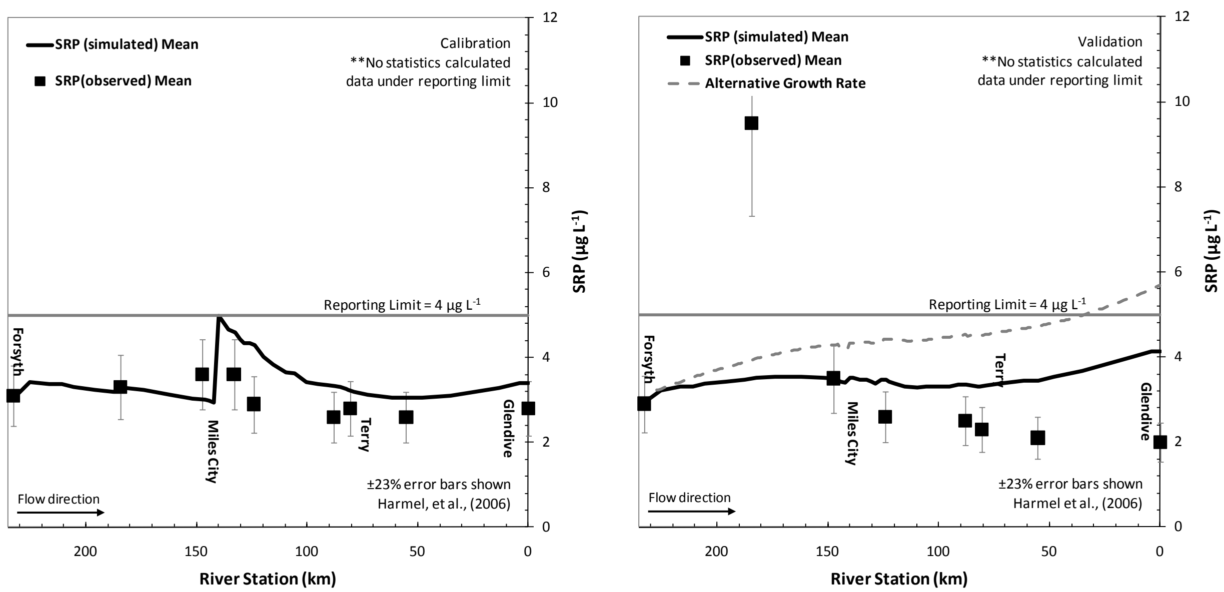

Figure 10-10. Soluble phosphorus simulations for the Yellowstone River during 2007.
(Left panel) Simulated SRP for the August calibration period. (Right panel) Same but for the September validation. Very little SRP was discharged by Miles City in September of 2007. No statistics were computed for SRP 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.

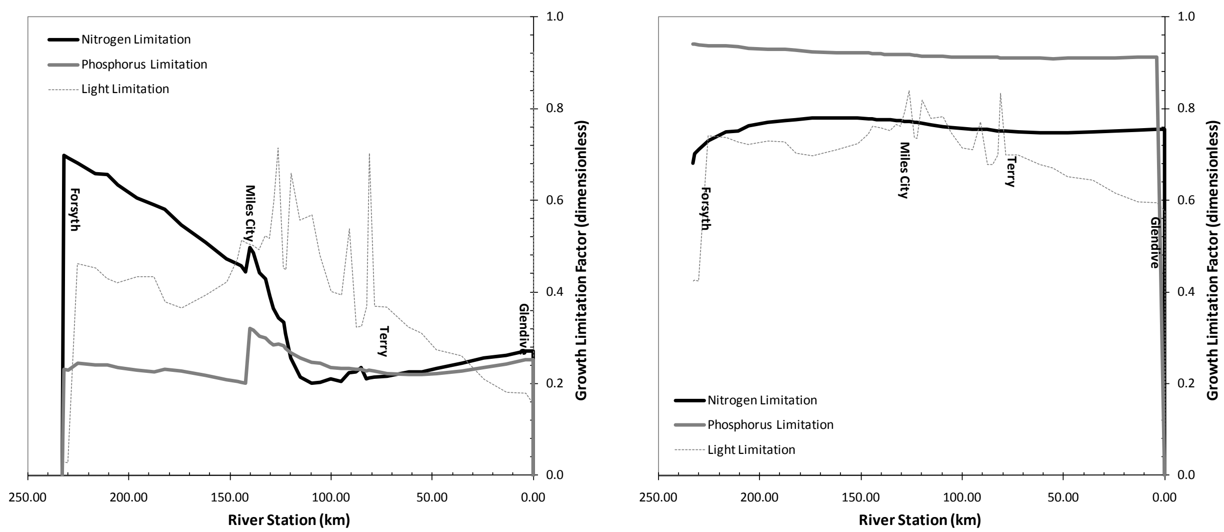

Figure 10-11. Nutrient and light limitation factors for the Yellowstone River during 2007.
(Left panel) Benthic algae nutrient and light limitation for the Yellowstone River. The lowest ordinate reflects the most limiting factor in any given case. (Right panel) Same but for phytoplankton. Benthic algae 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.

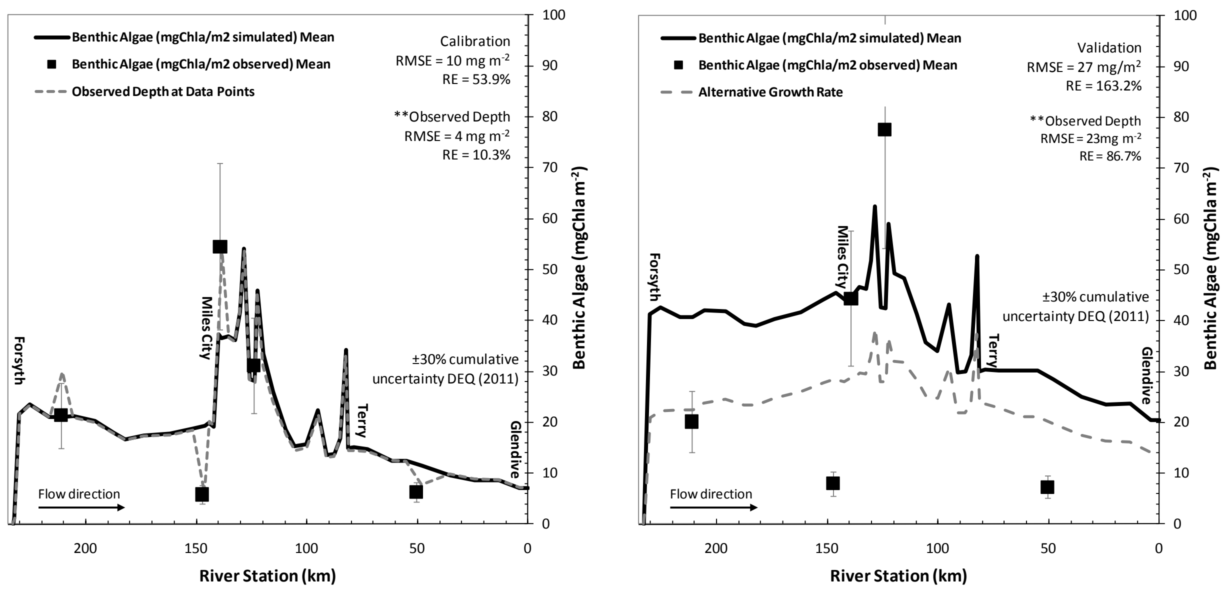

Figure 10-12. Simulated benthic algae Chl*a* for the Yellowstone River during 2007.
(Left panel) Simulated and observed benthic algae Chl*a* for the August 2007 calibration period. (Right panel) Same but for the September 2007 validation.

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.

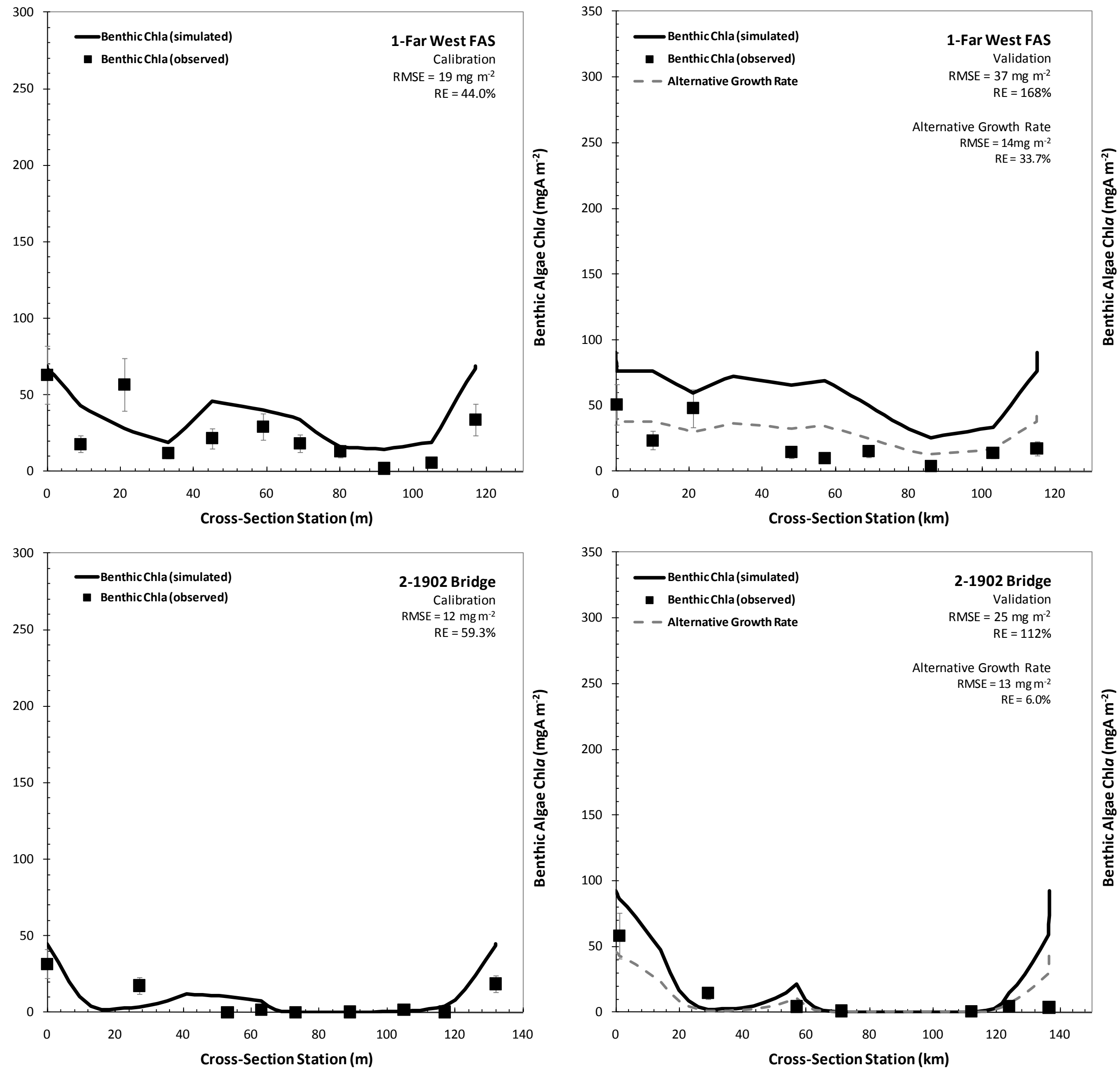

Figure 10-13. AT2K simulations of lateral algal distribution in Yellowstone River.
(Left panel) Simulated and observed values for each of the transects evaluated in August 2007. (Right panel) Same but for the validation. The alternative growth rate reflects the benthic algae growth rate determined previously to meet the productivity response of the river (see previous discussions). Field and lab variability estimated to be ±30% (DEQ, 2011).

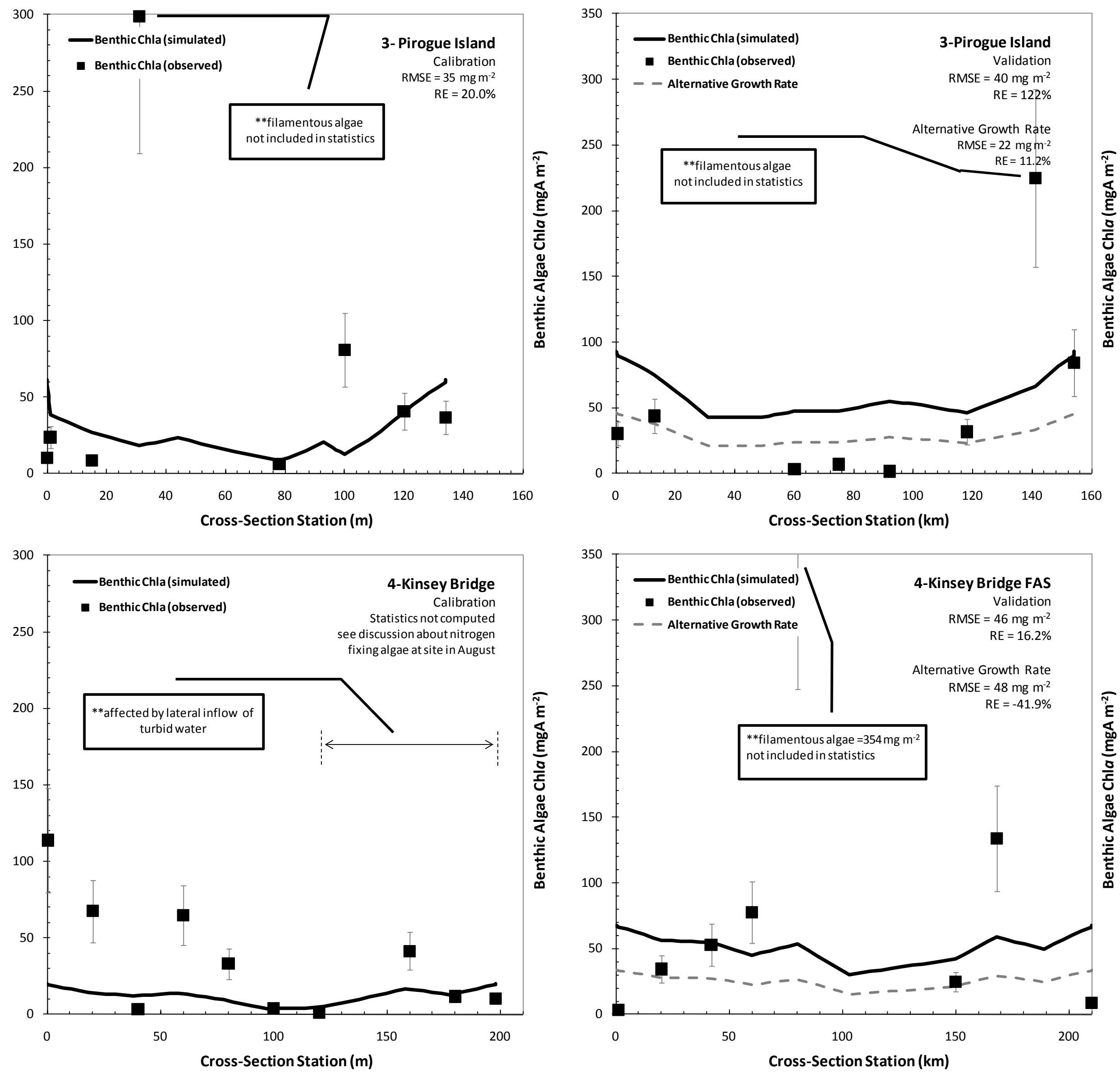

Figure 10-13 (cont.). AT2K simulations of lateral algal distribution in Yellowstone River.
(Left panel) Simulated and observed values for each of the transects evaluated in August 2007. (Right panel) Same but for the validation. The alternative growth rate reflects the benthic algae growth rate determined previously to meet the productivity response of the river (see previous discussions). Field and lab variability estimated to be ±30% (DEQ, 2011).

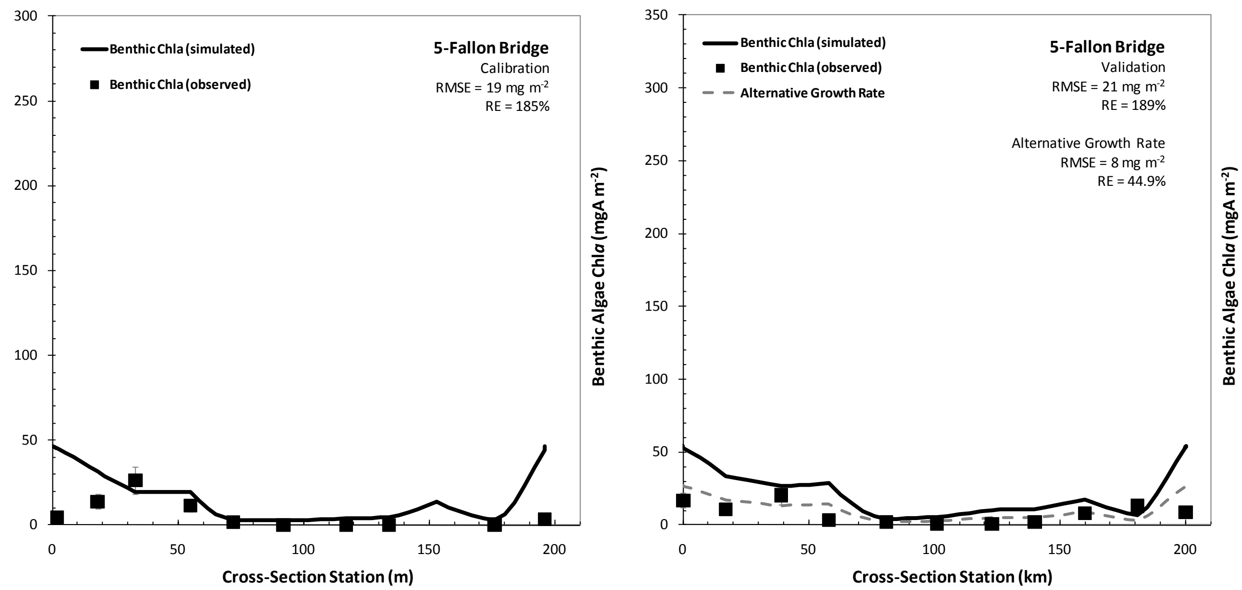

Figure 10-13 (cont.). AT2K simulations of lateral algal distribution in Yellowstone River
(Left panel) Simulated and observed values for each of the transects evaluated in August 2007. (Right panel) Same but for the validation. The alternative growth rate reflects the benthic algae growth rate determined 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.

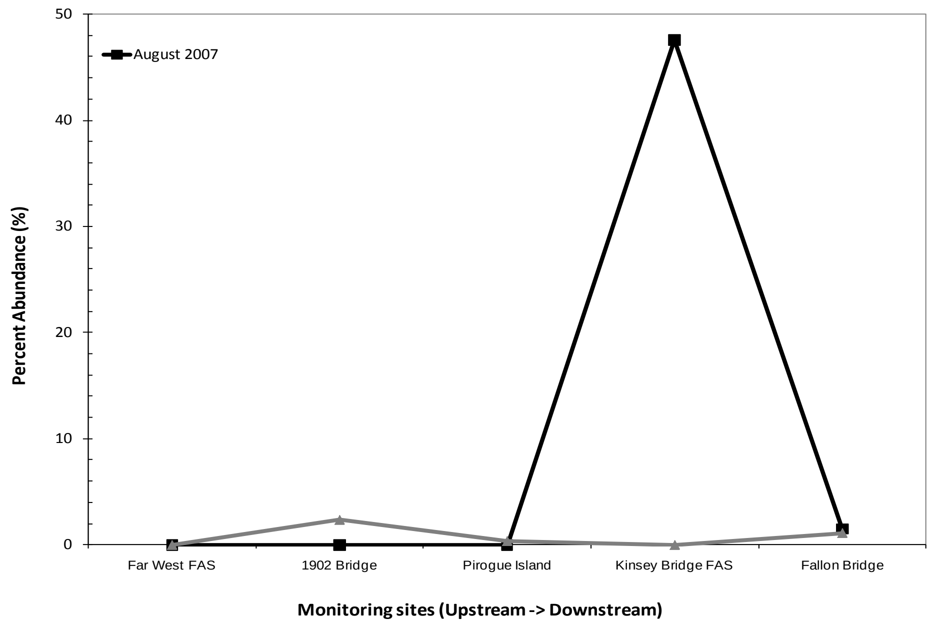

Figure 10-14. Percent abundance of nitrogen fixing *Epithemia sorex* at Kinsey Bridge FAS in 2007.
The Kinsey FAS site was the only site with a large percentage of nitrogen fixing algae, which is suggestive of very strong nitrogen limitation. It also explains the deviation between the model simulations and observed algal biomasses at this site. Data reproduced with permission from Charles and Christie (2011).

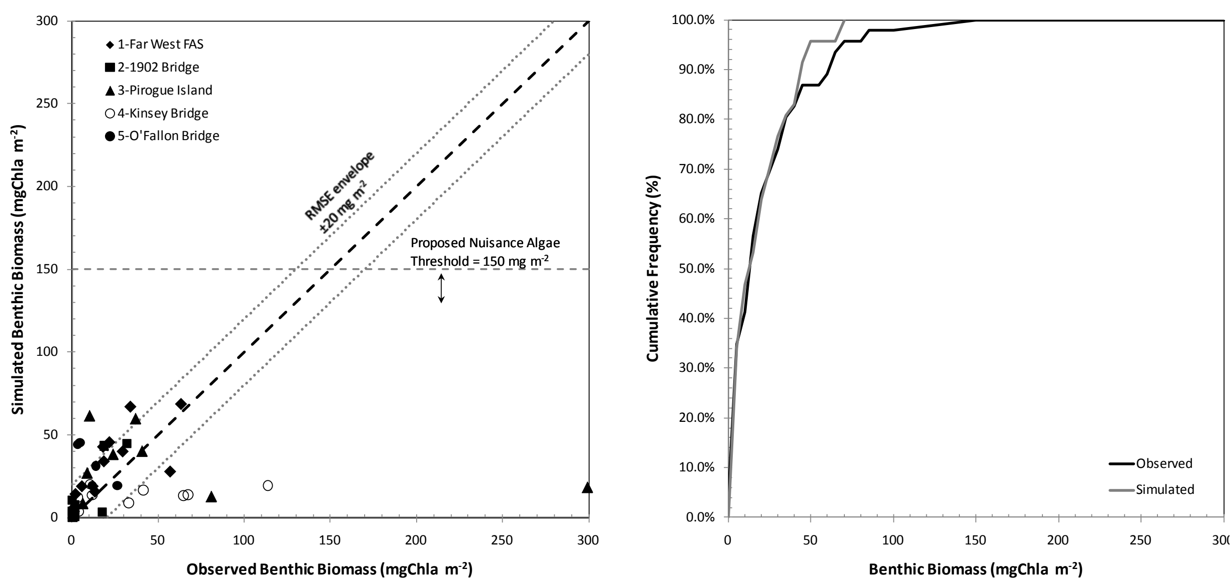

Figure 10-15. Benthic algal biomass lateral simulation reliability.
(Left panel) 1:1 plots of model simulations for the calibration period. A RMSE envelope is shown which represents a plausible margin of safety to account for simulation error as discussed previously. (Right panel) Cumulative frequency 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.

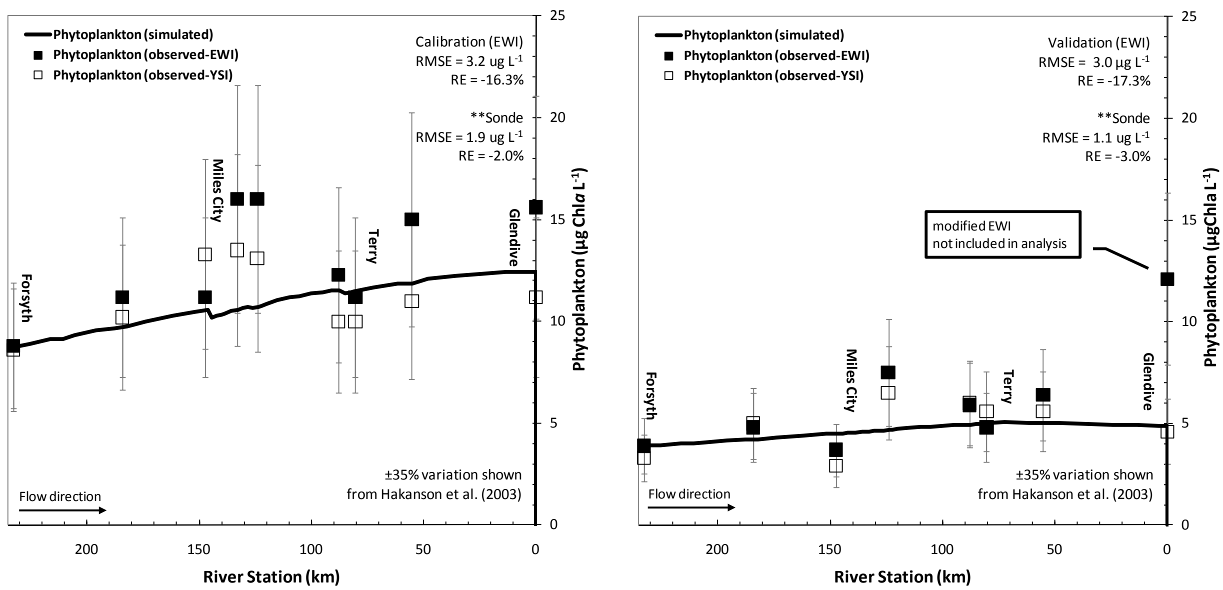

Figure 10-16. Phytoplankton simulations for the Yellowstone River during 2007.
(Left panel) Simulated and observed phytoplankton Chl*a* simulations for the August 2007 calibration period. (Right panel) Same but for the September validation. Both the EWI and sonde data are shown.

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

[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.

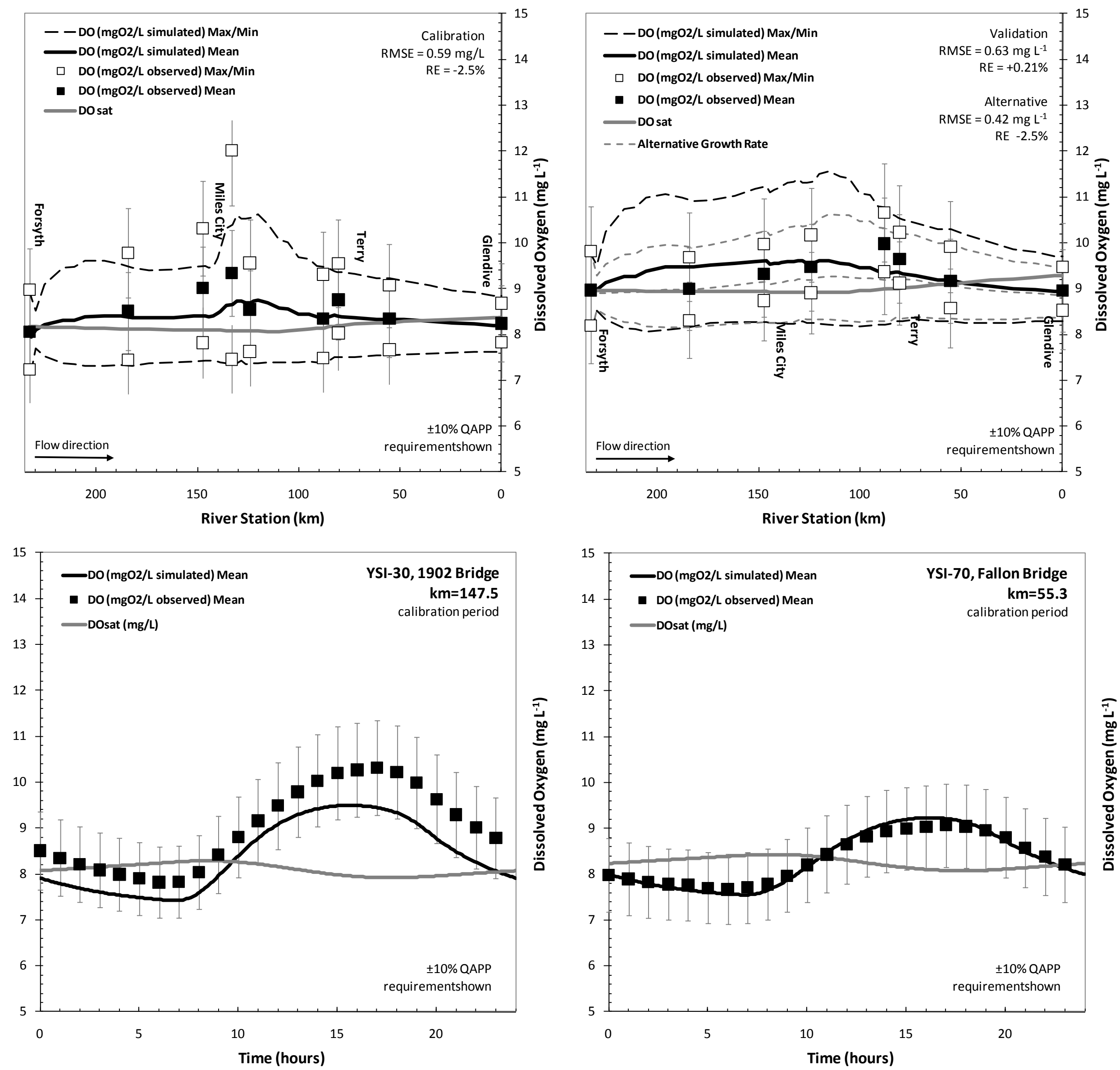

Figure 10-17. DO simulations for the lower Yellowstone River during 2007.
(Top left/right panel) Simulated and observed DO for August and September 2007. The highest areas of productivity are near Miles City and moderate downstream due to light limitation. (Bottom left/right panel) Diurnal simulations 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.

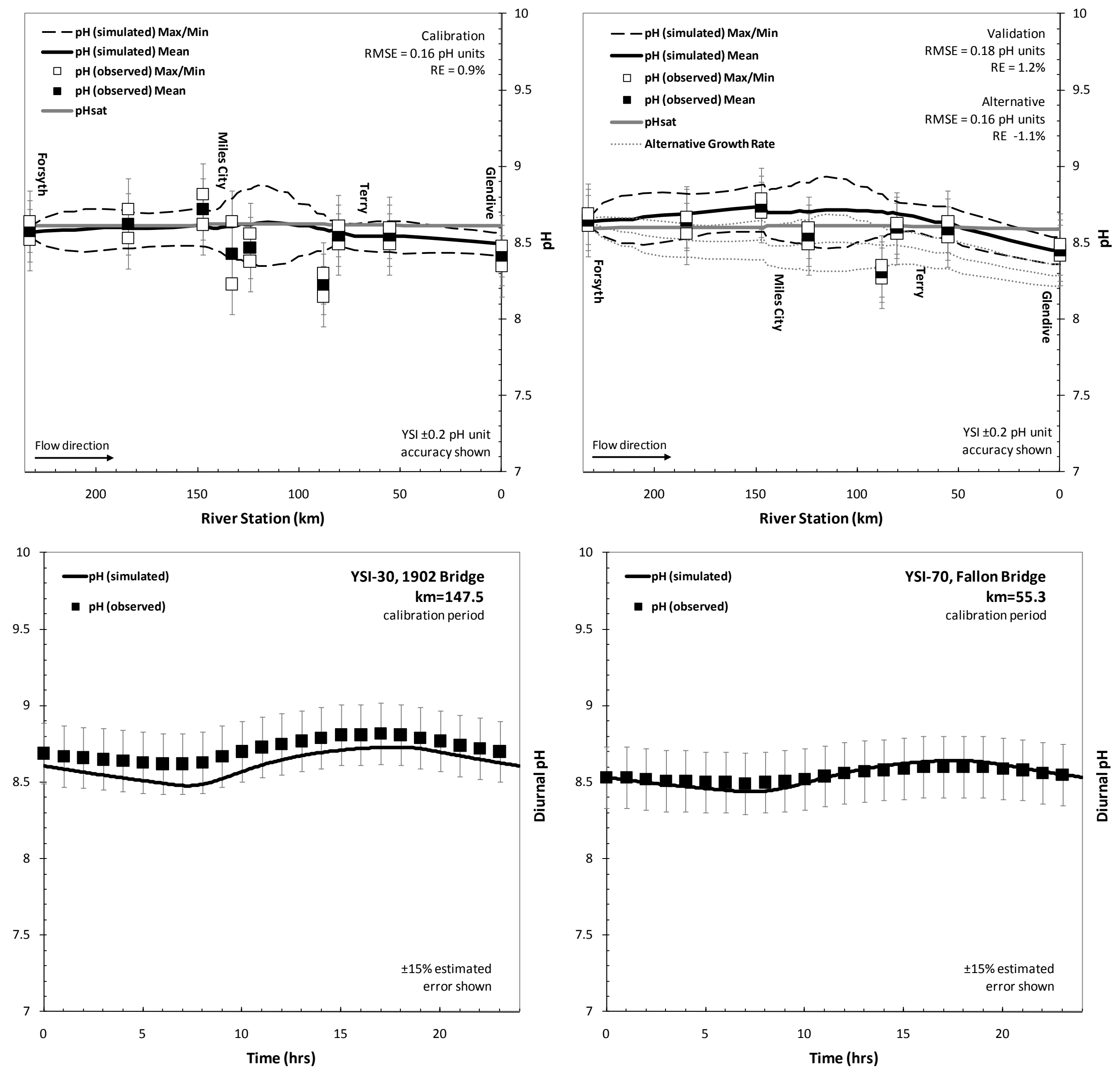

Figure 10-18. pH simulations for the Yellowstone River during 2007.
(Top left/right panel) Simulated and observed pH (S.U.) for the August and September 2007 calibration and validation periods. (Bottom left/right panel) Diurnal simulations for several of the sites previous; the 1902 Bridge (km 147.5) and Fallon Bridge upstream of O'Fallon Creek (km 55.3).

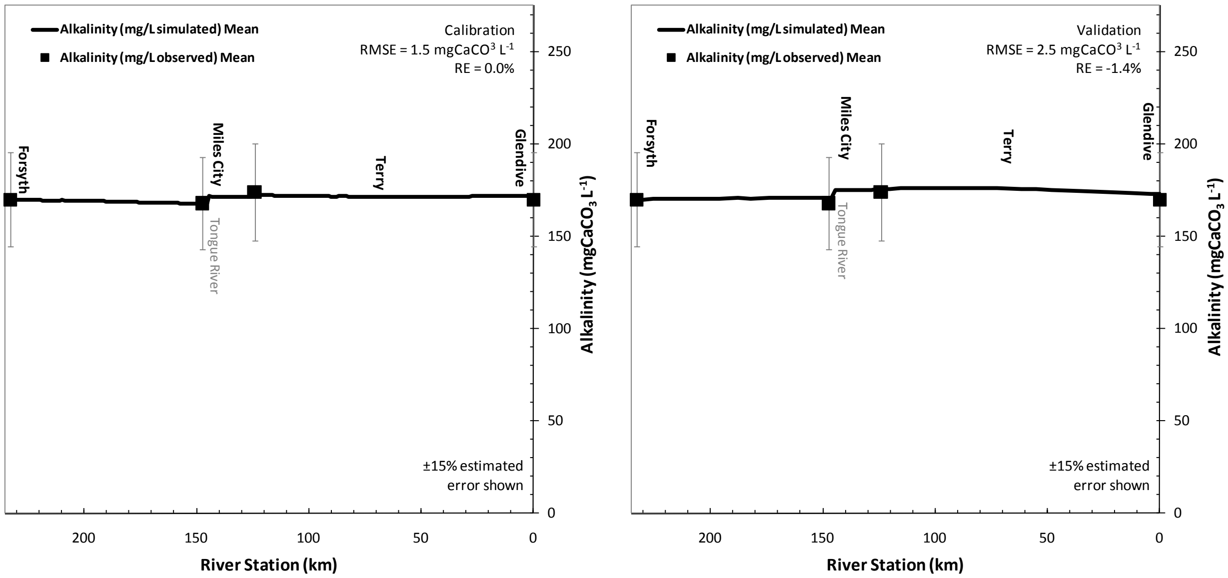

Figure 10-19. Alkalinity simulations for the lower Yellowstone River during 2007.
(Left panel) Simulated and observed alkalinity for the August 2007 calibration period. (Right panel) Same but for the validation period.

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).

[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 $\frac{1\text{mole } O_2}{1\text{mole C}} \times \frac{32\text{g } O_2}{1\text{mole } O_2} \times \frac{1\text{mole C}}{12\text{ g C}}$ was used.

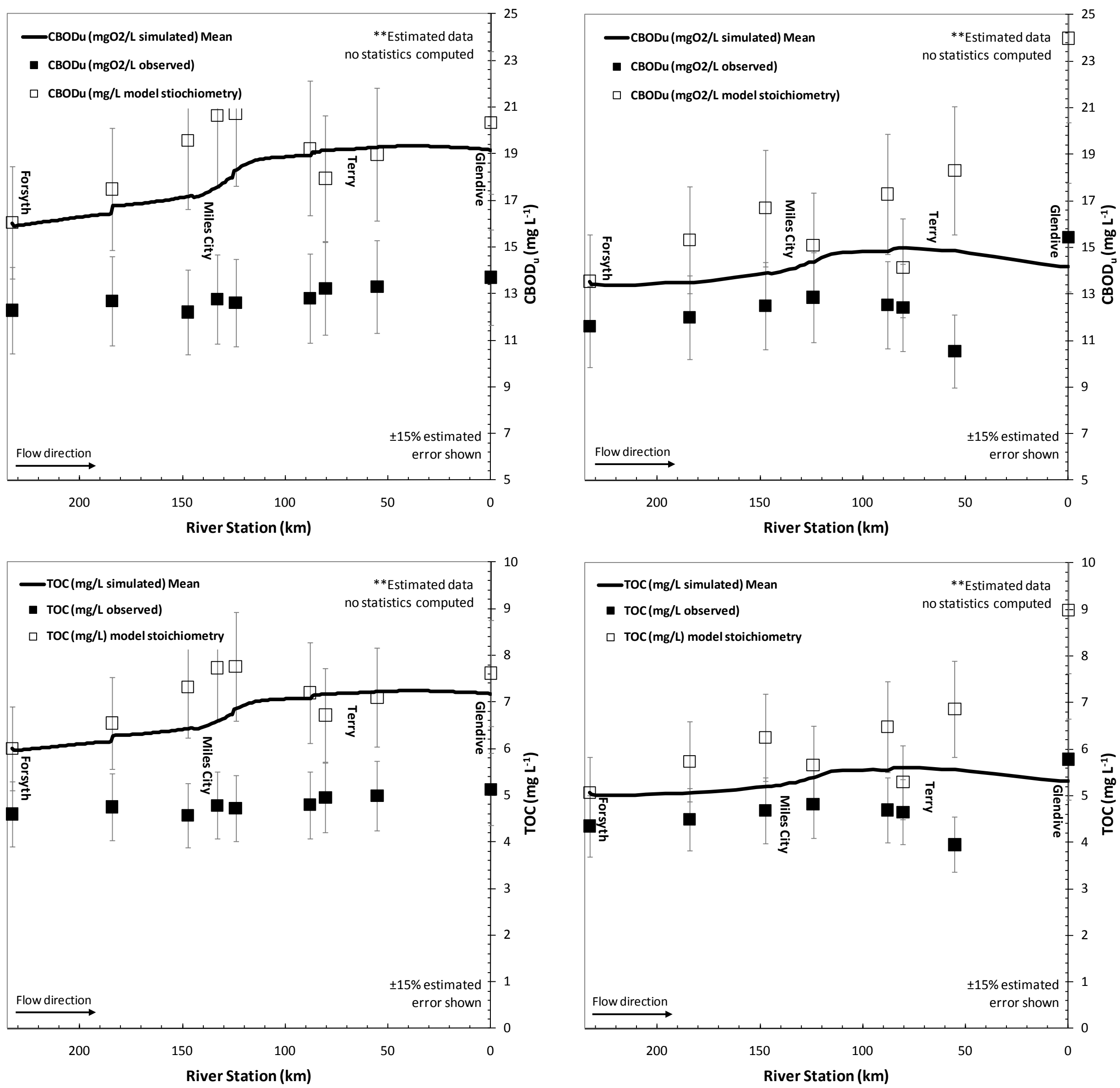

Figure 10-20. CBOD-ultimate and TOC simulations for the Yellowstone River during 2007.
(Top left/right panel) Simulated and estimated observed CBOD for the calibration and validation respectively. (Bottom left/right panel). Same but for TOC. The filled squares reflect the actual CBOD/TOC estimates for 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).

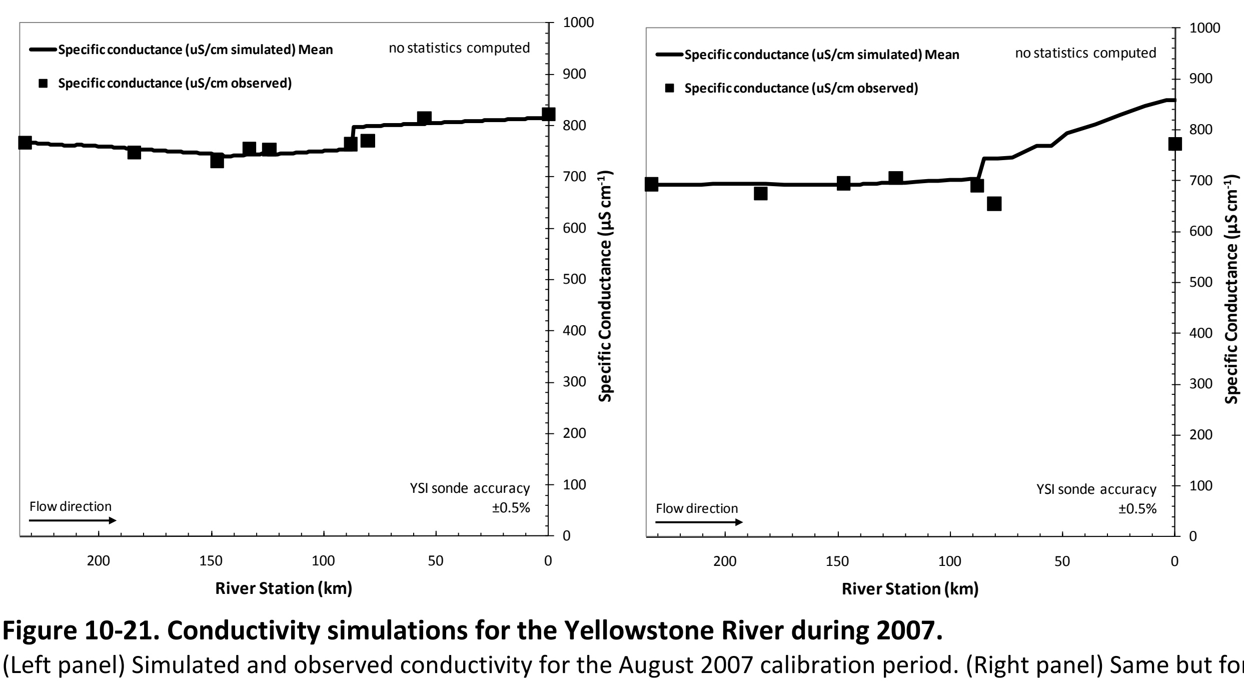

Figure 10-21. Conductivity simulations for the Yellowstone River during 2007.
(Left panel) Simulated and observed conductivity for the August 2007 calibration period. (Right panel) Same but for the validation period.

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.

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.

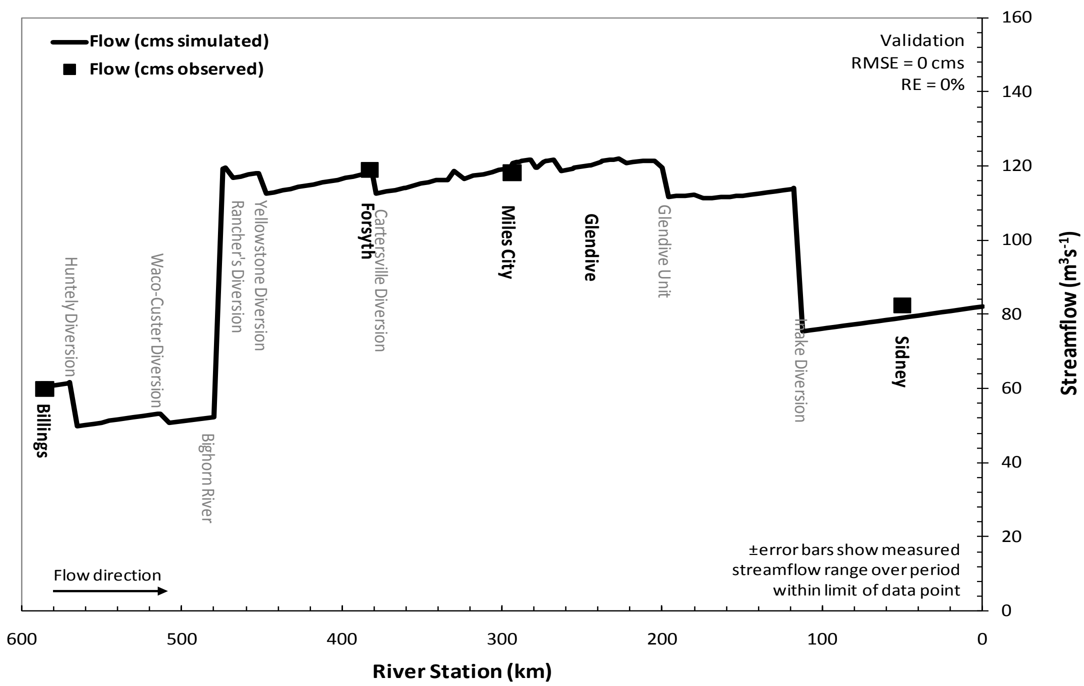

Figure 11-1. Simulated and observed streamflow for the Yellowstone River during 2000.

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.

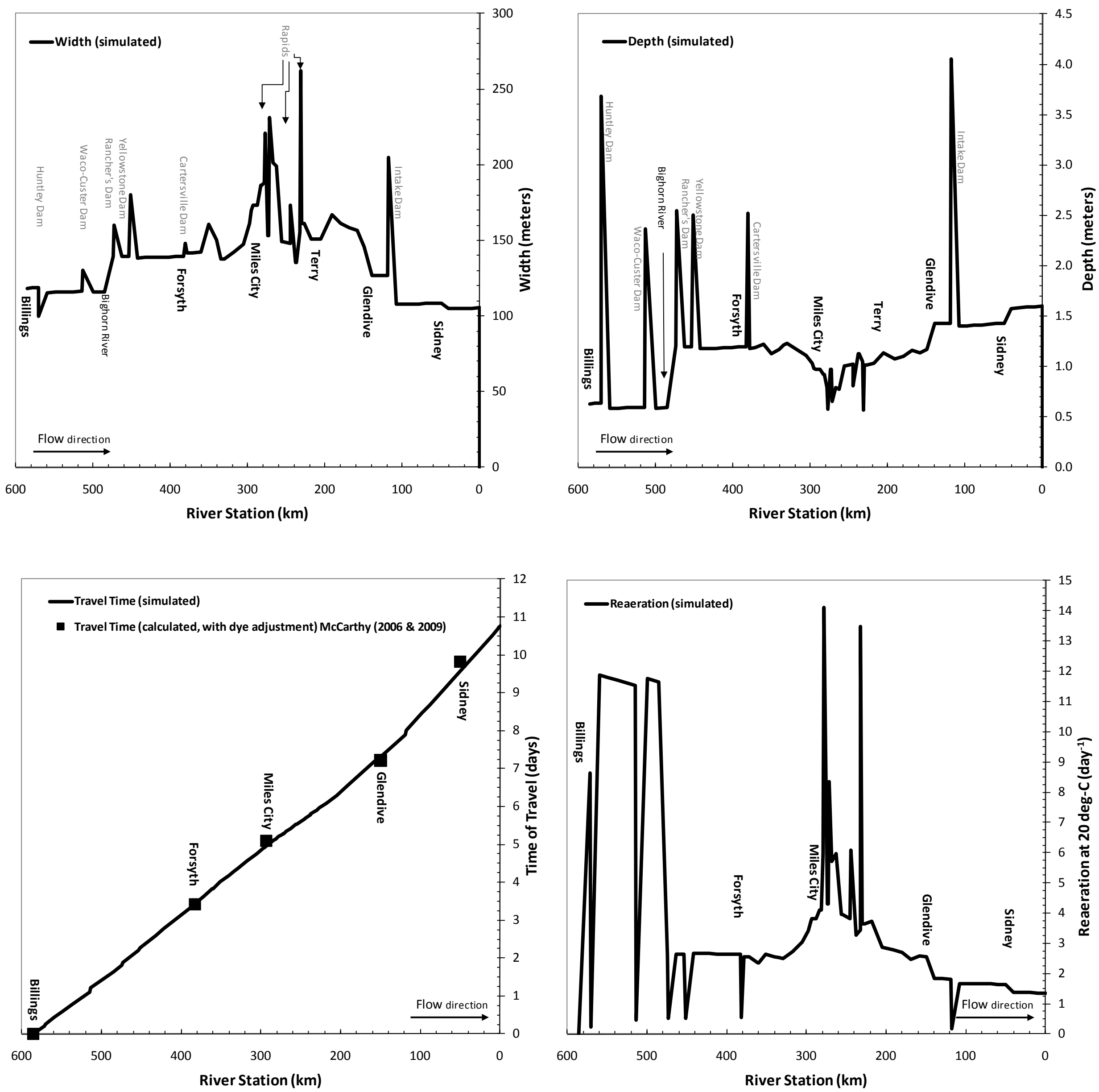

Figure 11-2. Mass transport indicators for the lower Yellowstone River during 2000.
(Top left/right panel). Simulated width and depth. (Bottom left/right panel) Travel-time and reaeration.

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.

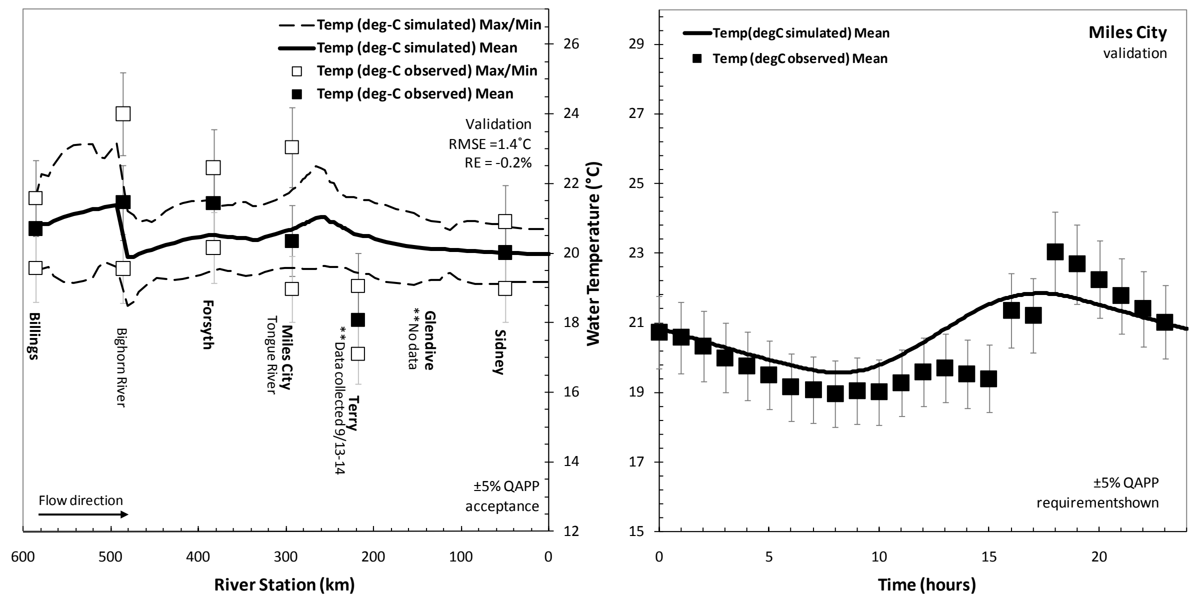

Figure 11-3. Temperature simulations for the Yellowstone River during 2000.
(Left panel) Simulated and observed values for the August 2000 validation period. (Right panel) Diurnal simulations for Miles City.

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

[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.

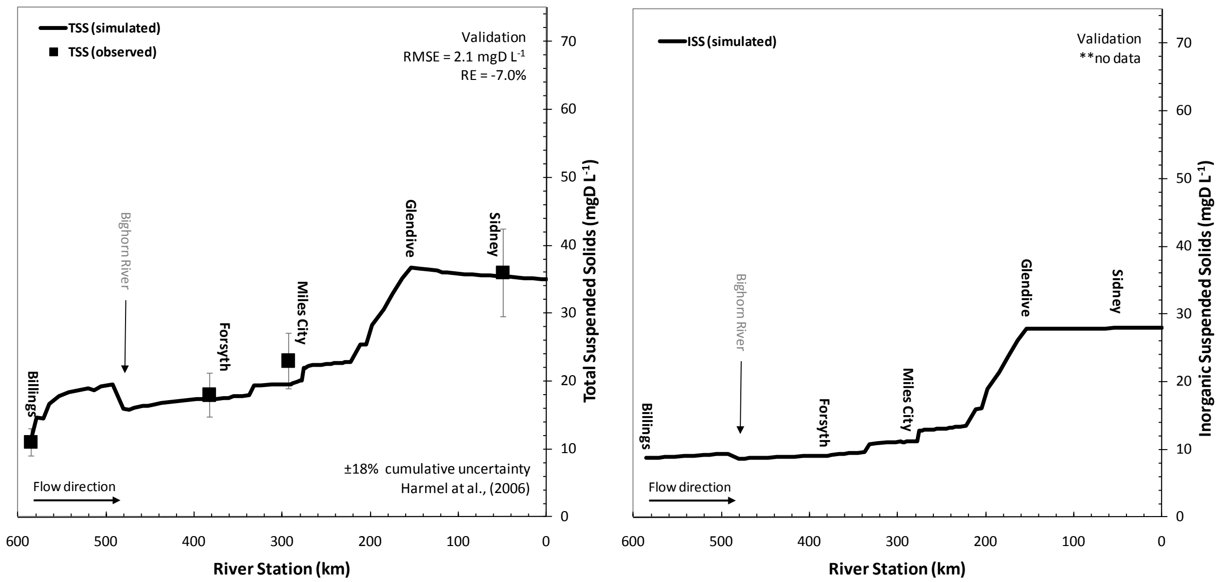

Figure 11-4. TSS simulations for the Yellowstone River during 2000.
(Left panel) Observed and simulated TSS. (Right Panel) Same but for ISS (no field data collected).

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.

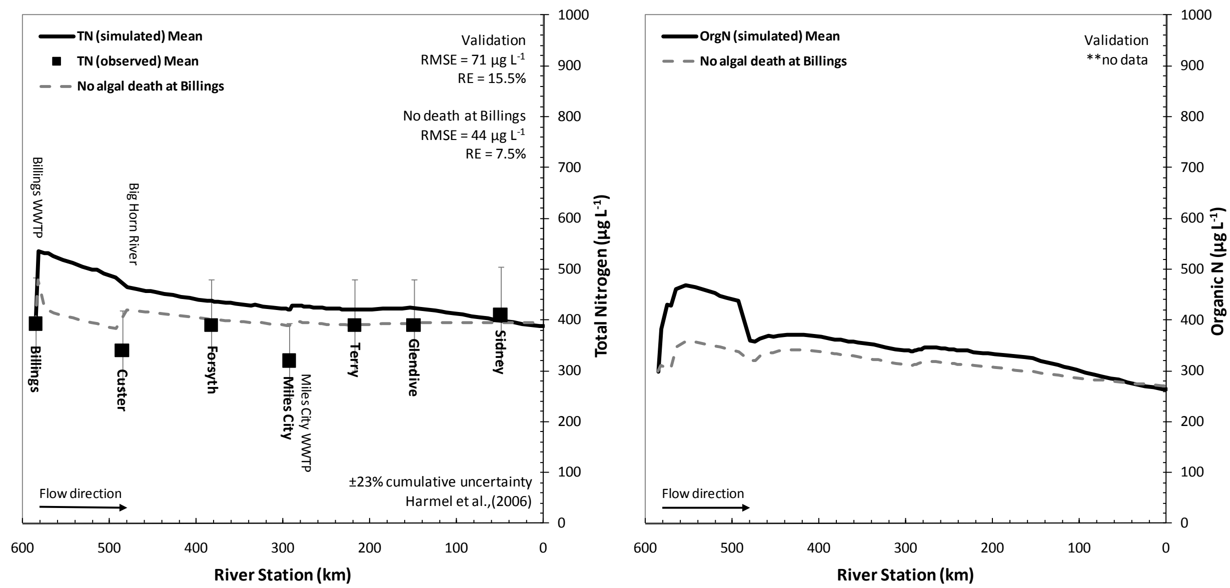

Figure 11-5. Total and organic N simulations for the Yellowstone River during August 2000.
(Left panel). Observed and simulated total nitrogen. (Right panel) Same but for organic nitrogen (for reference purposes only, no data collected).

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.

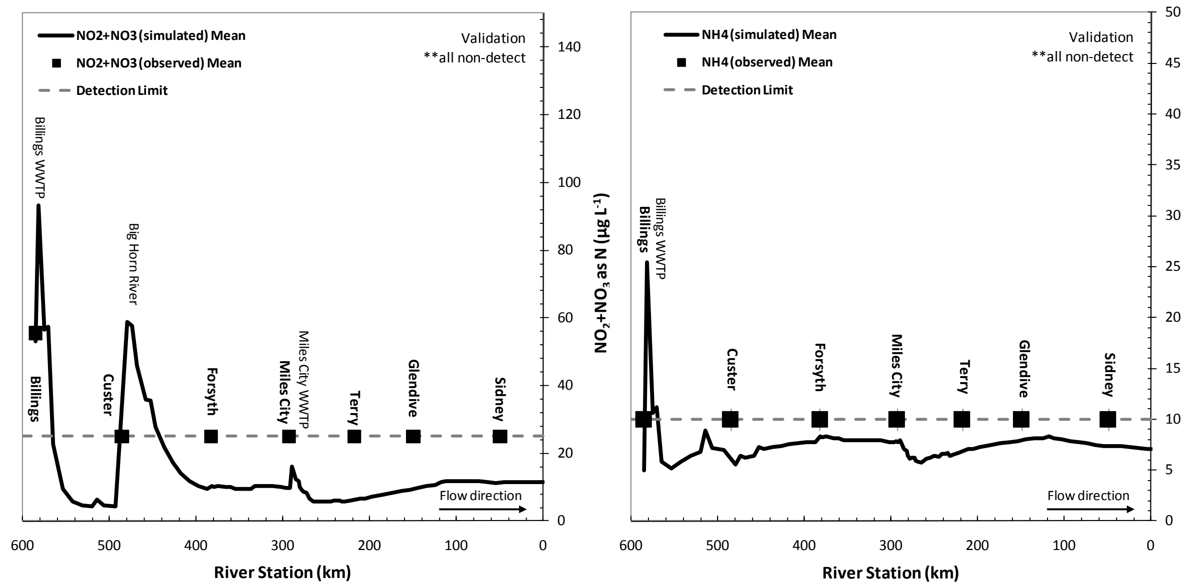

Figure 11-6. Nitrogen simulations for the Yellowstone River during August 2000.
(Left panel). Simulated and observed nitrate. (Right panel) Same but for ammonia. Both of the soluble N data are below the detection limits referenced by the dotted line.

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.

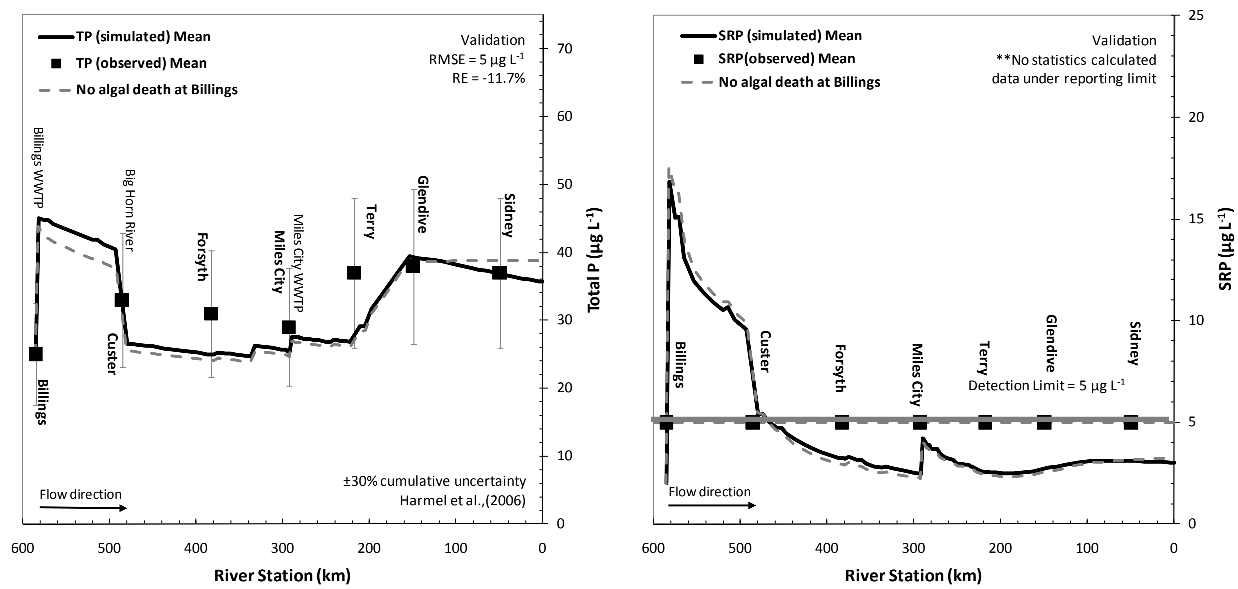

Figure 11-7. Phosphorus simulation for the Yellowstone River during August 2000.
(Left panel). Simulated and observed total phosphorus. (Right panel) Same but for SRP. Note that the USGS observation site at Billings is directly upstream of the WWTP.

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).

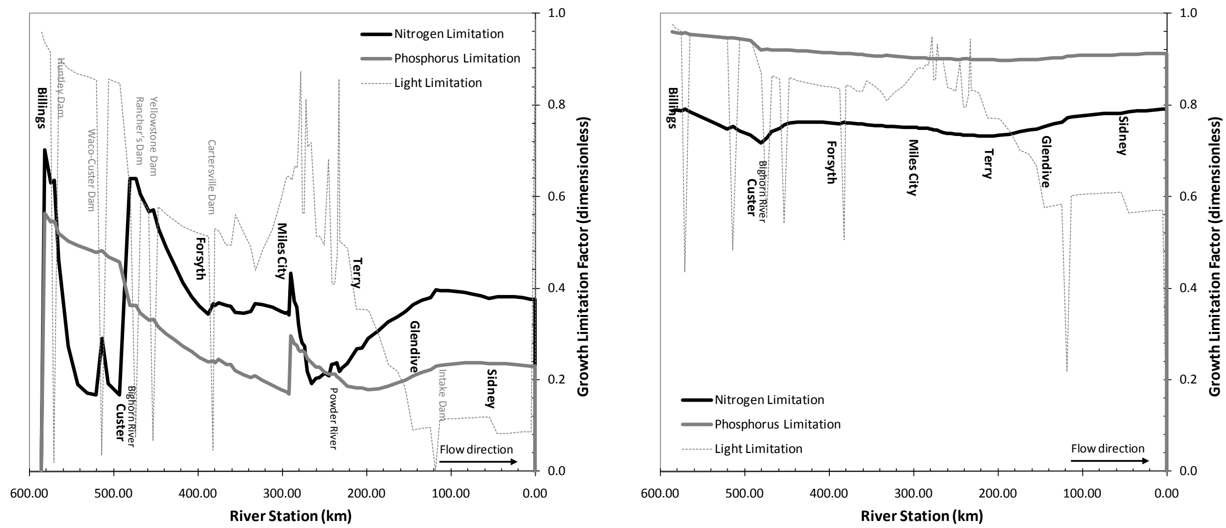

Figure 11-8. Nutrient and light limitation factors for the Yellowstone River during 2000.
(Left panel) Nutrient and light limitation factors for benthic algae. (Right panel) Same but for phytoplankton. The effect of the diversion dams are apparent on both benthic algae and phytoplankton as they increase 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.

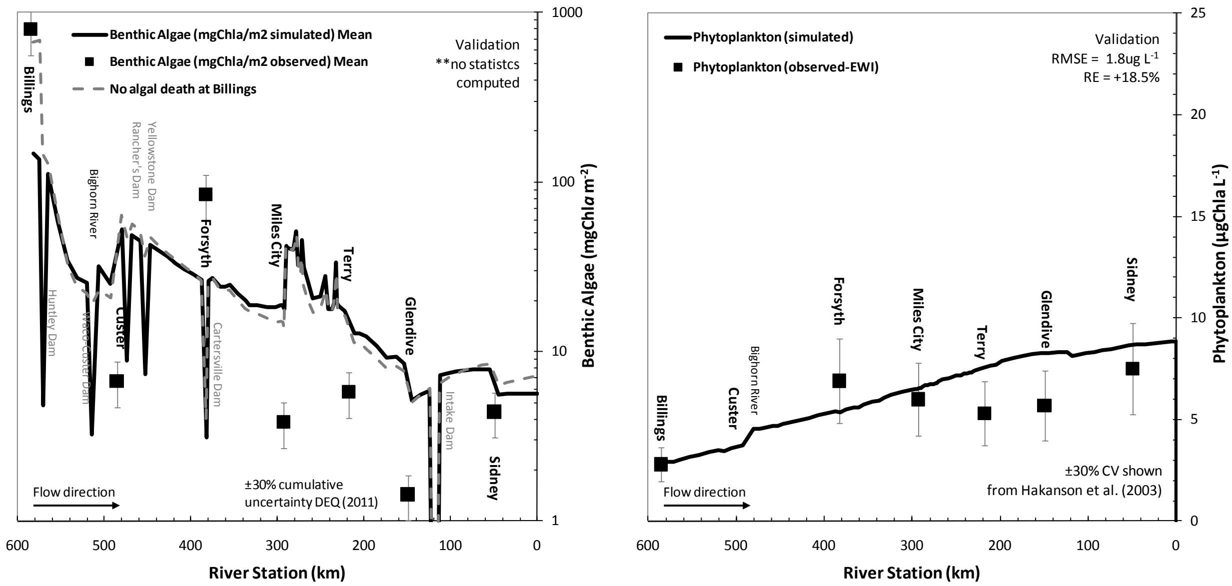

Figure 11-9. Algal simulations for the Yellowstone River during August 2000.
(Left panel) Simulated and observed benthic algae. (Right panel) Same but for phytoplankton.

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.

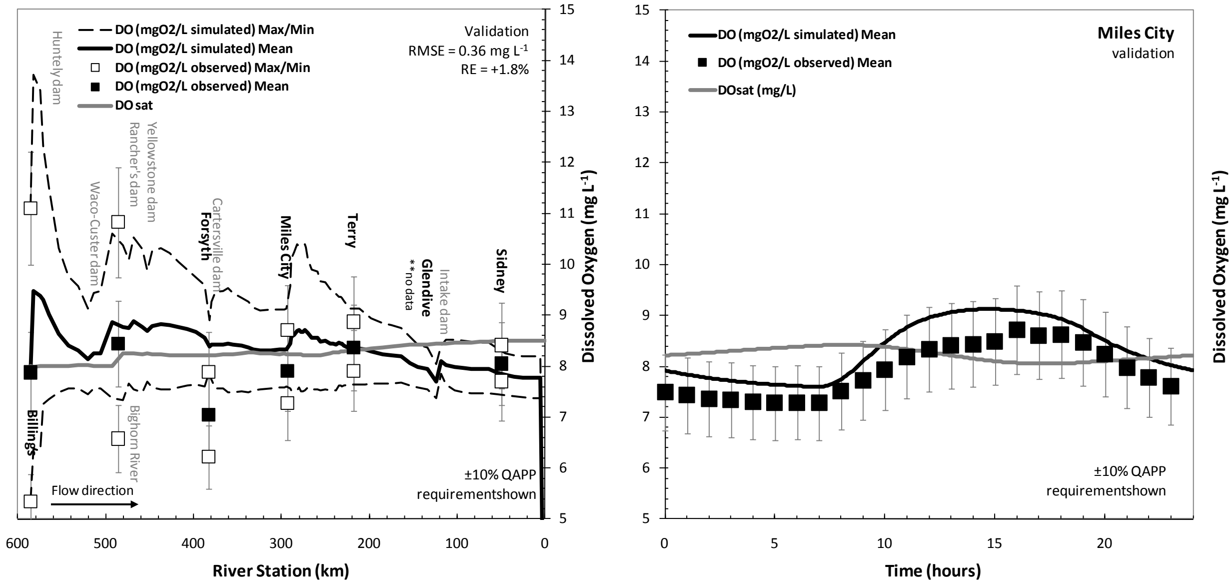

Figure 11-10. Dissolved oxygen simulations for the Yellowstone River during August 2000.
(Left panel) Simulated and observed DO. (Right panel) Diurnal DO simulations for Miles City.

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%.

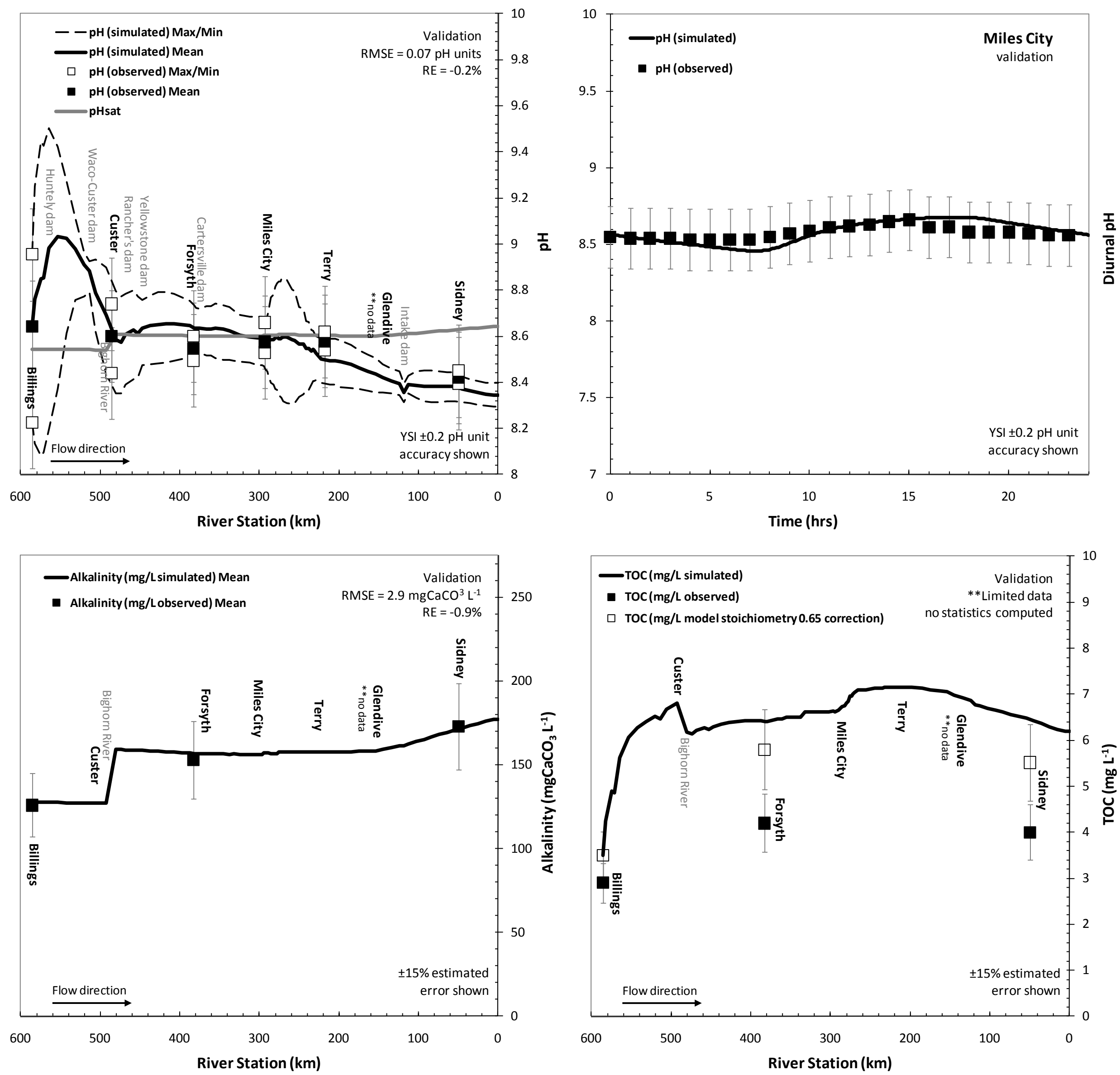

Figure 11-11. pH, alkalinity, and TOC simulations for the Yellowstone River during 2000.
(Top left/right panel) Simulated and observed longitudinal pH and diurnal pH simulation for Miles City. (Bottom left/right panel) Simulated and observed alkalinity and total organic carbon (TOC, as detailed in the 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.

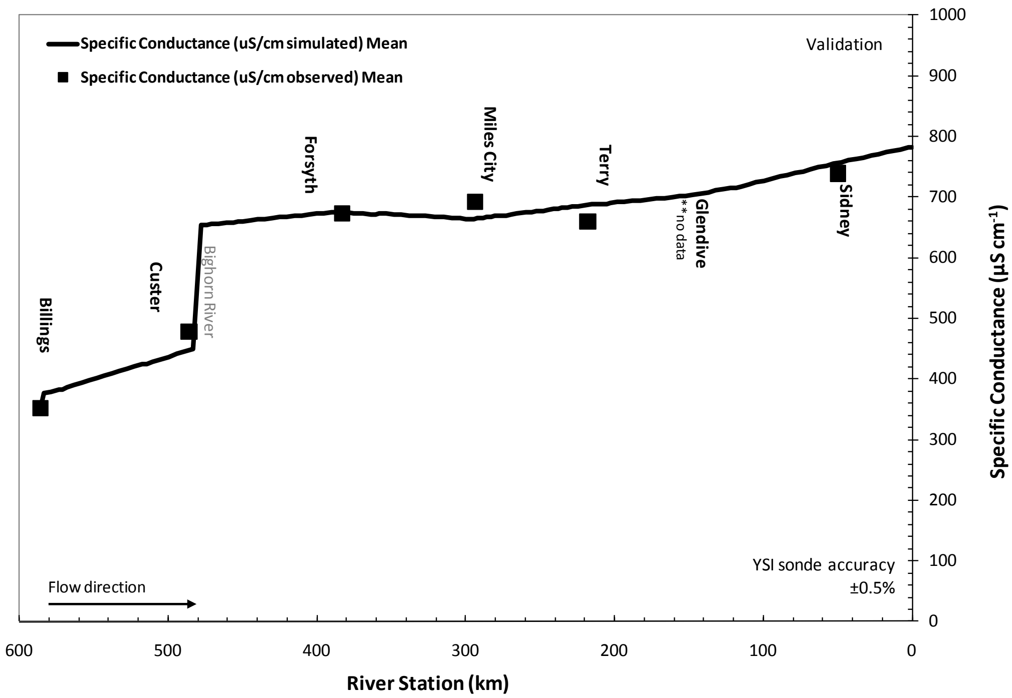

Figure 11-12. Simulated and observed conductivity for the Yellowstone River during 2000.

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).

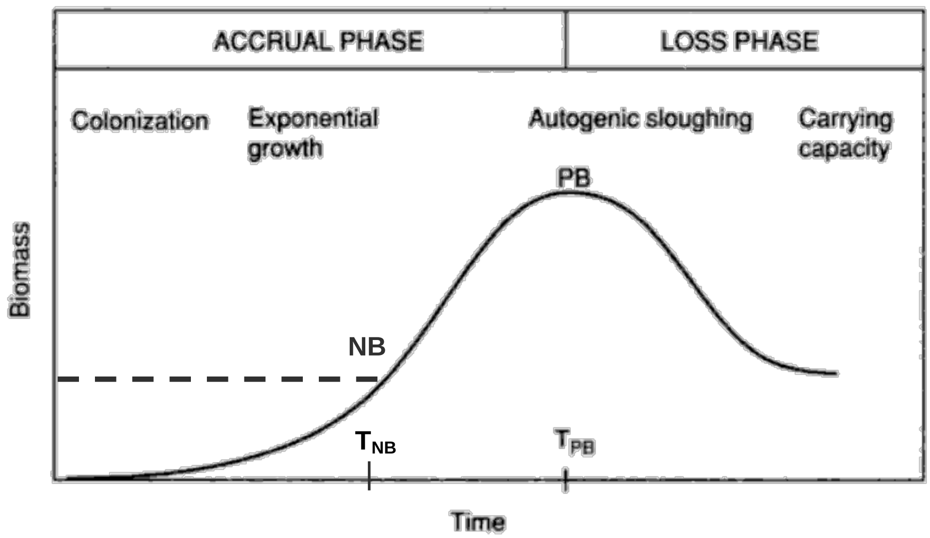

Figure 12-1. Idealized benthic algae growth curve.
Reproduced from Stevenson et al., (1996). Modified to include nuisance conditions. NB = nuisance biomass, TNB = time to nuisance algae , PB=peak biomass, TPB=time to PB from colonization.

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.

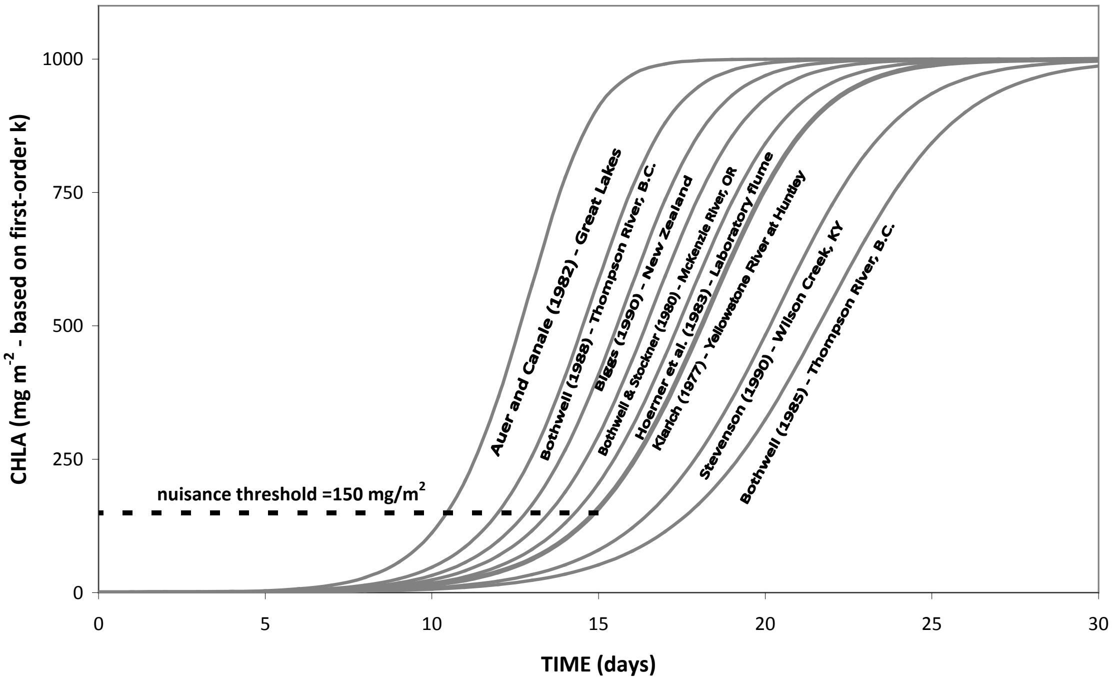

Figure 12-2. Estimated time to nuisance algal biomass under moderately enriched conditions. Each curve was generated using the *k* value reported in **Table 12-2**. To be consistent with the study completed on the Yellowstone river by Klarich (1977)[83], an estimate of 14 days was believed to be an appropriate (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

[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.)

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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 1^{st}, 2^{nd}, 3^{rd}, 4^{th}, 7^{th}, 8^{th}, 9^{th}, and 10^{th} 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.

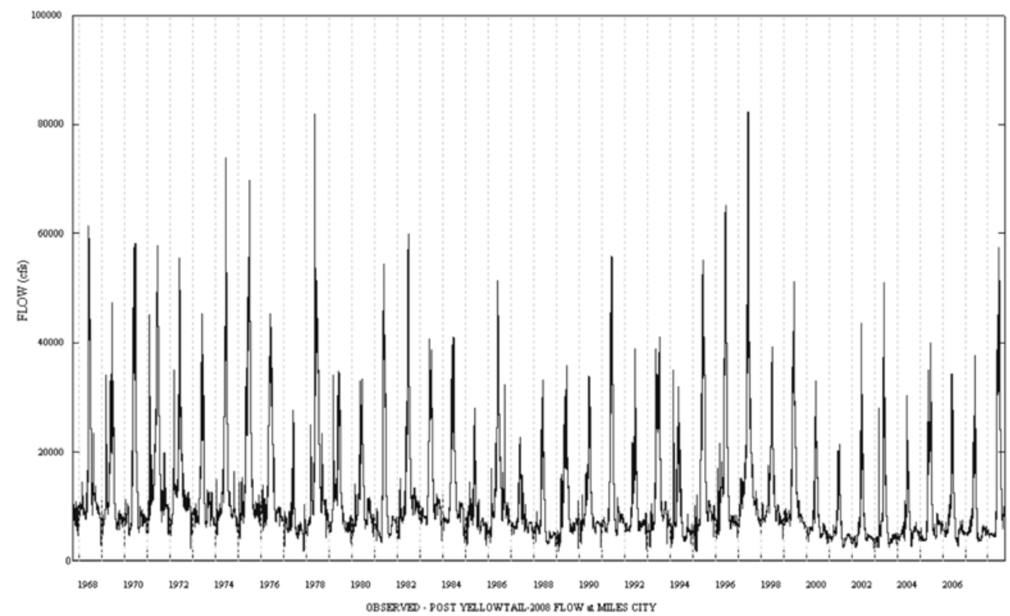

Figure 12-3. Period of record used in low-flow frequency at Miles City.

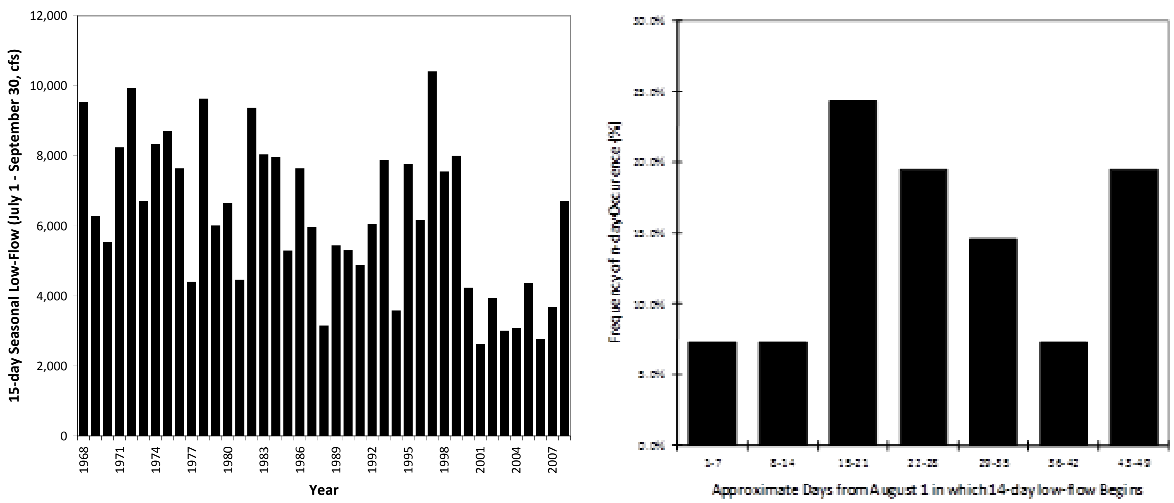

Figure 12-4. Low-flow analysis for Yellowstone River at Miles City (1968-2008).
(Left) Annual 14-day seasonal low-flow data over the period of record at Miles City. (Right) Number of days following August 1 in which the 14-day low flow began at Miles City (originally calculated using 15 days).

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.

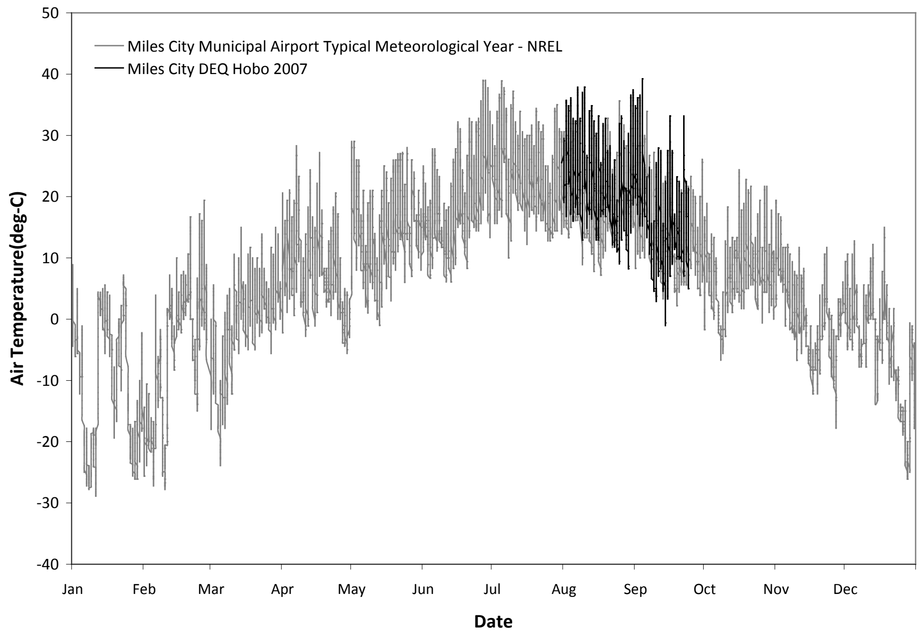

Figure 12-5. Example TMY air temperature plot for 742300 Miles City Municipal Airport.
Field observations from 2007 are shown for reference purposes.

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.

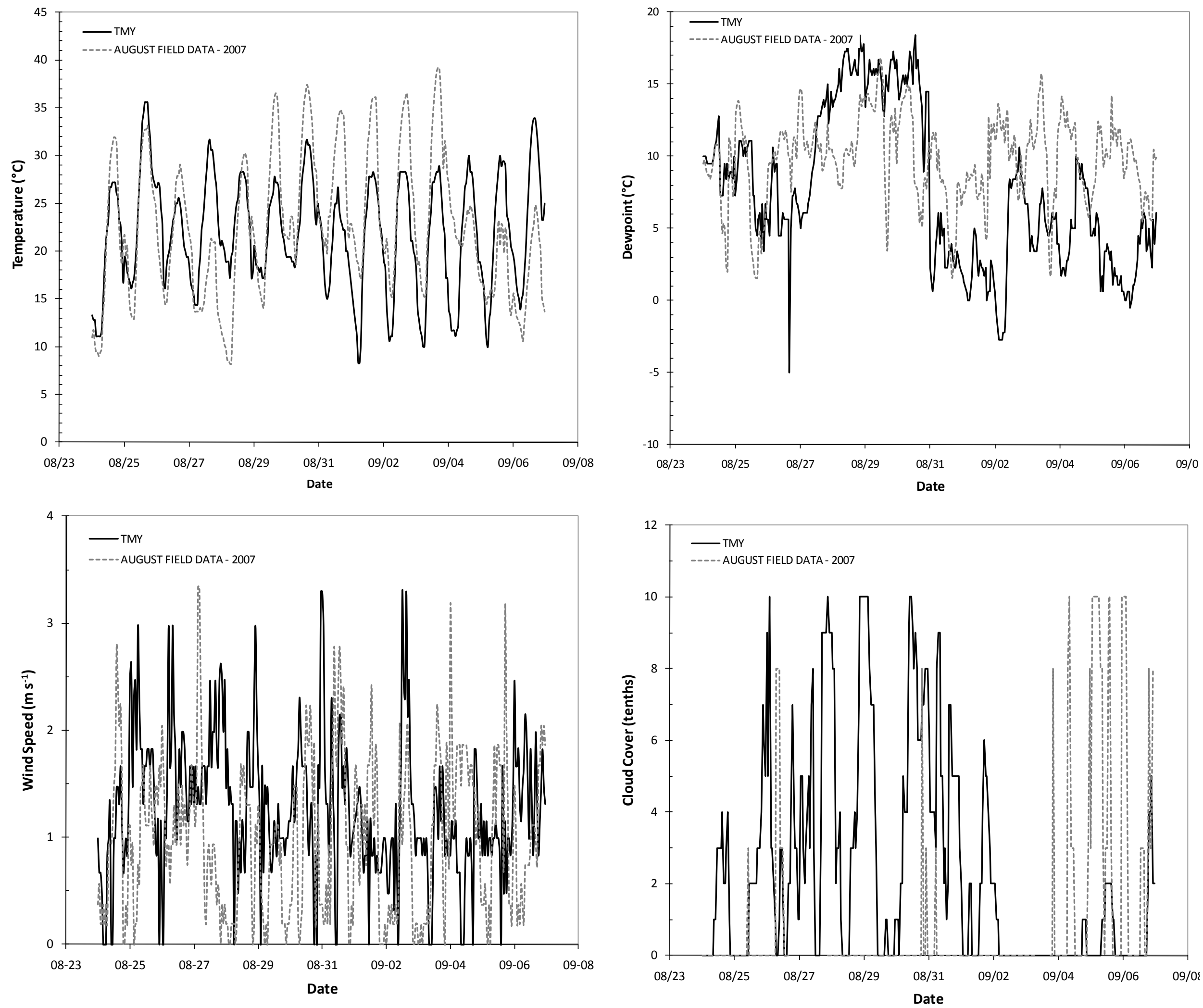

Figure 12-6. Plots of adjusted TMY in comparison with August 2007 field data.
(Top left/right) Air temperature and dew point temperature (˚C). (Bottom left/right) Wind speed (m s^{-1}) and cloud cover (tenths). Adjustments are based on **Table 12-3**.

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.

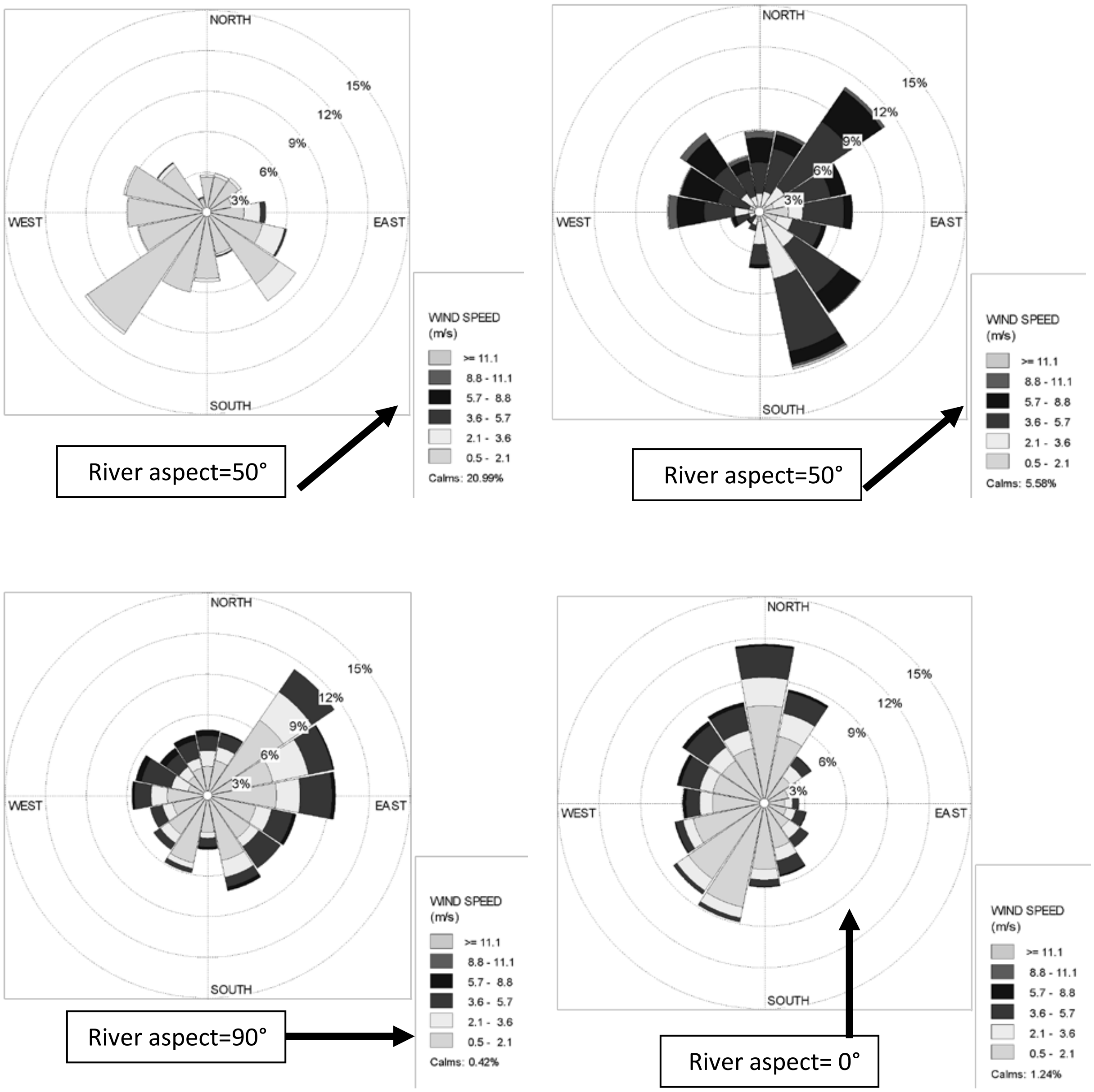

Figure 12-7. Wind Rose data on the lower Yellowstone River.
(Top left/right panel) Wind observations at the DEQ Roche Juan Island and Miles City APT. (Bottom left/right panel) Same but for Terry and Glendive AgriMet. Percent calms increased substantially between the river station and Municipal airport 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 (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 µg 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.

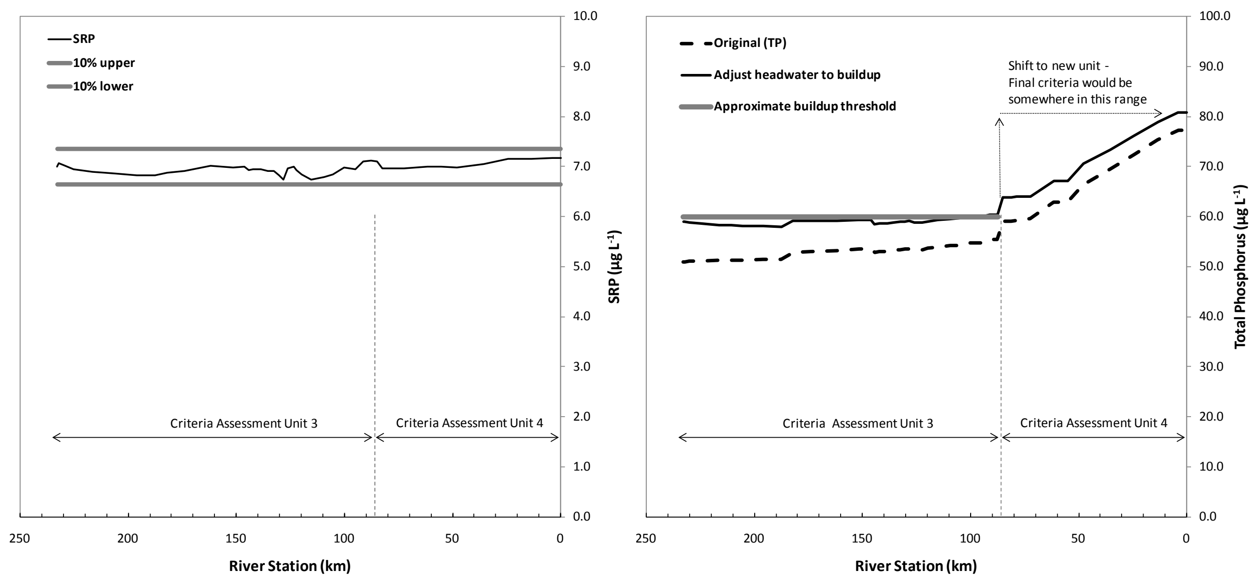

Figure 13-1. Example of the iterative procedure required to assign headwater boundary conditions. (Left panel) Soluble phosphorus simulation shown for an example TP optimization (in this instance a target SRP concentration was 7 µg L^{-1}). (Right panel) Modeled TP with (solid line) and without (dashed 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).

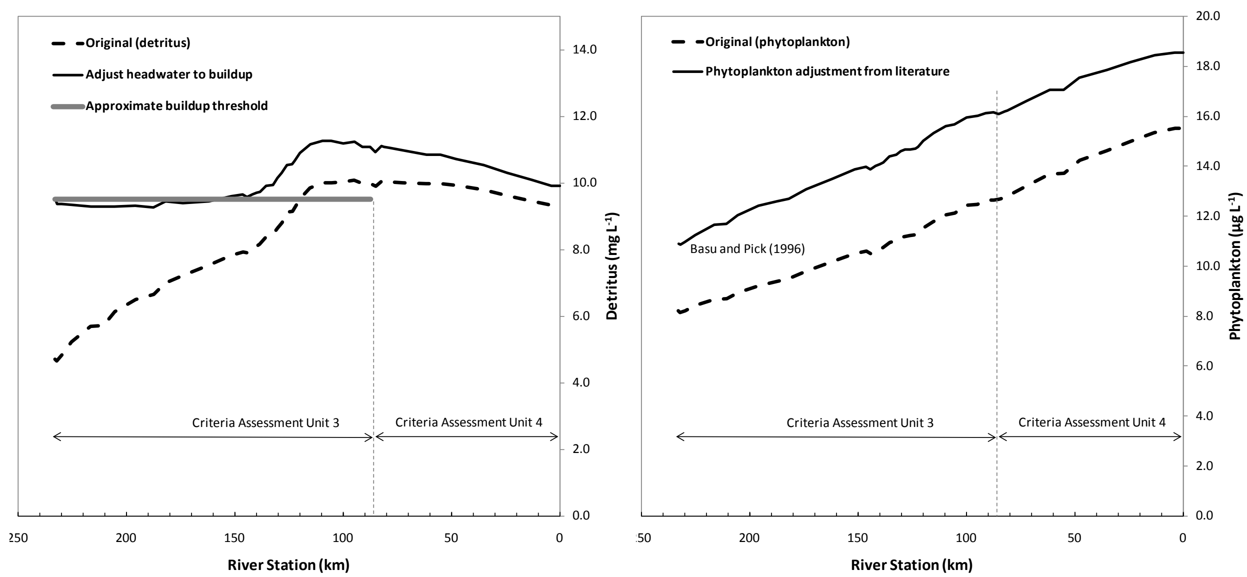

Figure 13-2. Consideration of headwater detritus and phytoplankton concentrations.
(Left panel) Longitudinal plots of detritus with (solid line) and without (dashed line) adjustment for future eutrophied conditions where approximate downstream buildup was used as the guiding factor. (Right panel) Same 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.

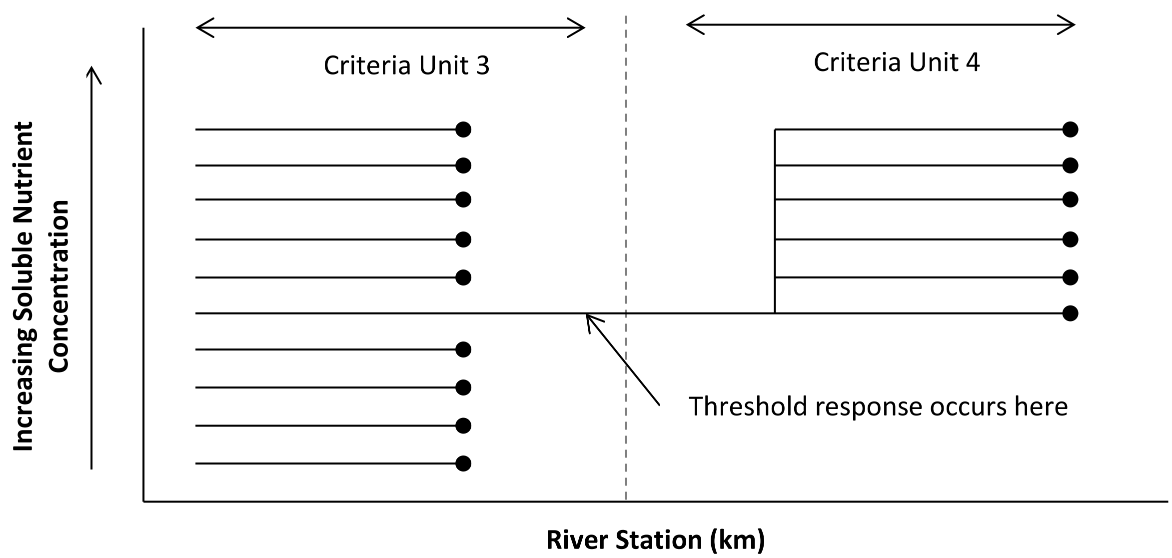

Figure 13-3. Approach toward criteria development.
Soluble nutrients are accreted in the upper river (Criteria Unit 3) until a limiting response is achieved. The threshold then predicates the boundary conditions for Unit 4. Subsequent incremental runs can then be carried out to identify the most limiting 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.

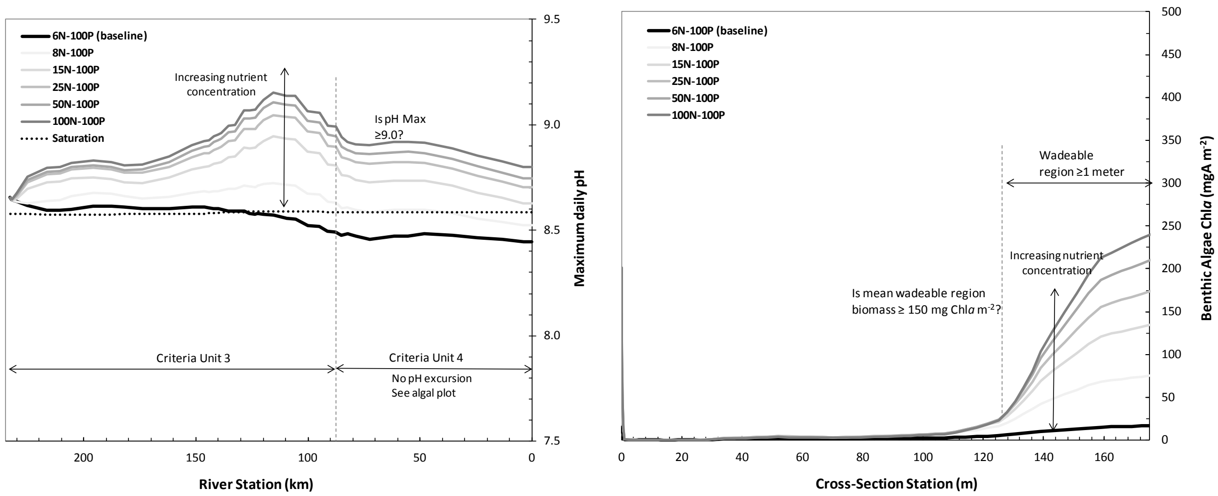

Figure 13-4. Example of a water quality endpoint determination.
(Left panel) Derivation of pH impairment using the differential between baseline and subsequent nutrient addition Q2K runs. (Right panel) Same but for benthic algae, considering the wadeable region in AT2K.

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.

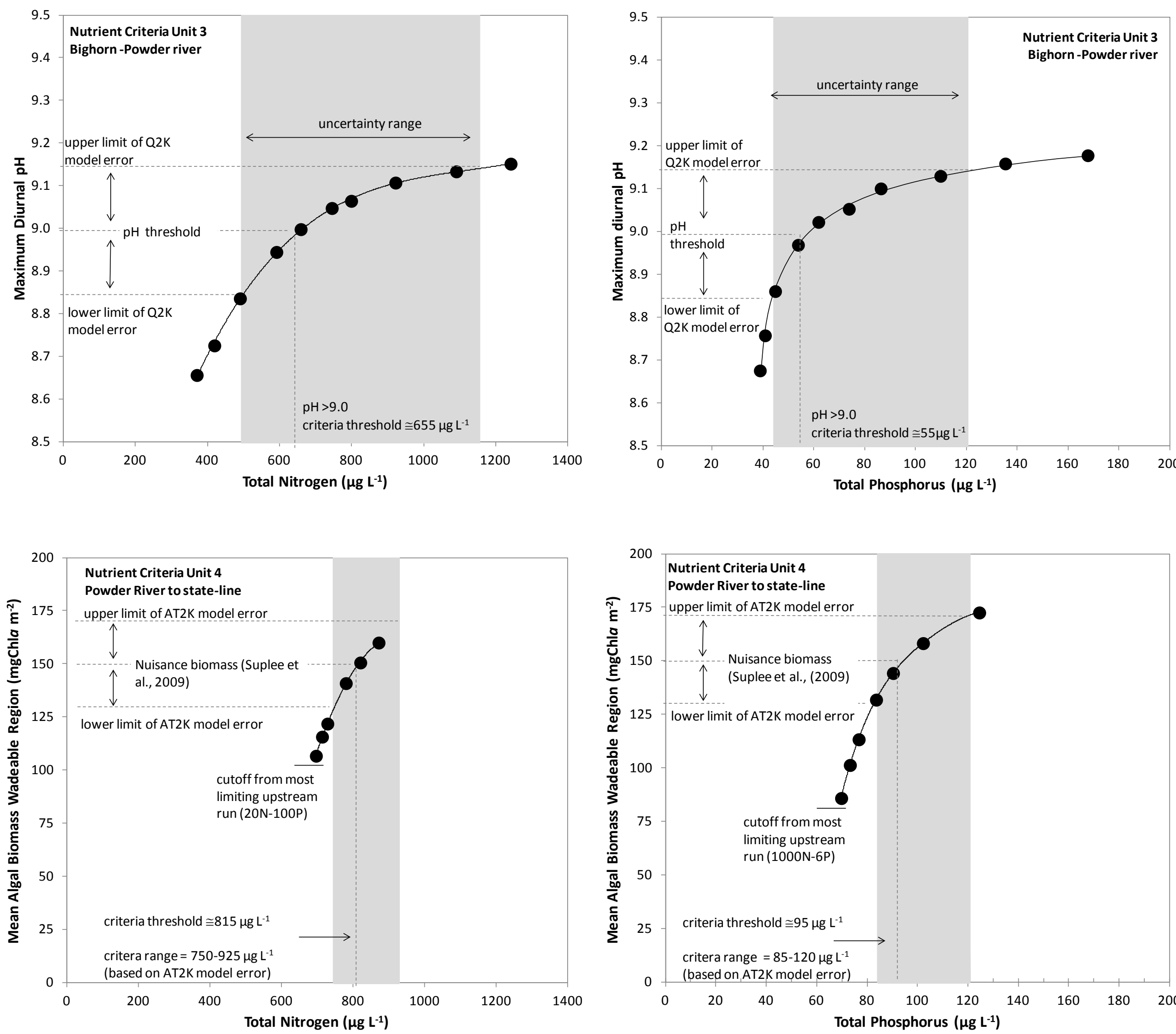

Figure 13-5. Non-linear response relationship between nutrients, pH, and benthic algae.
(Top left/right panel) Most limiting pH response associated with TN and TP concentrations in Criteria Assessment Unit 3 (maximum diurnal pH), which was the basis for establishing the criterion. Criteria are recommended 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 ($\mu g\ L^{-1}$)	Total Phosphorus ($\mu g\ L^{-1}$)
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.

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[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,

[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 5^{th} 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.

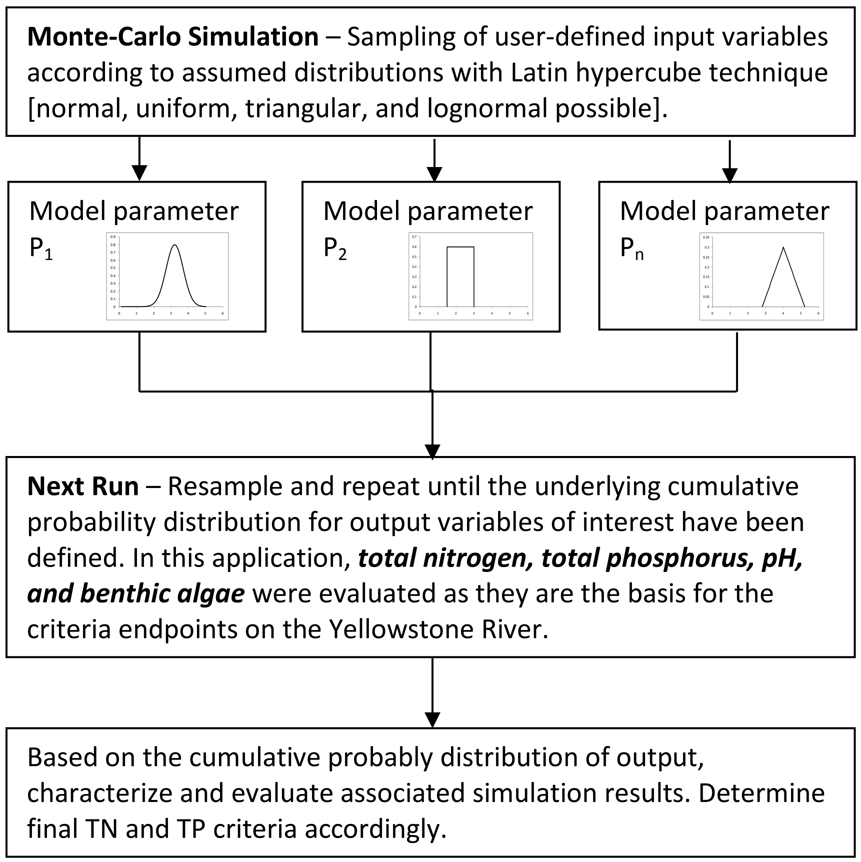

Figure 14-1. Example of Monte Carlo simulation procedure for Yellowstone River.

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).

<table>
<tr><th>Parameter</th><th>Units</th><th>Min[1]</th><th>Avg[2]</th><th>Max[1]</th><th>Distribution</th><th>Coefficient of Variation (COV) (%)</th><th>Literature range[3]</th></tr>
<tr><td>Flow</td><td>$m^3 s^{-1}$</td><td>0</td><td>---</td><td>7.439</td><td>n/a</td><td>0[4]</td><td>n/a</td></tr>
<tr><td>Temperature</td><td>°C</td><td>10</td><td>---</td><td>29</td><td>normal</td><td>10</td><td>1-8</td></tr>
<tr><td>Conductivity</td><td>$\mu S\ cm^{-1}$</td><td>428</td><td>---</td><td>3,920</td><td>lognormal</td><td>5</td><td>1-15</td></tr>
<tr><td>Inorganic solids[5]</td><td>$mgD\ L^{-1}$</td><td>1.0</td><td>---</td><td>20,610</td><td>lognormal</td><td>35</td><td>n/a</td></tr>
<tr><td>Dissolved oxygen</td><td>$mgO_2\ L^{-1}$</td><td>3.9</td><td>---</td><td>17.1</td><td>normal</td><td>25</td><td>2-15</td></tr>
<tr><td>CBODfast</td><td>$mgO_2\ L^{-1}$</td><td>2.8</td><td>---</td><td>21.6</td><td>lognormal</td><td>60</td><td>5-40</td></tr>
<tr><td>Organic-N</td><td>$\mu g\ L^{-1}$</td><td rowspan="8" colspan="6">not evaluated since already perturbated and considered as part of nutrient addition scenarios detailed in Section 13.0.</td></tr>
<tr><td>Ammonia-N</td><td>$\mu g\ L^{-1}$</td></tr>
<tr><td>Nitrate-N</td><td>$\mu g\ L^{-1}$</td></tr>
<tr><td>Organic-P</td><td>$\mu g\ L^{-1}$</td></tr>
<tr><td>Dissolved-P</td><td>$\mu g\ L^{-1}$</td></tr>
<tr><td>Phytoplankton</td><td>$\mu g\ L^{-1}$</td></tr>
<tr><td>Internal-N</td><td>$mgN\ mgA^{-1}$</td></tr>
<tr><td>Internal-P</td><td>$\mu g\ L^{-1}$</td></tr>
<tr><td>Detritus[6]</td><td>$mgD\ L^{-1}$</td><td>0.1</td><td>---</td><td>2,160</td><td>lognormal</td><td>35</td><td>n/a</td></tr>
<tr><td>Alkalinity</td><td>$mgCaCO_3\ L^{-1}$</td><td>142</td><td></td><td>461</td><td>normal</td><td>35</td><td>n/a</td></tr>
<tr><td>pH</td><td>pH units</td><td>6.8</td><td>---</td><td>8.7</td><td>normal</td><td>5</td><td>n/a</td></tr>
</table>

[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].

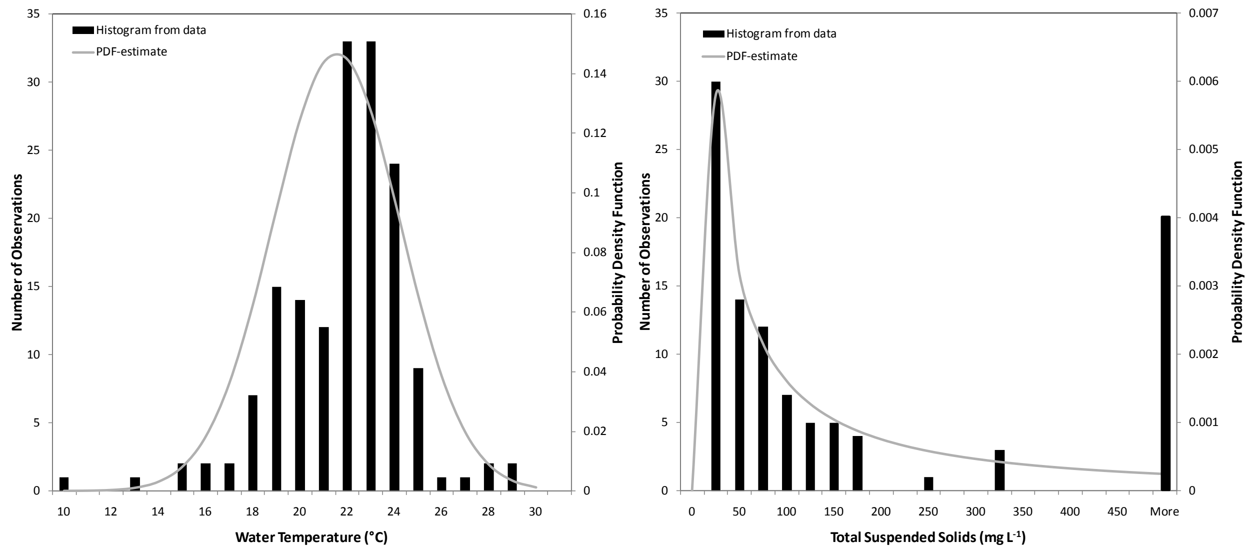

Figure 14-2. Example point load PDF used in Yellowstone River Monte Carlo Analysis.
(Left panel) Normal PDF for water temperature point load to the Yellowstone River. (Right panel) Same but lognormal distribution (natural logarithm) for total suspended solids.

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.

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[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**.

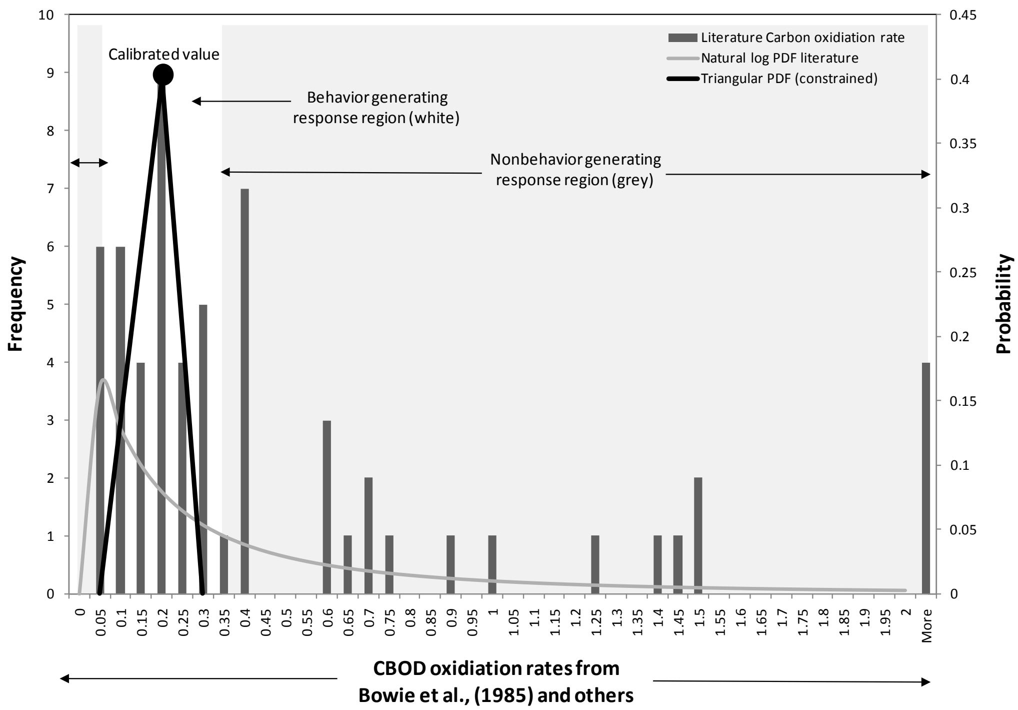

Figure 14-3. Example PDF assignment for model rate coefficients on the Yellowstone River.
The $CBOD_{fast}$ oxidation rate illustrates how the parameter space (and distribution) was first defined using the literature and was then refined using Bayesian principles/*posteriori* analysis of model runs to 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.

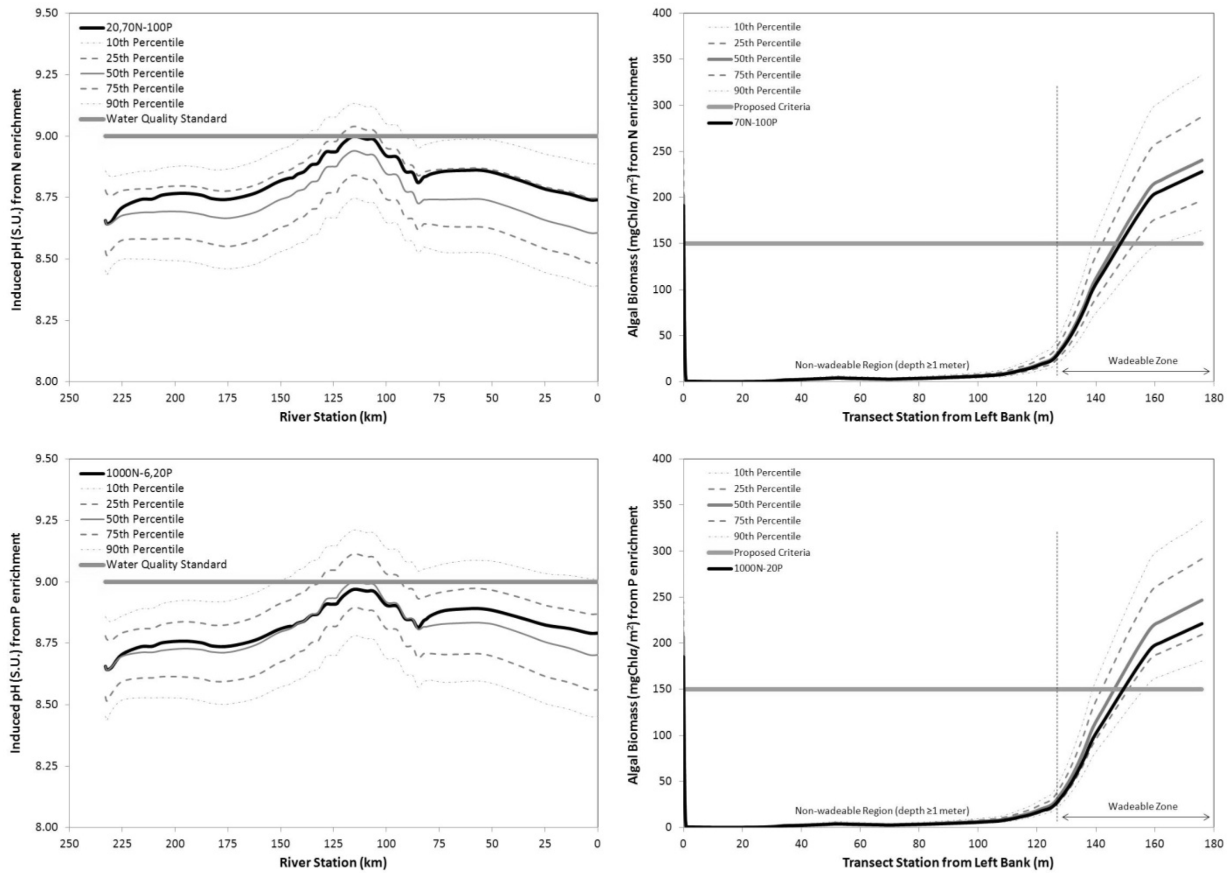

Figure 14-4. Estimated model output variance for ecological response endpoints.
(Top left/right panel). Uncertainty associated with pH and benthic algae predictions from the Q2K and AT2K models for the nitrogen enrichment scenario. (Bottom left/right panel). Same but for phosphorus enrichment. The 10^{th}, 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.

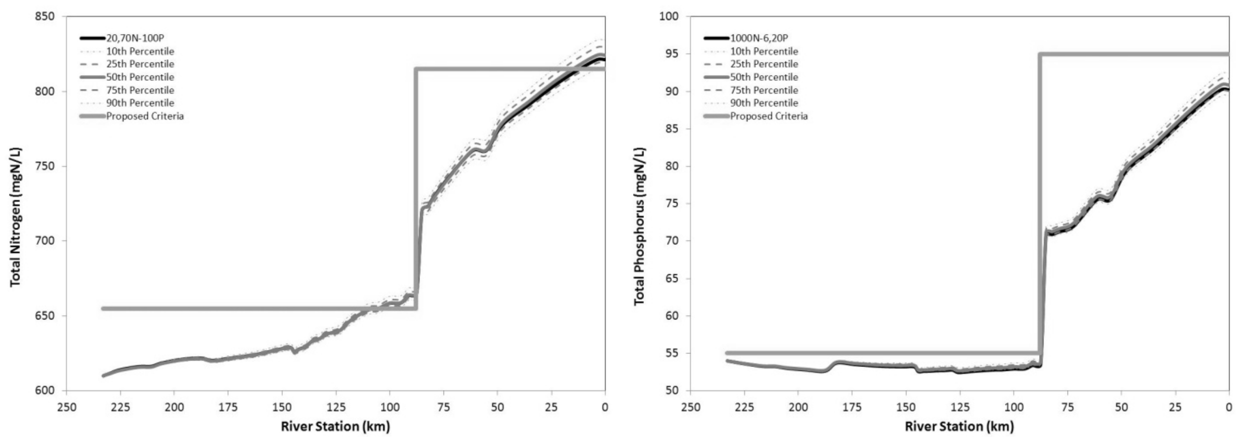

Figure 14-5. Estimated model output variance around computed nutrient criteria.
(Left panel). Output variance for TN according to the uncertainty runs detailed previously. (Right panel). Same but for TP. The changing variance in total nutrients is related to the decision did not perturbate nutrient loads 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.

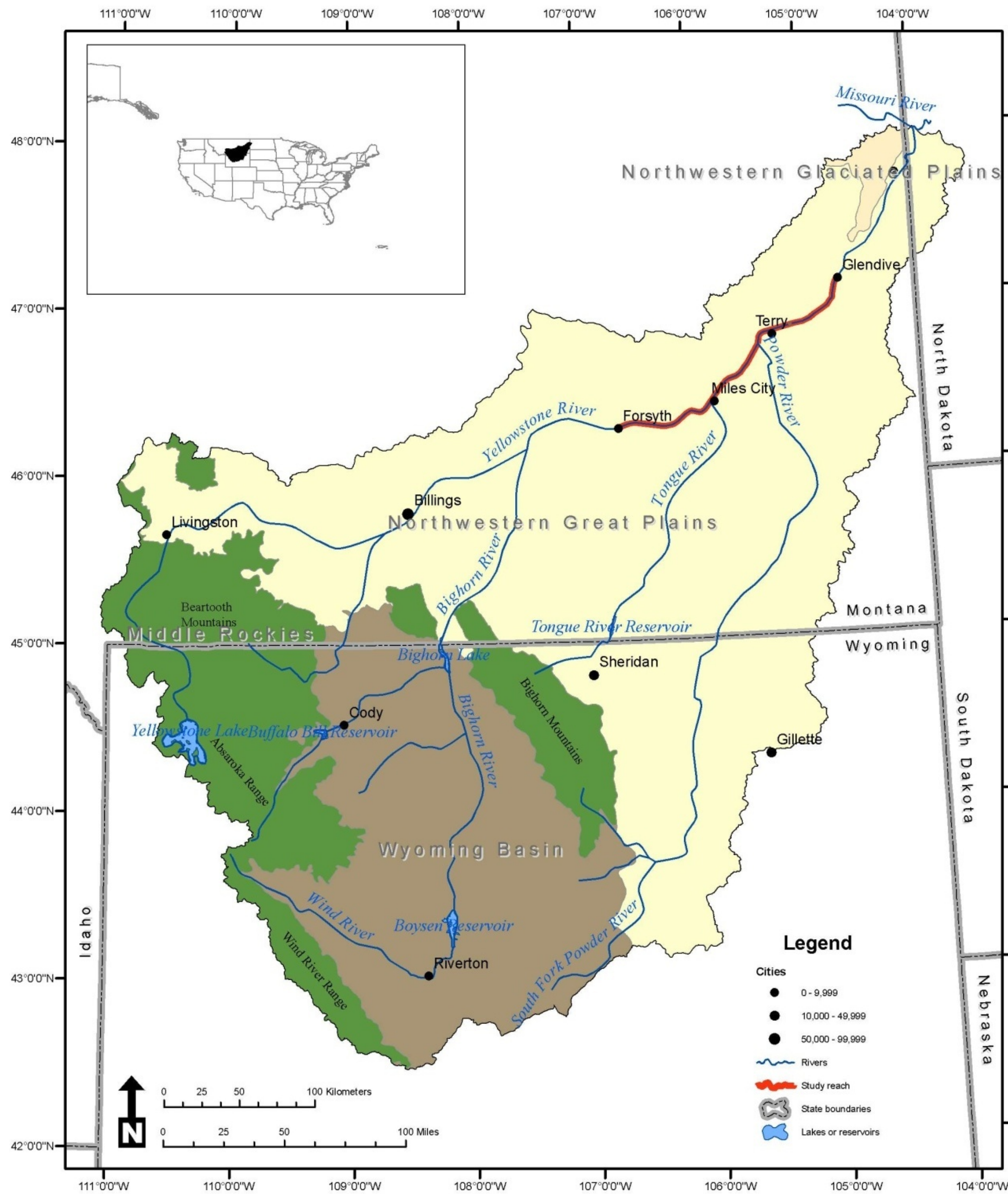

Figure 15-1. Level III ecoregions in relation to water quality criteria recommendations.
The study reach is located in the center of the Northwestern Great Plains ecoregion. However, much of the water flowing into the river originates from the Middle Rockies and Wyoming Basin and ecoregion. Hence it 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).

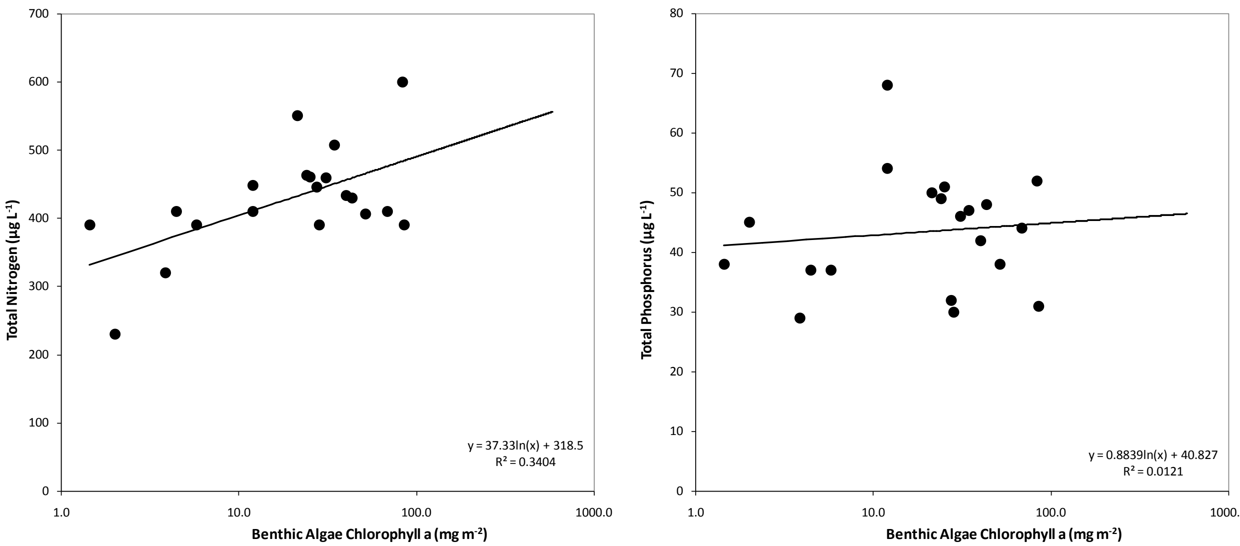

Figure 15-2. Relationship between nutrients and benthic biomass on the lower Yellowstone River. (Left panel) Relationship between TN and benthic algae. (Right panel) Same but for TP. Data shown for the months of August and September (data from Forsyth to Sidney, MT).

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 (μg L^{-1})	TP Outcome (μg 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.

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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:

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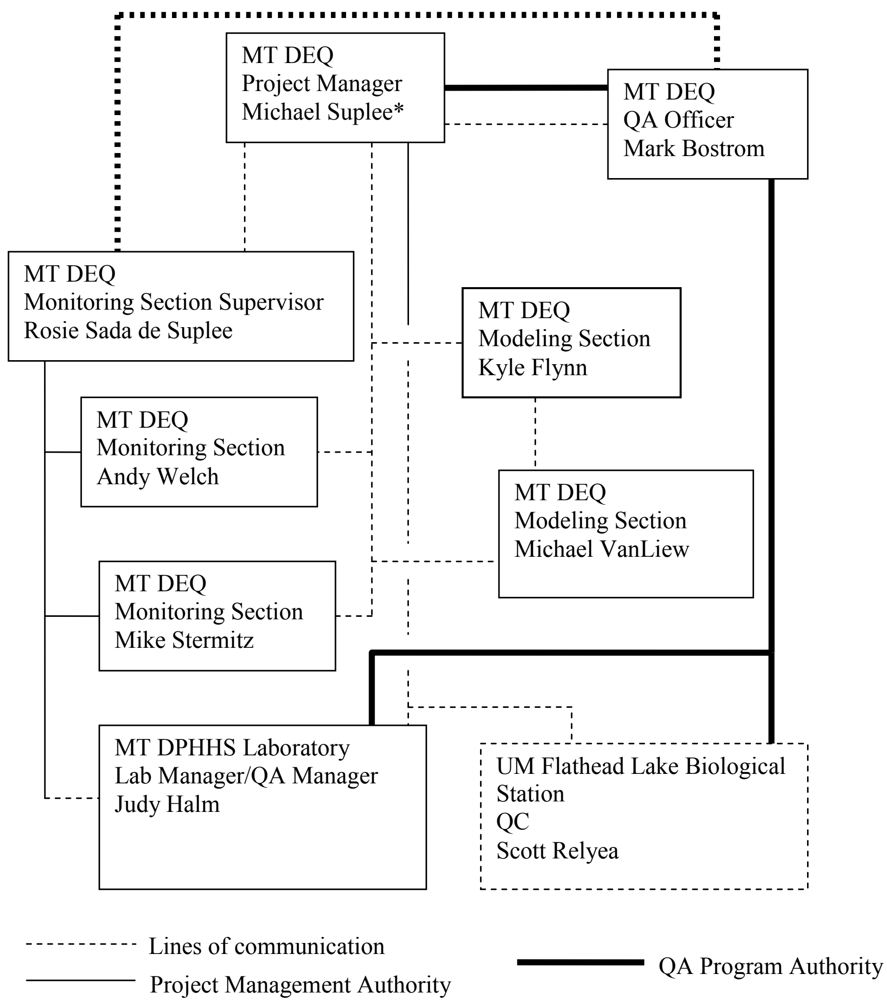
APPENDICES

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.).

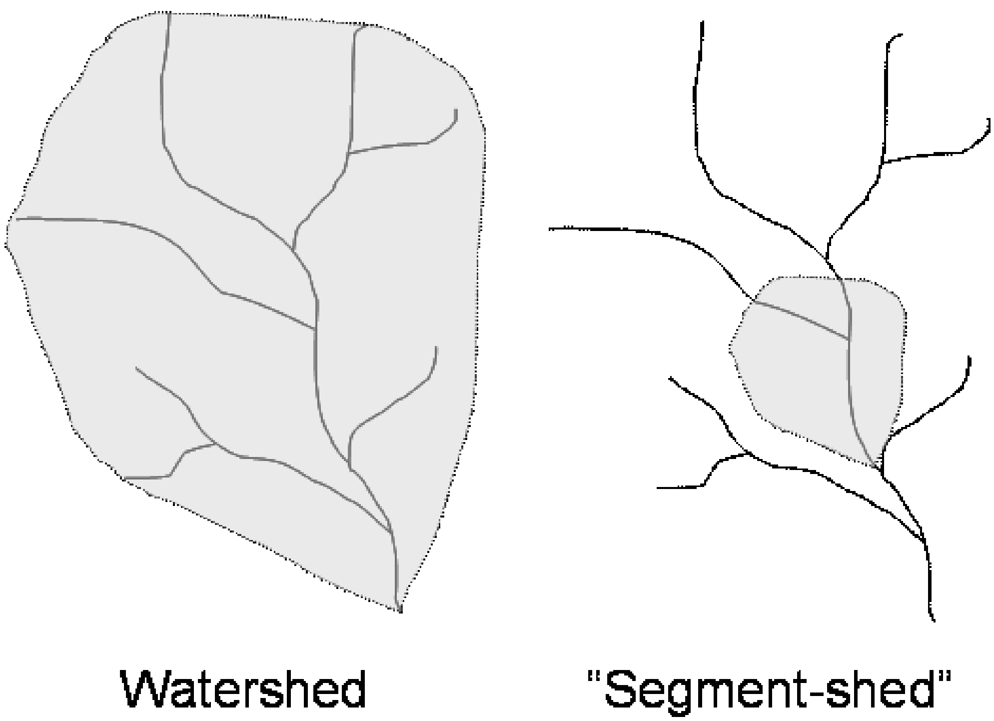


Figure 1 – Conceptual diagram illustrating the watershed versus segment-shed ideas. The segment-shed is recommended for considering the area contributing to land cover/land use information above a large river site.

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.

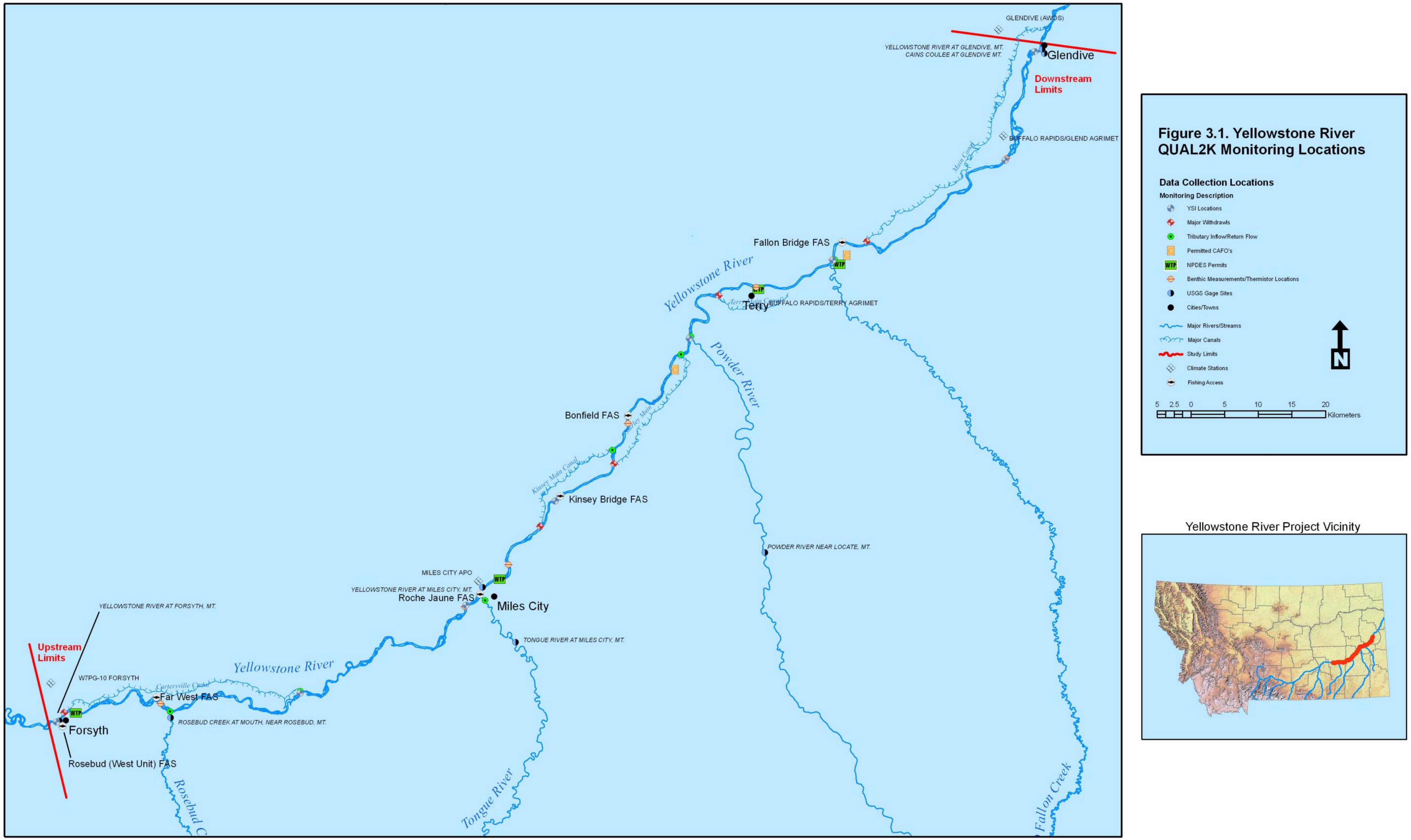


Figure 3.1. Yellowstone River QUAL2K Monitoring Locations

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).

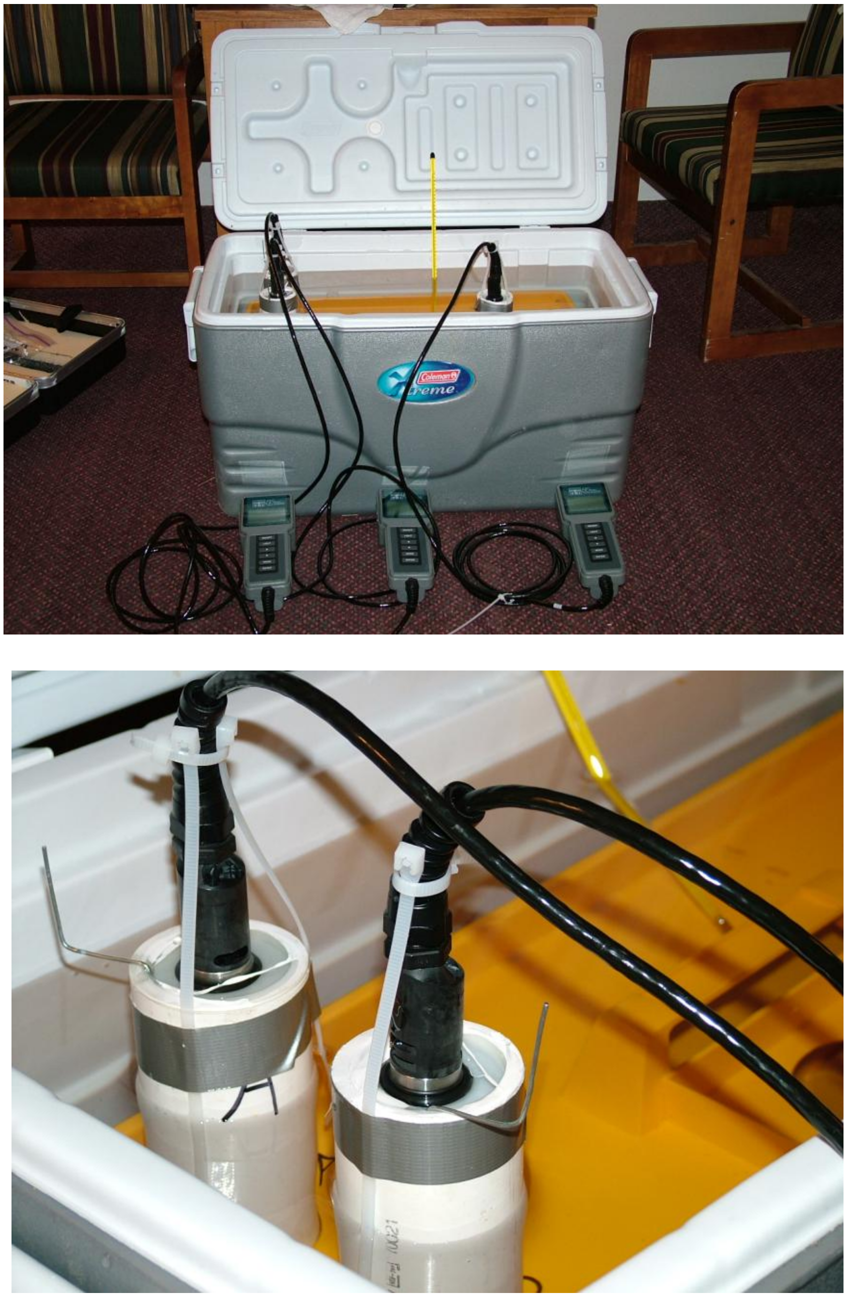


Figure 3.2. Measurement of sediment oxygen demand in sediment core samples, Yellowstone River, August 2006. A. Paired sediment cores in their water bath, with YSI model 85 DO meters attached. The tube on the right only contained river water and was used to measure BOD. B. Close-up of the sealed sediment 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.

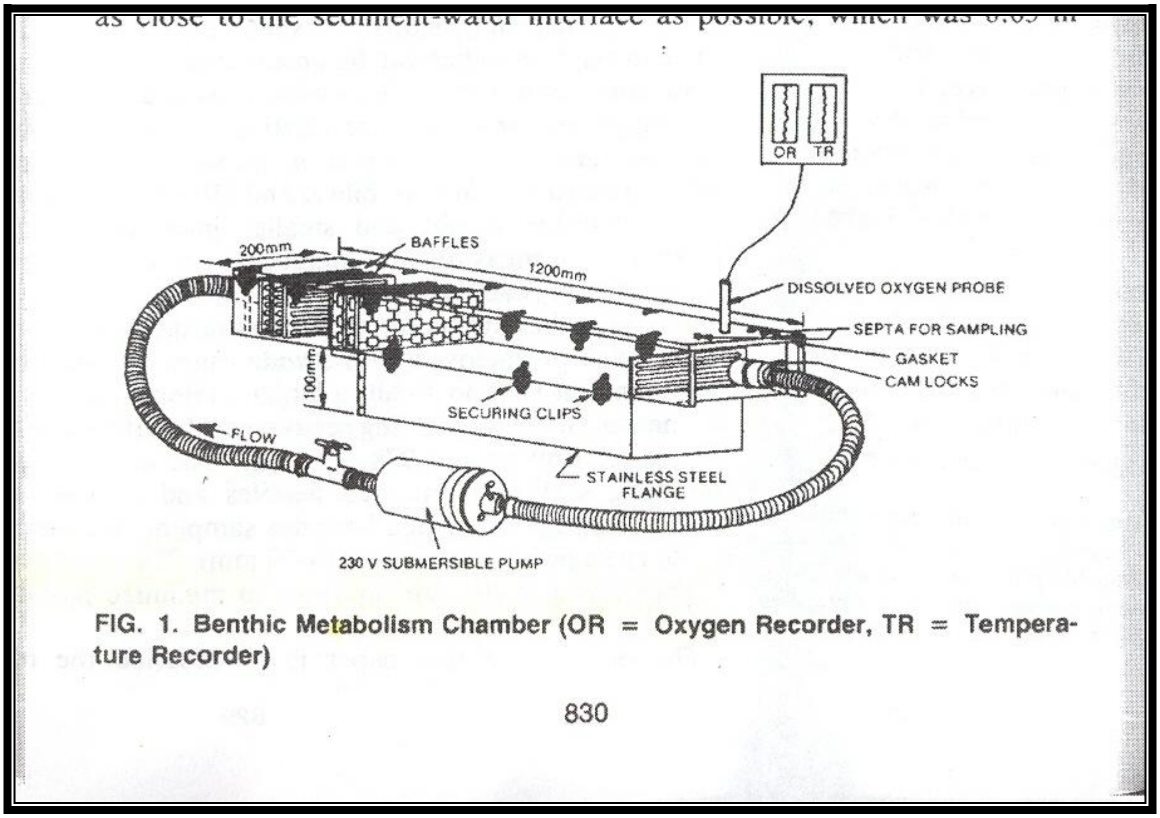


Figure 3.3. General diagram of the flow-adjustable SOD chamber proposed for use in the project, from Hickey (1988). The final design will be a modification of this basic layout. For example, a flexible skirt will be added around the circumference of the chamber to assure a good seal to the river bottom 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.

$$\text{Minimize } Z=\sum_{i=1}^{i_n}[OBS_i - PRED_i]^2 \tag{2}$$

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:

$$\text{Normalized Sensitivity Coefficient (NSC)} = \frac{\Delta Y_o / Y_o}{\Delta X_I / X_I} \tag{3}$$

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
Ammonia (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:

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Prepared By:

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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
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Bob Bukantis (WQ Standards Section Supervisor) Date

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Michael Pipp (Data Management Section Supervisor) Date

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Rosie Sada (WQ Monitoring Section Supervisor) Date

____________________________ ______________
Mark Bostrom (QA Officer) Date

TABLE OF CONTENTS

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.**

USING A COMPUTER WATER-QUALITY MODEL TO DERIVE NUMERIC NUTRIENT CRITERIA FOR A SEGMENT OF THE YELLOWSTONE RIVER

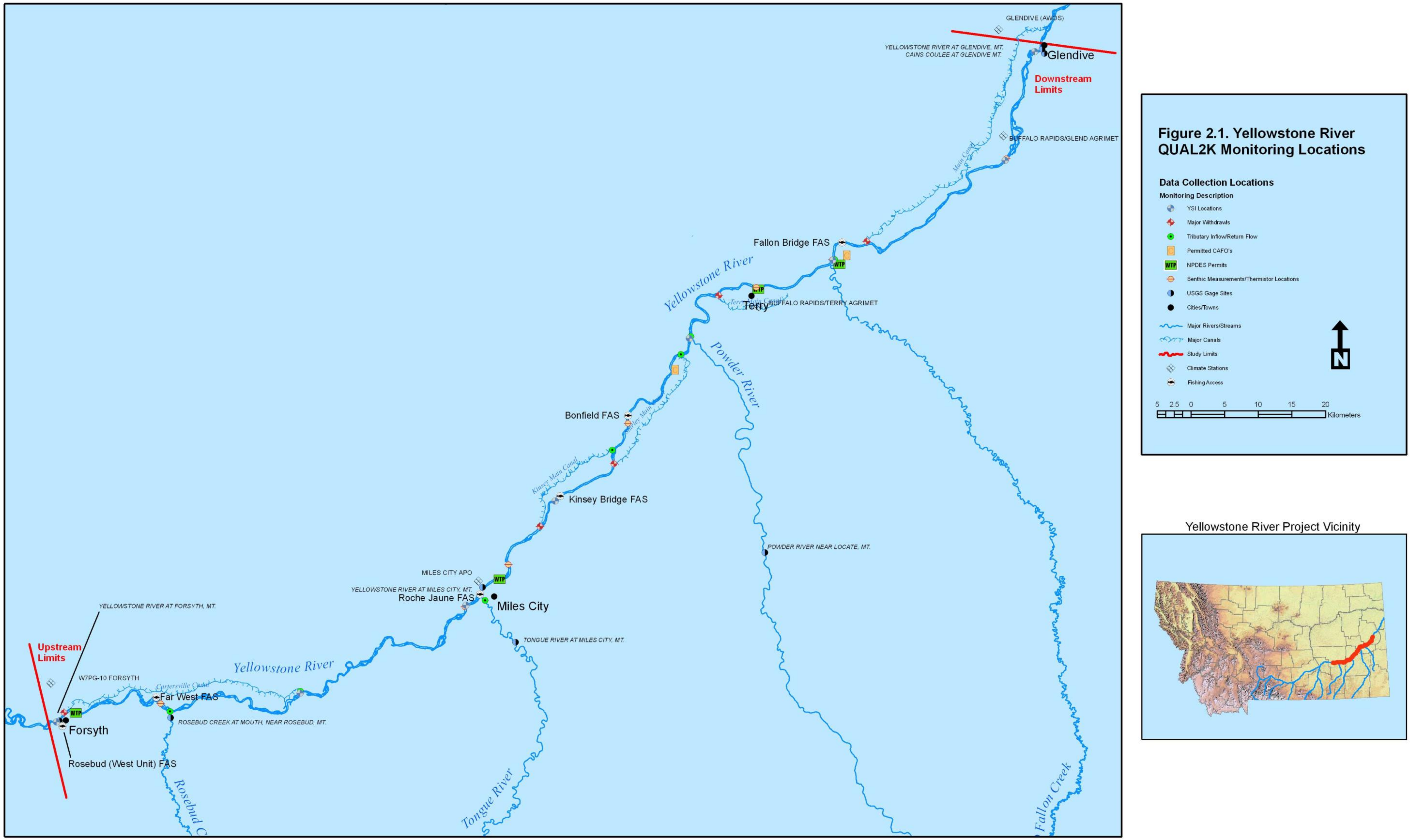

Figure 2.1. Yellowstone River QUAL2K Monitoring Locations

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:

$$\mathrm{SOD} = \frac{aV - bV}{S} \qquad (1)$$

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.

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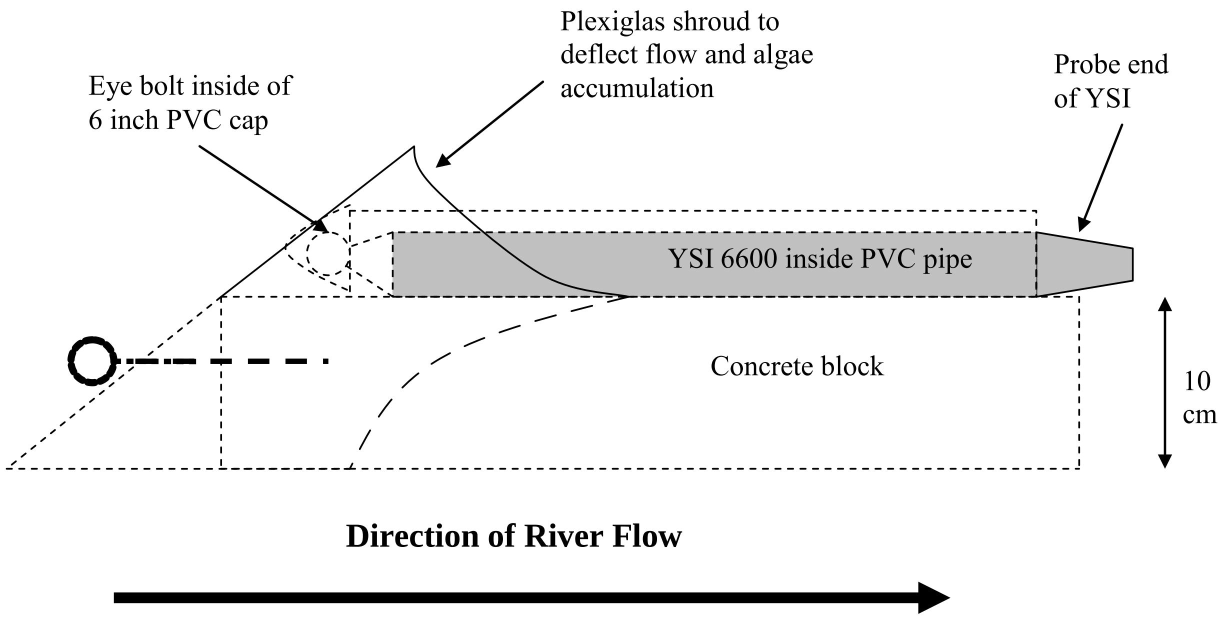

Fig. 3.1. Profile view of the YSI 6600EDS sonde deployment system.

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.

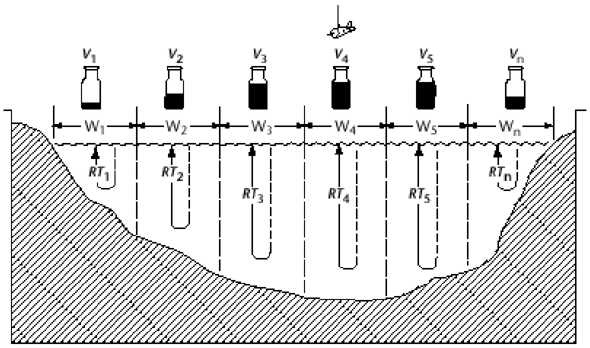

Figure 3.2. Equal Width Increment (EWI) Schematic.

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.
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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

11.0 References

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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)* :

$$\text{mg } O_2/L = \frac{(\text{ml titrant})(\text{molarity of thiosulfate})(8000)}{(\text{ml sample titrated})\ (\text{ml of BOD bottle-2 / ml of BOD bottle})}$$

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:

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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	

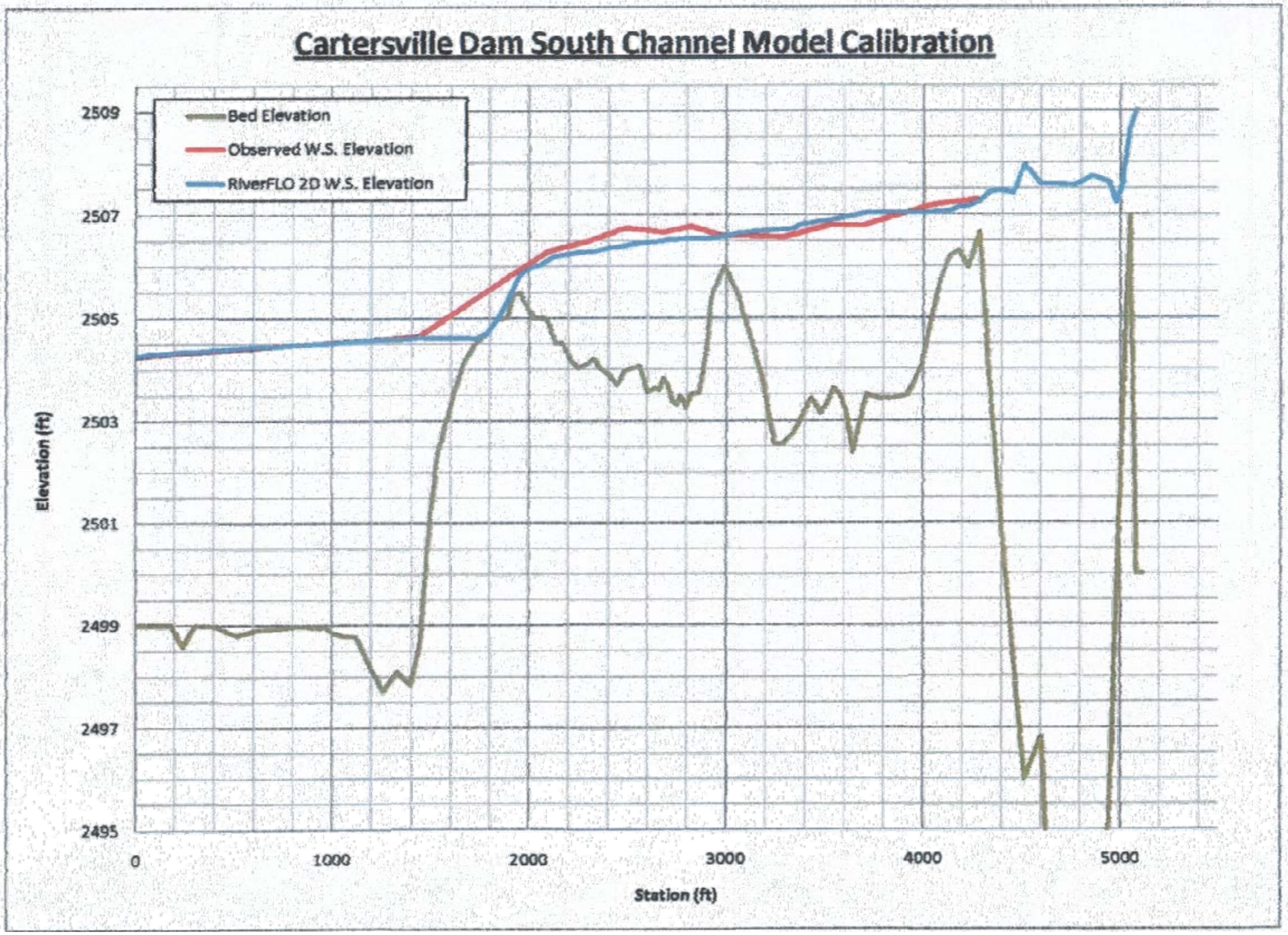
Total Discharge = **0.319** **ft^3/sec**

TOTAL DISCHARGE

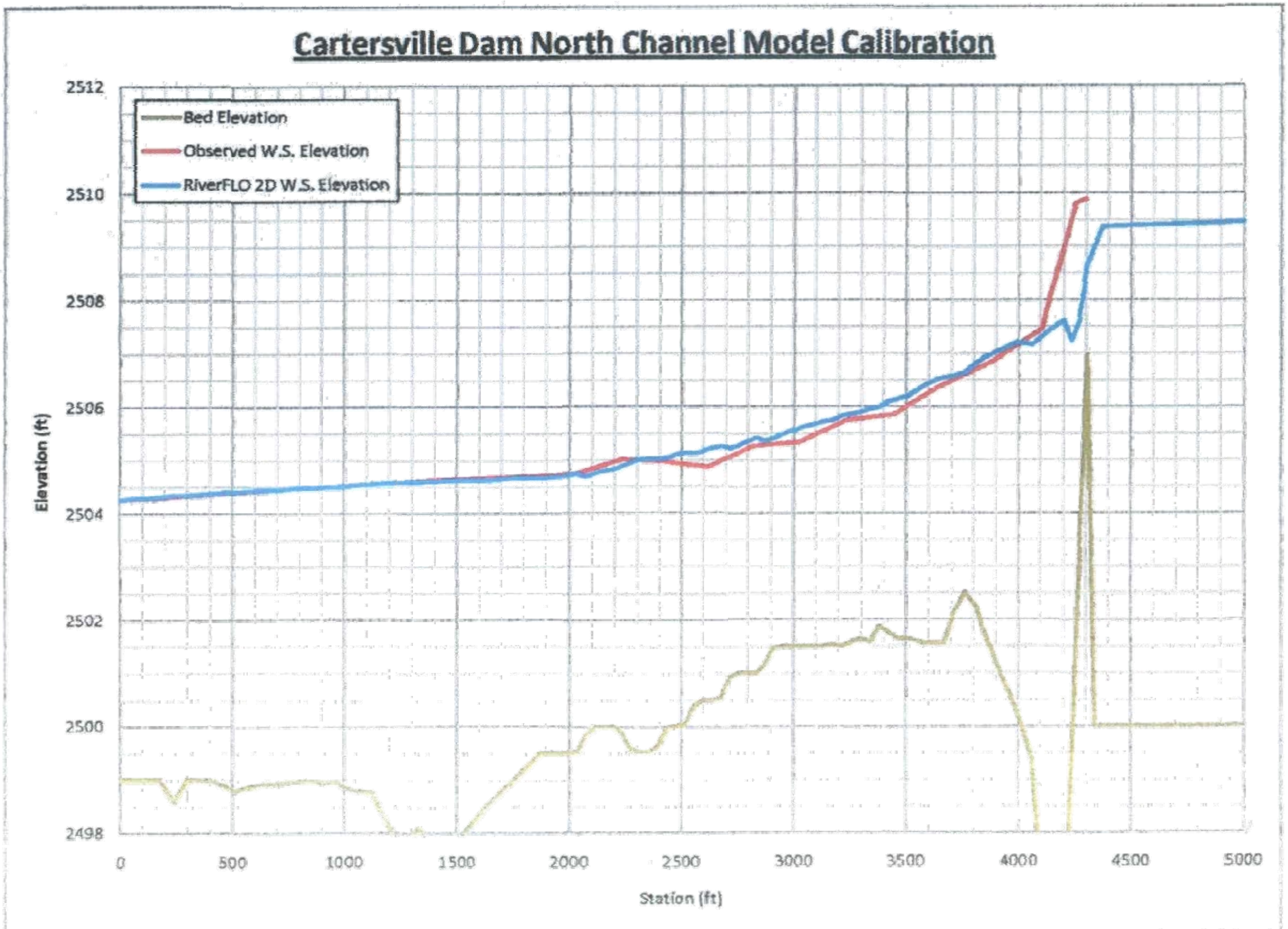
Date:	9/16/2007	**Site Visit Code:**	**Y0376**
Waterbody:	Shirley Main Canal	**Station ID:**	**Y17SRYMC01**
Personnel:	R. Sada, M. Suplee		

	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)
Station (ft)
2512
2510
2508
2506
2504
2502
2500
2498
0
500
1000
1500
2000
2500
3000
3500
4000
4500
5000

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

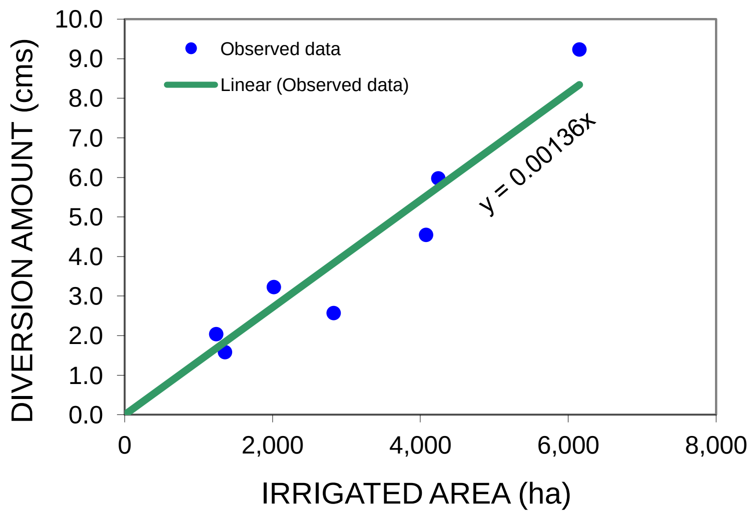
[3]From Kimberly-Penman AgriMet calculations, multiplied by irrigated area

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

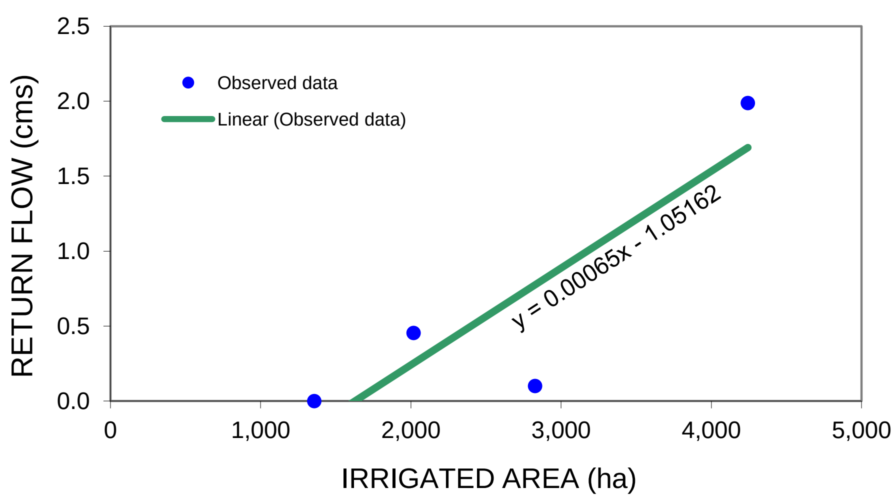
[2]Estimated using regression of irrigated area and measured diversion data

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

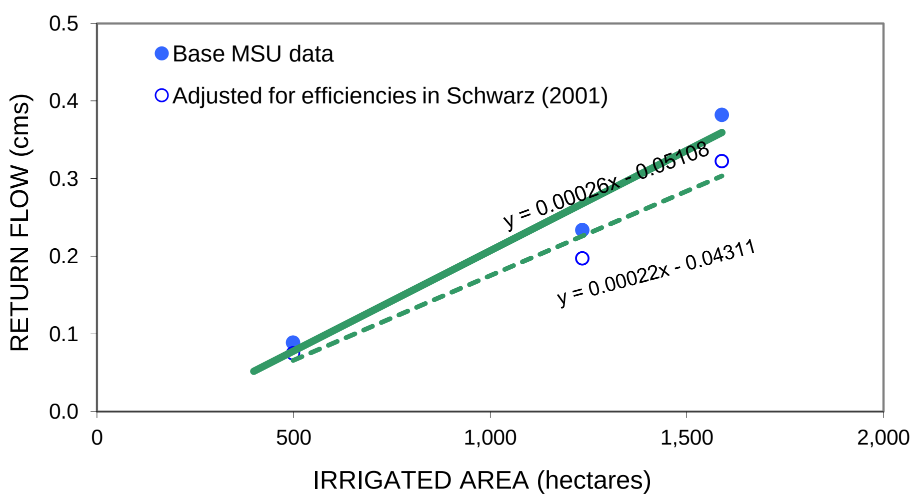
[2]Estimated using regression of irrigated area and measured return flow

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

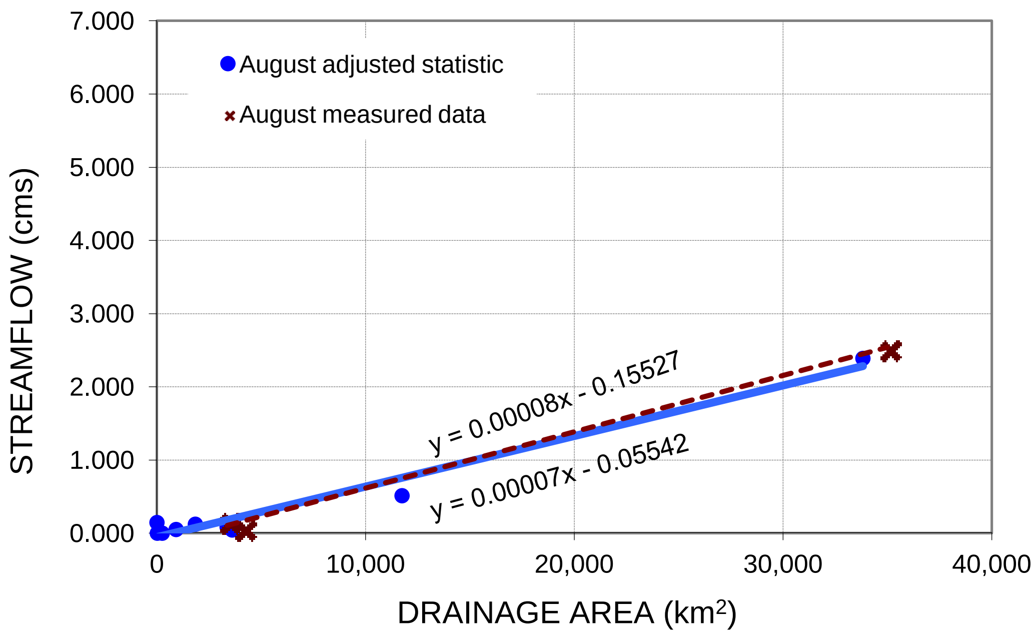
Description	Gage ID	Area [1] (mi^2)	Area (km^2)	
Yellowstone River nr Terry MT	6326530	63,447	164,257	
Yellowstone River at Glendive MT	6327500	66,788	172,906	
		3,341	8,649	
Field data representing remaining area(s)				
O'fallon Creek at mouth (1:24,000 HUC file)	NA	1,578	4,085	
Unmonitored Drainage 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 (miles²)	Drainage Area (km²)	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



[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

[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			Above T&Y			T&Y-Miles	Actual Diversion
8/17/2005	166	A	8/17/2005	336	A	170.0	166.5
8/18/2005	179	A	8/18/2005	330	A	151.0	166.5
8/19/2005	203	A	8/19/2005	328	A	125.0	154.8
8/20/2005	212	A	8/20/2005	324	A	112.0	152.8
8/21/2005	213	A	8/21/2005	321	A	108.0	152.6
8/22/2005	189	A	8/22/2005	305	A	116.0	158.9
8/23/2005	207	A	8/23/2005	297	A	90.0	163.0
8/24/2005	157	A	8/24/2005	289	A	132.0	164.5
8/25/2005	144	A	8/25/2005	283	A	139.0	163.5
8/26/2005	139	A	8/26/2005	279	A	140.0	162.5
AVG	180.9		AVG	309.2		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)							
	Clear Cr	Clear Cr	Clear Cr	Whoopup	Whoopup	Sand	Sand	Sand
Date	#1	#2	#3	Cr #1	Cr #2	Cr	Cr #2	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

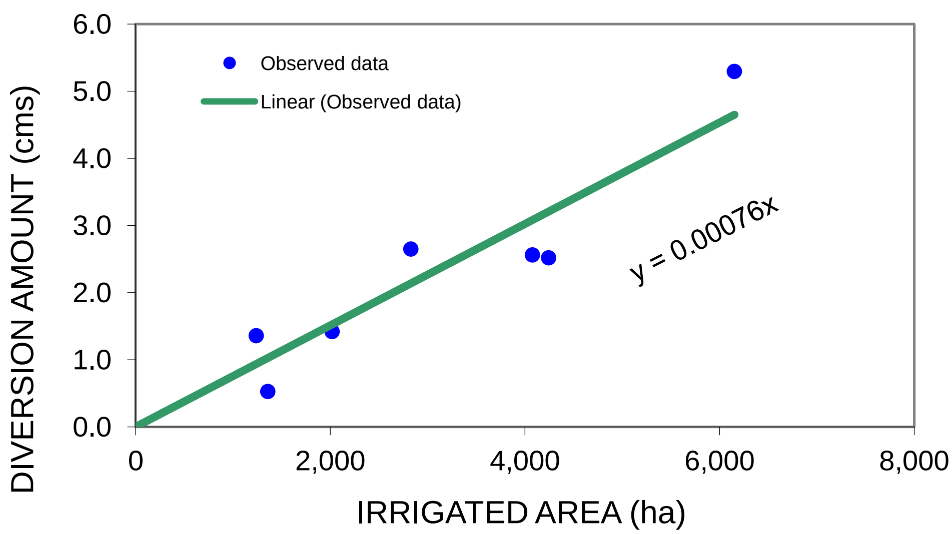
[3]From Kimberly-Penman AgriMet calculations, multiplied by irrigated area

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

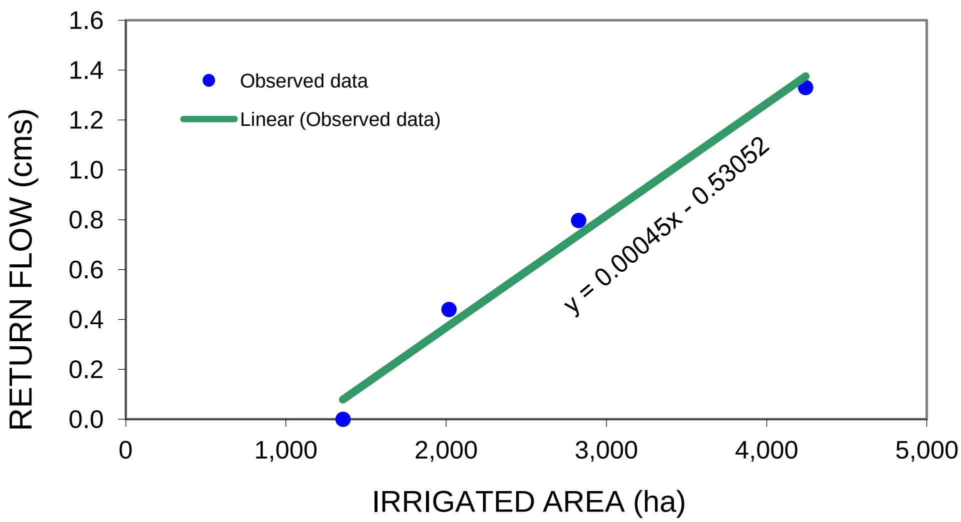
[2]Estimated using regression of irrigated area and measured diversion data

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

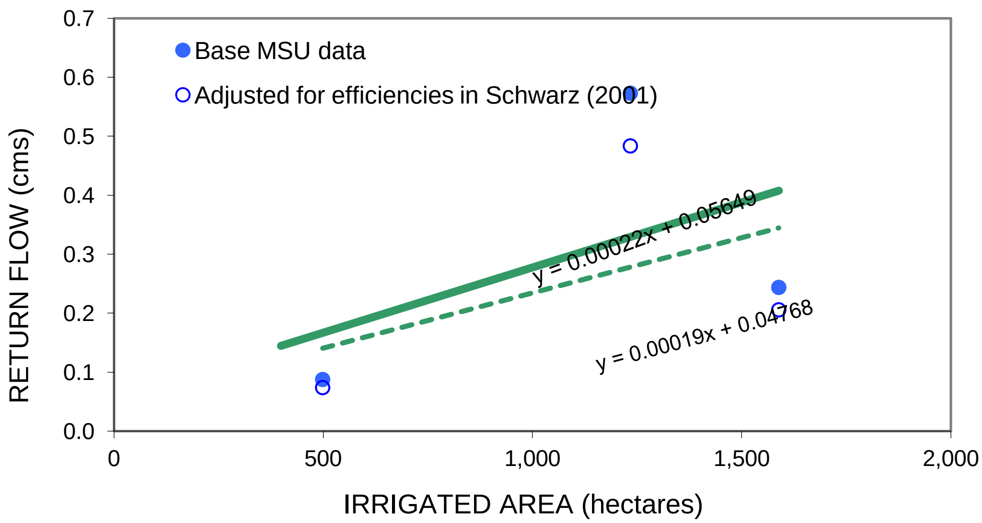
[2]Estimated using regression of irrigated area and measured return flow

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

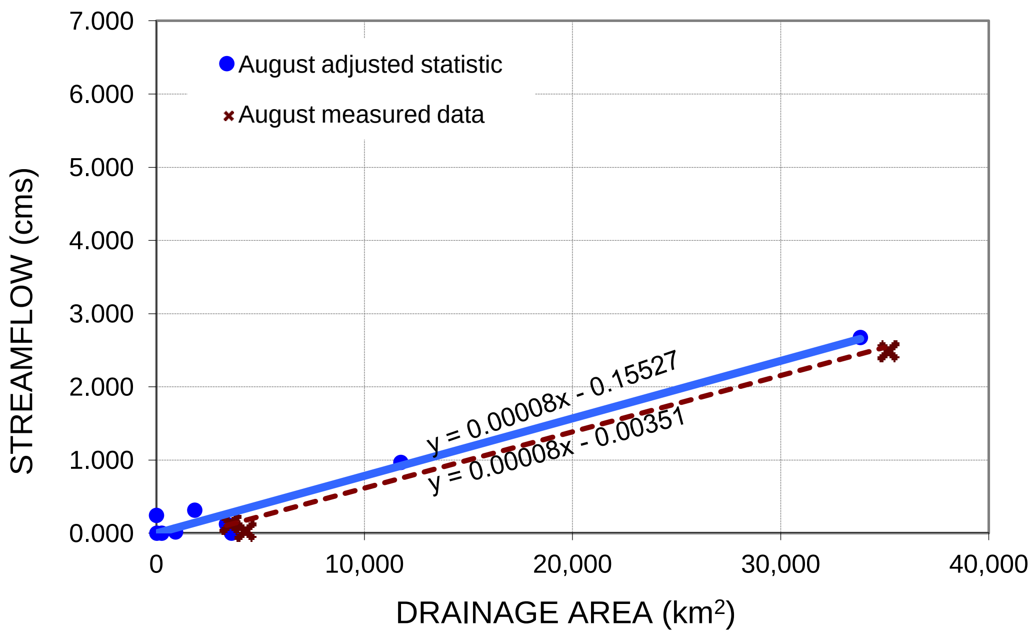
Description	**Gage ID**	**Area [1] (mi^2)**	**Area (km^2)**	
Yellowstone River nr Terry MT	6326530	63,447	164,257	
Yellowstone River at Glendive MT	6327500	66,788	172,906	
		3,341	8,649	
Field data representing remaining area(s)				
O'fallon Creek at mouth (1:24,000 HUC file)	NA	1,578	4,085	
Unmonitored 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 (miles^2)	Drainage Area (km^2)	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



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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 (≈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

[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.
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Montana Department of Environmental Quality
Water Quality Modeling Program
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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 <u>not</u> 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 did 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 not think that 150 mg Chla/m^2 impairs aquatic life uses in large rivers whereas it does 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 25th 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 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 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.

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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 $1/6^{th}$ 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.