Wadeable Streams of Montana's Hi-line Region: An Analysis of Their Nature and Condition, with an Emphasis on Factors Affecting Aquatic Plant Communities

AND

Recommendations to Prevent Nuisance Algae Conditions

Prepared by
Michael Suplee, Ph.D.

May 2004

Montana Department of Environmental Quality
Water Quality Standards Section
1520 E. 6th Ave
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This project could not have been completed without the efforts of a number of people. I would like to thank the private landowners for granting us permission to carry out this study on their property, and I would like to thank the peoples of the Fort Peck Reservation for the same. I would particularly like to thank Dr. Vicki Watson of the University of Montana for her suggestions early in the project on study design and site selection, and for the innumerable conversations we had throughout the project that helped to improve it. I would also like to acknowledge John Lhotak and Marianne Zugel, both graduate students of Dr. Watson, whom conducted the vast majority of the field-work and much of the laboratory analyses. I would like to thank Rosie Sada de Suplee of the Montana Department of Environmental Quality for her assistance with early scouting efforts in the Hi-line region, and Dr. Jeroen Gerritsen of Tetra Tech, Inc. for early suggestions on study design. Thanks also to Dave Peck of the USEPA who spent three days working with me converting the raw EMAP data into useable metrics. I would also like to thank Dr. Loren Bahls of *Hannaea* for his recommendations on appropriate diatom metrics for the study, and Wease Bollman of *Rhithron* Associates, Inc. for her suggestions on potential future directions for assessing prairie streams using macroinvertebrates. Finally, thanks to all the individuals, both within and outside of the Montana Department of Environmental Quality, who reviewed and provided very useful feedback on an earlier draft of this report. Funding for the project was provided by a grant from the U.S. Environmental Protection Agency, Region VIII, Denver, CO, and by the Montana Department of Environmental Quality.

ERRATA SHEET

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Page 2: Paragraph three, sentence four – "1:24K scale" should read "1:100K scale".

Page 7: Paragraph one, sentence four –"1:24K scale" should read "1:100K scale".

Page 20: Table 3.2 – In the "Maximum Possible" column, the value for the Redwater River should be 6, not 9, and its score in the "Final Ranking Score (% of max)" column should be 50, not 33. This change does not, however, alter the final "Stream Condition Group" to which the Redwater River belongs (it remains in the Most Impacted group).

EXECUTIVE SUMMARY

The Montana Department of Environmental Quality is charged with the protection of beneficial uses of state waters, which include but are not limited to drinking, recreation, growth and propagation of fishes, and agriculture. Water quality standards are designed to protect beneficial uses. One of Montana's water-quality standards indicates that state waters must be free from substances attributable to municipal, industrial, agricultural practices or other discharges that will create conditions which produce undesirable aquatic life. However, the number of cases where "undesirable aquatic life" has been quantified is limited. The best example comes from the Clark Fork River. Algae levels on the river bottom were linked primarily to municipal nutrient discharges (nitrogen and phosphorus), were deemed a nuisance that impacted recreational uses, and as a result standards were adopted to control both the algae and the nutrients.

Although excess algae problems are not unique to the Clark Fork River, less effort has been expended in other parts of the state to examine the problem. The U.S. Environmental Protection Agency (EPA) has published recommended limits for nutrients and algae for various regions of Montana, however by the EPA's own admission these recommendations are preliminary and more work at the state level is needed. The main objective of this study was to define nuisance levels for benthic and other algae in wadeable streams of Montana's Hi-line region, and to identify those factors that control algal and other aquatic plant (e.g., macrophyte) biomass. Substantial effort in this study was focused on nitrogen (N) and phosphorus (P), as it was assumed that one or both of these nutrients would be the main factors controlling algae and aquatic plants.

Hugging the Canadian border and falling to the east of the Rocky Mountains is Montana's Hi-line region. The region is part of the Northern Great Plains, and is covered mainly with semi-arid grasslands used extensively for livestock grazing and growing cereal grain crops. The main strategy of the study was to collect data from stream sites at each end of a human disturbance gradient. Data from sites at the high-quality end of the disturbance gradient (those with minimal detrimental impacts from human disturbance) were to be compared to data collected from sites at the opposite, highly impacted end of the gradient. Algal biomass and nutrient concentrations measured in the high-quality sites could then be used as points of comparison for other wadeable streams in the region. "Comparison" sites must support all of their beneficial uses. One caveat to the comparison site approach is that the sites must have similar enough chemical and physical characteristics to the lower quality sites to serve as meaningful indicators of the pre-impact condition. It should be noted that the study occurred during two years of an ongoing drought that began in 1998.

A total of twenty-one sites were sampled. Ten sites were located in the Milk River basin and were sampled in 2001 and eleven where located in the Lower Missouri basin and were sampled in 2002. Samples were collected from May through September in any given year. Three sites in the Milk River basin were sampled in both years of the study.

A weight-of-evidence approach was used to rank the sites by their condition and was based on physical habitat, algae metrics sensitive to nutrients, and fishery quality. Using this approach, four comparison (high quality) sites and nine most-impacted sites were identified. The other eight sites fell somewhere in between on the condition gradient.

Overall, the data from the study suggest that stream incisement (downcutting) is one of the major stressors to aquatic plants and other aquatic life in these streams. This is probably the result of scouring flows that can periodically occur in incised channels. Nutrients were important because they were found to play a role in affecting excessive algae growth.

TSS and turbidity were both significantly correlated (negatively) to total aquatic plant biomass, percent streambed cover by macrophytes, and percent streambed cover by filamentous algae. Entrenchment ratio (the degree to which a channel is downcut, or incised) was significantly correlated (positively) to both total aquatic plant biomass and floating & benthic algal biomass. (Total aquatic plant biomass includes floating algae, benthic algae, and macrophytes.) A high entrenchment ratio indicates that a stream has better access to its flood plain, and will generally have fewer and less severe scouring flows. Thus the entrenchment ratio-aquatic plant correlations show that more incised streams had lower benthic plant biomass.

Permeability of regional soils was significantly correlated (positively) to the entrenchment ratio of the study streams. Study stream sites located in basins having a large proportion of low-permeability soils were more incised then channels located in basin where more permeable soils dominated. The data indicated that in basins where more than 90% of the land area is comprised of impermeable to moderately-slow permeability soils, stream channels are likely to be incised. Stream channel incisement in this region is probably driven by the combined influences of plant cover of the uplands and riparian, inherent soil permeability, and climate factors such as intensity of rainfall and drought. Once the vegetated cover is lost or diminished, streams in more impermeable basins are probably more vulnerable to incisement and headcutting. An historical investigation into regional stream morphology suggested that some of the incisement in the streams may have occurred in the late 19th and early 20th centuries immediately following the periods of open-range grazing and early large-scale homesteading.

Although the stream sites were all located in one general region, they were very diverse in many of their basic characteristics (riparian type, morphology, etc.). To better understand the data it was necessary to develop stream subgroups. A grouping scheme was developed using cluster analysis based on the entrenchment ratio, mid-channel riparian canopy density, and percent woody vegetation of the riparian corridor. Two groups ("Red" and "Green") were developed which comprised nineteen of the twenty-one sites (the remaining two sites were unique in the study) and are described here:

Red group: Comprised of Rosgen F and G streams having low entrenchment ratios (i.e., they were incised), low canopy density (0-14.4%), and low to intermediate (0-58%) woody vegetation density.

Green group: Comprised of moderately to slightly entrenched streams (Rosgen B, C, or E classes) with low canopy density (0-4.4%) and low to moderate (0-35%) woody vegetation density.

The Red and Green groups were useful aides to understanding differences in aquatic plant communities observed among the sites. Streams of the Red group had lower total aquatic plant biomass, lower density of floating & benthic algae, fewer sites with macrophytes, and higher relative proportions of phytoplankton compared to streams of the Green group. Stream morphology apparently played an important role in this. The significant (negative) relationships between TSS and aquatic plant biomass apparently resulted because TSS (and turbidity) were acting as surrogate measures of high erosion, substrate instability, and high scouring flows, all of which can be higher in Red group streams. Taken together, the findings suggested that channel incisement is one of the major stressors to aquatic plants and other aquatic life in these streams.

Data from four different lines of evidence strongly indicated that N was the most limiting nutrient to floating and benthic algae. The lines of evidence were: algal cellular C:N:P ratios; water column nutrient concentration ratios; algal alkaline phosphatase activity; and large numbers of heterocysts on cyanobacteria. It was also found that floating & benthic algal biomass was significantly ($p \leq 0.05$) correlated (positively) to stream water $NO_{2/3}$ concentrations.

An approach was developed for selecting nutrient and algal biomass criteria based on the quality of the comparison sites. This approach indicated that values at the 75th percentile of the nutrient or algal-biomass distributions were appropriate. (The approach taken was different than one proposed by the EPA, although EPA's approach also recommends the 75th percentile.) In the comparison sites the mean summer floating & benthic algal Chl *a* density was 43.6 mg/m^2, the value at the 75th percentile was 61 mg /m^2, and ninety-five percent of the samples were below 106 mg/m^2. Filamentous algae streambed cover at the 75th percentile of the comparison sites was 25%, and mean summer streambed cover did not exceed 30% in the study sites with the exception of one site (Big Sandy Creek) that was apparently polluted with excess N. Stream water Chl *a* was used as a measure of phytoplankton biomass and the value at the 75th percentile of the comparison sites was 14 µg/L; ninety-five percent of the comparison site samples were below 20 µg Chl *a*/L.

The study revealed that too little as well as too much algae and aquatic plant biomass is indicative of problems in the region's streams. Excess algae impacted the recreational beneficial uses of some study streams, while the complete lack of benthic plants (particularly macrophytes) impacted the recreation, fisheries, aquatic life, and waterfowl beneficial uses of other study streams.

A number of specific recommendations have been made to better protect beneficial uses in Hi-line streams. All of the numeric values listed would be applicable to wadeable streams of the region and are intended for the growing season, May through September.

1. Streambed cover by filamentous algae: for a single sample, the value should not exceed 30%; for an average over the growing season, the value should not exceed 25%.

2. Floating & benthic algae: for a single sample, the value should not exceed 110 mg Chl a/m^2; for an average over the growing season, the value should not exceed 65 mg Chl a/m^2.

3. Stream water Chl *a*: for a single sample, the value should not exceed 20 ug/L; for an average over the growing season, the value should not exceed 15 µg/L.

4. The nitrate + nitrite concentration should not exceed 2.3 µg/L, which should prevent nuisance floating & benthic algae. This is the most important nutrient among those listed here, as it showed a significant correlation to floating & benthic algae density.

5. The total N concentration of stream water should not exceed 1,044 µg/L.

6. The total P concentration of stream water should not exceed 153 µg/L. This value is of less significance than the N recommendations because stream algae appear to be N limited.

7. The TDP concentration should be kept below 49 µg/L and the SRP concentration below 8 µg/L. These value are of less significance than the N recommendations, because stream algae appear to be N limited.

In this region the presence of a healthy and widely distributed macrophyte community should be taken as indicative that the streams have a reasonable level of morphologic stability. Their complete absence from a stream should be taken as potentially indicating that the site has streambed stability problems and scouring flows.

In the Hi-line region, watershed restoration efforts should consider the soil permeability in the basins of the streams in question. To slow or diminish regional stream incisement, streams located in basins with high proportions of low permeability soils will probably require more careful management than those in higher-permeability basins.

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GLOSSARY OF TERMS AND MEASUREMENT UNITS

Glossary of Terms

AFDW: Ash free dry weight. The mass of a material (usually organic) obtained by first drying the material at 105 ^{o} C, and then heating the material to 500 ^{o} C for 1 hour.

Aquatic macroinvertebrate: An organism found in bodies of water, frequently associated with the bottom, that does not having a spinal column and is visible with the naked eye.

ARM: Administrative Rules of Montana.

Bankfull: The water discharge at which channel maintenance is the most effective; that is, the discharge at which moving sediment, forming or removing bars, forming or changing bends and meanders, and generally doing the work that results in the average morphologic characteristics of channels. Reoccurs about every 1.6 years.

Benthic: On or associated with the sediments or bottom of a body of water.

Biomass: The total mass or amount of living or dead organisms in a particular area or volume.

Chlorophyll *a*: The major green pigment found in the chloroplasts of plants, including algae.

Density: Quantity of a number per unit area, volume, or mass.

Diatom: Any one of a number of microscopic algae, one celled or in colonies, whose cell walls consist of two box-like parts or valves made of silica.

Floating Algae: In this report, the term floating algae is used to refer to macroscopic, mainly filamentous algae that were floating in still water areas (e.g., *Spirogyra*).

Geomorphology: The science dealing with the nature and origin of the earth's topographic features.

Incised: Used in reference to streams, meaning to have cut down into the surrounding landscape.

Intermittent: a stream or stream reach that is below the local water table for at least some part of the year, and obtains its flow from both surface run-off and ground water discharge.

Macrophyte: Aquatic plants capable of achieving their generative cycles with all vegetative parts submerged or supported by the water.

MCA: Montana Code Annotated.

Metric: A characteristic of a biological assemblage (e.g., fishes, algae) that changes in some predictable way with increased human influence.

Molar: The gram molecular weight of an element or compound.

Morphology: Any scientific study of form and structure.

Periphyton: The microscopic flora and fauna that grow or are associated with the bottom of a body of water, and includes microscopic algae, bacteria, and fungi.

Phaeophytin: A chemical compound resulting from the degradation of chlorophyll.

Phytoplankton: Free living, generally microscopic algae commonly found floating or drifiting in waterbodies such as the ocean, lakes and streams.

Temporal: Of or associated with time.

Measurement Units

CFS	cubic feet per second
cm	centimeter
ft	feet
in	inches
hr	hours
L	liter
m	meter
m^2	square meter
m^3	cubic meter
mg	milligram
mm	millimeter
nmol	nanomole
NTU	nephelometric turbidity units
µS	microSiemens
µg	microgram

SECTION 1.0
GENERAL INTRODUCTION AND HISTORICAL PERSPECTIVE ON STREAM CHANGES IN THE HI-LINE REGION

1.1 General Introduction

1.1.1 Motivation for the Study

One of the main purposes of the Montana Department of Environmental Quality (MT DEQ) is to protect the beneficial uses of state waters. Beneficial uses of state waters include drinking, recreation, swimming, growth and propagation of fishes, waterfowl, and agriculture (ARM 17.30.620 through 17.30.629), and are protected by water quality standards. The Montana Water Quality Act specifically directs the Board of Environmental Review to formulate and adopt water quality standards (75-5-301(2)(a) MCA), and in the vast majority of cases the ‘formulate’ component of this statute falls to the MT DEQ (the Board of Environmental Review and the Legislature have the authority to adopt water quality standards). One particular water quality standard indicates that state waters must be free from substances attributable to municipal, industrial, agricultural practices or other discharges that will create conditions which produce undesirable aquatic life (ARM 17.30.637(1)(e)). However, the number of cases where “undesirable aquatic life” has been quantified is limited. One example can be found in the Clark Fork River. In the Clark Fork River it was concluded that human-caused nutrient sources substantially control the growth of benthic algae, and that very dense growth of the algae is a visual and aesthetic nuisance; therefore, it is undesirable aquatic life. In 2002 the Montana Board of Environmental Review adopted regulations limiting the amount of benthic algal biomass that is allowed to grow on the river’s bottom to a summertime maximum of 150 mg Chl a/m^2. Benthic algae biomass at or below 150 mg Chl a/m^2 is expected to maintain the algae below a nuisance threshold (Welch et al. 1988). Because nutrients were found to play a role in controlling the benthic algae, nitrogen (N) and phosphorus (P) concentrations standards were also adopted (ARM 17.30.631).

The data leading to the standards for the Clark Fork River came from a number of scientific studies that were carried out over a twenty-year time period by state, federal, university, and other entities. And although excess algae problems in Montana are unlikely to be unique to the Clark Fork River, much less effort has been expended to examine the problem in other waterbodies, or in other parts of the state. General methods for establishing appropriate algal biomass levels and nutrient concentrations, as well as recommended criteria for algal biomass and nutrient concentrations for specific regions of Montana, have already been published by the US Environmental Protection Agency (EPA; USEPA 2000a; USEPA 2001). A key tenet of EPA’s guidance and recommendations, and for that matter of the state law cited in ARM 17.30.637(1)(e), is that excess plant biomass is undesirable. However, the EPA recognizes and clearly states that their recommendations are preliminary, and that further work at the state level needs to be undertaken in order to refine the nutrient and algal biomass criteria they have published (USEPA 2001). The main objective of this study was to define appropriate benthic algal biomass and nutrient concentrations for wadeable streams in the region of Montana locally known as the Hi-line, and to identify those factors that control or affect benthic aquatic plant growth. As a secondary objective other physical, water quality, and selected biological characteristics of the streams were measured in order to characterize the streams.

1.1.2 Location of Study

The Hi-line region of Montana encompasses the land area adjacent to U.S. Highway 2, from the Rocky Mountain front near Browning in the west, to the North Dakota border near Bainville in the east, to the Canadian border to the north (see Fig. 2.1 in Section 2.0). The land area is part of the glaciated prairie sub-region of the North American Great Plains (Hunt 1974; Covich et al. 1997). Prior to European settlement, the area was mainly a semi-arid mixed prairie (Weaver and Albertson 1956), and is now used mainly for grazing and growing cereal grain crops. Closely corresponding to the general Hi-line region is the geographic area referred to as the Northwestern Glaciated Plains ecoregion (Omernik 2000; green shaded area of Fig. 2.1). Ecoregions are geographic constructs of land areas that have similar geologic, soil, climate, and vegetation characteristics (Omernik 1987). The present study was restricted almost exclusively to the Northwestern Glaciated Plains ecoregion, under the assumption that the region's streams would have similar characteristics unique to the region.

1.1.3 General Approach to the Study

The fundamental strategy of the study was to locate wadeable streams sites at each end of a human-disturbance gradient and then to use a comparative data approach, wherein data collected from stream sites at the high-quality end of the gradient (i.e., minimal detrimental impacts from human disturbance) could be compared to data collected from stream sites at the opposite, highly impacted end of the disturbance gradient. The stream sites from the high-quality end of the spectrum (comparison sites) provide our best understanding of the potential behavior of these stream systems when least changed by human disturbance. "Human disturbance" includes all activities controlled by man, including crops and grazing practices. "Wadeable" streams were 3rd or 4th order reaches (1:24K scale; Strahler 1964) that were shallow enough that they could be sampled by wading or at most by the use of a small raft, and did not include larger rivers such as the Milk River.

It was assumed that the algal biomass and nutrient concentrations measured in the comparison sites would be appropriate for other wadeable streams in the region. One caveat to this approach is that the comparison sites need to be sufficiently similar in certain key aspects to the impacted sites to serve as meaningful indicators of the pre-impact nature and condition of the impacted sites. This is important, for although the study was restricted to a single geographic area there still exists substantial variability among streams within the region. After the initial characterization of key physical, chemical, and riparian habitat aspects of the sites, sub-groups were identified which could be better used for intra-group comparisons. This was done to assure that the comparison sites were as applicable as possible.

The study was initiated in the spring of 2001 and data collection efforts proceeded throughout the summer. It was recommenced in the spring of 2002 and then proceeded throughout that summer. The study occurred during two very dry years of an ongoing drought in the region that began in 1998 (NWS 2004). Therefore, the data in this report may be different than what would have been measured during a wetter climatic period.

1.2 Historical Perspective on Stream Changes in the Hi-line Region

Frequently noted in the field was the observation that many streams showed evidence of incisement (downcutting) within approximately the past 100 years. This was evidenced mainly by abandoned floodplains that retained older but living cottonwood galleries (Fig. 1.1), a condition that has also been observed by others (Jones 2003). Recruitment of Great Plains cottonwood (*Populus deltoides*) on prairie river floodplains is associated with flood processes (Bradley and Smith 1985), and the trees shown in Fig. 1.1 were surely established on what is now an abandoned flood plain. These cottonwood trees usually don’t live past about 100 years, although some individuals may live 200 years (Tree Guide 2004). Frequently, the old floodplains were 4-7 meters above the current channels. Nothing specific to the Hi-line region could be found on the topic of tributary stream incisement in the scientific literature, although there are a number relevant facts found in the general history of the region (presented below). Therefore, an historian was commissioned to search for specific information concerning stream incisement, since about 1880, of the Hi-line region’s streams, with special emphasis to be placed on the streams examined in the present study. The historian searched through many records and collected information from a variety of sources, including state and local historical societies, museums, local county libraries, the National Archives in Washington, D.C., railroad archives, and personal interviews (Confluence Consulting Inc. 2003). But before getting to the findings of the Confluence Consulting Inc. (2003) report a brief, general history of land uses in the region will be presented.

After the Lewis and Clark expedition passed through the area along the Missouri River in 1805, the fur trade dominated the region until the civil war (DeVoto 1947; Tirell 1997). During this time period there was a decline in demand for beaver pelts as men switched to silk hats, and for this and other reasons buffalo (*Bison bison*) hides became the region’s most profitable commodity. Buffalo were exterminated wholesale beginning in the 1870’s, and by 1883 they were gone (Tirell 1997). In the late 1880’s, after the unusually harsh winter of 1886-87 killed off large herds of open-range cattle south of the Missouri River, the large open-range cattle outfits entered the Hi-line region. Huge herds of cattle and sheep were grazed in the region over the next 20 years, until the hard winter of 1906-7 killed thousands of cattle and sheep and caused most of the large open-range outfits to leave (Spritzer 1999). But by this time a railroad line built by James Hill in 1887 had begun to draw thousands of settlers to the region, settlers who began to fence the prairie, break the sod, and plant wheat and other cereal grains on small homesteads.

(The rail line built in 1887 by James Hill was renamed the Great Northern in 1890, and was referred to as the Hi-line; this is also the source of the region's local name.) In 1908 the Great Northern railroad began an unprecedented promotion scheme to get large numbers of people to move to the region, and was highly successful. By 1917 a relatively wet period ended in drought, and of the nearly 80,000 people who had moved to eastern Montana over the previous 10 years, 60,000 had left by 1922 (Spritzer 1999). For those that stayed in the Northern Great Plains, farms and ranches became larger (Hart 1957). Drought again hit the region hard in the 1930's, and then again in the early 1950's; droughts of varying lengths and magnitudes have occurred in each decade since, up to the present (Diaz 1983; NWS 2004).

Concurrent with the changes in land use were changes to the hydrologic regime of the region. The St. Mary diversion was finished in 1917 and transfers water from the St. Mary River drainage to the headwaters of the Milk River drainage (Bradley and Smith 1985). Because the water then flows to Canada before returning to the United States, flow deliveries of both the Milk and St. Mary rivers are regulated by international agreements (Goos and Carswell 2002). The Fresno dam was built in 1939 on the Milk River 20 km upstream of Havre, MT, and Fort Peck dam was completed on the Missouri River near Glasgow, MT in 1940 (Spritzer 1999). Battle and Frenchman creeks (two streams in the present study) originate in Canada and are regulated by Canadian dams built in the late 1930's that allow for inter-basin water storage and transfers (Goos and Carswell 200).

Figure 1.1. Example of stream incisement (Battle Creek). The now-abandoned floodplain is above the channel where the cottonwood trees line the stream. Measures of entrenchment ratio (Rosgen 1996) indicated that even a 50-year magnitude flood would not reach the old floodplain.

The historical research commissioned by MT DEQ to investigate changes in tributary streams of the Hi-line region (Confluence Consulting Inc 2003) did locate some valuable photographs and other records. However, facts relating to changes in the region's wadeable streams were generally scant. Twenty-two early maps of the region (late 19th and early 20th centuries) were located, mostly produced by the General Land Office (GLO), but none of them contained notes on morphology or vegetative cover along the streams (field notes of this type have been found on GLO maps of the Yellowstone River, and on GLO maps from the state of Oregon). A series of aerial photographs from the late 1930's were of limited use, although some clearly showed the incised nature and general topography of regional streams.

The search located a number of personal photographs from the region, photos showing an array of riparian types from cottonwood galleries to willow-dominated to open grasslands. Some of the earliest personal photos (late 19th and early 20th centuries) show some streams that are incised, while others are not. Two historic photographs from known locations were of particular interest. One was taken in 1927 on Battle Creek about 17 stream miles north of the location shown in Fig. 1.1. In 2001 a site just a few stream miles further north of the 1927 photograph was surveyed and photographed as part of another project. In both photos men are standing along the streambank, and it is clear that the bank heights are virtually the same. So in this case no stream incisement since 1927 can be noted. In another photograph taken somewhere on the NF Battle Creek between 1914 and 1923, clear indications of stream incisement and heavy erosion are obvious (Fig. 1.2). The latter photograph is evidence that significant stream incisement had already begun, at least in some locations, by the early 20th century.

Conversations with longtime Hi-line residents provided additional information on regional stream incisement (Confluence Consulting Inc 2003). Interviewees discussed memories of their own experiences as well as stories they remembered from previous generations living in the area. Most residents described alternating conditions of drought and high precipitation, and indicated that they thought that high flows from rainstorms could easily erode streambank sediments. Residents further noted that the regional soils are highly erodible. Nobody that was interviewed recalled a specific time period or event, or series of events, leading to substantial or notable stream incisement. One resident mentioned that streams where there is more gravel were generally less incised. Finally, a number of residents noted that high springtime flows have not occurred very often in the past 20-30 years, due to reduced snowfall and more flood control, diversion, and water-storage projects.

A photo-documentation of Northern Great Plains landscape changes that have occurred since 1910 was prepared by Klement et al. (2001). The report shows time-series photographic records of sites in and outside of Montana, although none are in the Hi-line region. Only a few of the photos have streams in them, and those that do don't show obvious signs of incisement over the course of the photo series.

What can be drawn from these facts and findings? Since it is known that cottonwoods usually propagate on floodplains in association with flood processes (Bradley and Smith 1985), then it is clear that many of the streambeds were once located at a higher elevation then they are today, and that they periodically overtopped their banks and spread out on adjacent floodplains. However, none of the Hi-line residents that were interviewed recalled experiencing or hearing about an event, series of events, or unusual conditions that caused stream downcutting (Confluence Consulting Inc. 2003). But the historic record shows that most people arrived in the regions after 1908, and therefore if major changes had already occurred they would have appeared to be the natural state of things. Similarly, the various photo series in Klement et al (2001) do not demonstrate major stream downcutting in the Northern Great Plains over the past 80 years, but stream photos were few and again all of them were taken after 1910. Then there is the Hi-line photograph from NF Battle Creek taken somewhere between 1914 and 1923 that does show major channel downcutting and erosion (Fig. 1.2). The evidence, although slim, suggests that much of the downcutting observed in small streams in this area may have occurred late in the 19th and early 20th century, possibly in response to the intense grazing of the open range period followed by the breaking of the sod. The latter topic will be examined more thoroughly in Section 4.0.

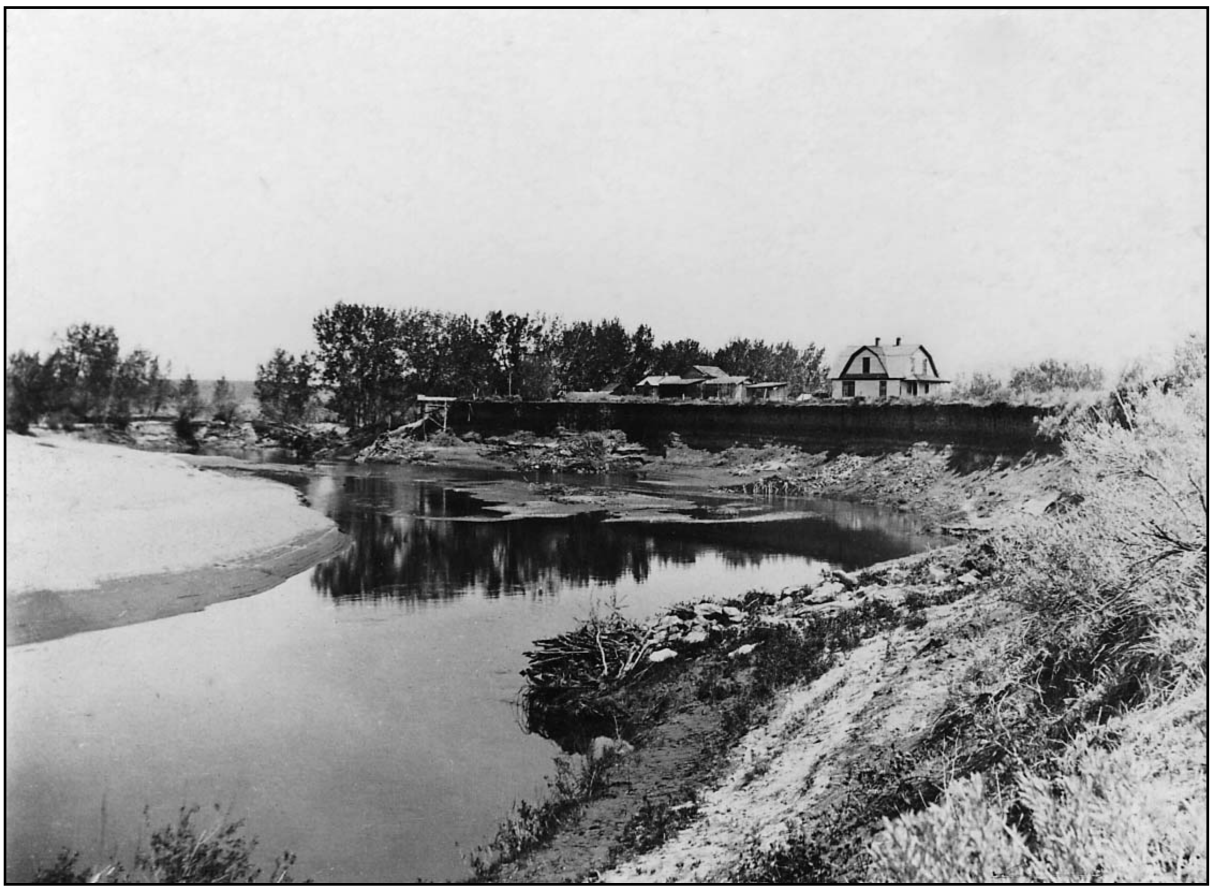


Figure 1.2. North Fork Battle Creek, sometime between 1914 and 1923.
Note the severe downcutting and sediment deposition midchannel.

SECTION 2.0
SITE SELECTION, SITE LOCATIONS, AND ROUTINE SAMPLING METHODS

2.1 Site Selection Process

As discussed in Section 1.0, sites were selected with the intent of choosing locations at each end of a habitat quality gradient; those in the best condition, and those most impacted. An initial effort to accomplish this was made using a simple geographic information system (GIS) approach (Tetra Tech, Inc. 2001). Fifth-field HUC basins (Seaber et al. 1987) were overlain in GIS with Multi-Resolution Land Characteristics (MRLC) land-use data and Strahler stream order (Strahler 1964). Stahler stream order was used to select streams segments that were 3rd or 4th order (1:24K scale), which were likely to be wadeable and to have remained perennially flowing or intermittent. Based upon the potential for each MRLC land use to contribute to man-caused sources of nutrients, each basin was ranked on a scoring scale from 100 to –100 as a function of the proportion of each land use contained within its boundaries. (Examples of land uses that are assumed to have high potential for nutrient contributions are “row crop” and “urban areas”, while an example of a low-nutrient potential land use is “natural cover”.) High basin scores indicated little potential for nutrient impacts to waterbodies within the basin, while low scores indicated the reverse.

It became apparent during field reconnaissance trips in spring 2001 that the fifth-field HUC basin scores were too coarse to adequately identify specific sites as “impacted” or “reference”. A similar conclusion was arrived at independently in a similarly structured wetland study along the Hi-line that took place in 2002 and 2003 (Jones 2004). In the present study, riparian conditions often changed from fence-line to fence-line, depending upon local land use management. Some MRLC land uses were simply unable to capture certain types of impacts. For example, grazing pressure varied tremendously throughout the basin as did its impacts on stream habitat and water quality, however the MRLC classified all grazing lands simply as “natural cover”, a supposedly low-nutrient contributing land-use type. For these reasons, sites were ultimately selected based on best professional judgment (BPJ) which included; consideration of the overall scores from the fifth-field HUC GIS effort, scores on the DEQ stream reach assessment form, overall site appearance, and land accessibility. Both DEQ and University of Montana staff participated in site selection during both years of the study. Final sites were picked after visits to a number of potential sites throughout the region.

2.2 Site Locations

A total of 21 segments on 21 different streams were studied in 2001 and 2002. Site locations are detailed in Table 2.1. In 2001, sites were restricted to the Milk River basin east of where it re-enters Montana from Canada. In 2002 the study was expanded to include sites outside of the Milk River basin but still within the Northwestern Glaciated Plains ecoregion (Omernik 2000). The Northwestern Glaciated Plains ecoregion is shown in green in Fig. 2.1, along with the site locations (yellow dots). Three sites (Rock, Willow N, and Porcupine creeks) were sampled in both years in order to measure year-to-year variability of important parameters.

Table 2.1. Sampling frequency and location of northeastern Montana stream sites in the study.

Site Name	Year(s) Sampled	Months Sampled	Reach Length (m)	County	4th Field HUC	Latitude (DMS)	Longitude (DMS)
Big Sandy Creek	2001	May, June, Aug., Sept.	200	Hill	10050006	48 27 08	109 55 10
Clear Creek	2001	May, June, Aug., Sept.	150	Blaine	10050004	48 18 19	109 29 19
Battle Creek	2001	May, June, Aug., Sept.	280	Blaine	10050008	48 39 02	109 13 49
Beaver Creek	2001	May, June, Aug., Sept.	400	Phillips	10050014	48 15 04	107 34 20
Larb Creek	2001	May, June, Aug., Sept.	150	Valley	10050014	48 08 45	107 17 30
Frenchman Creek	2001	May, June, Aug., Sept.	280	Phillips	10050013	48 45 23	107 12 41
Rock Creek	2001 & 2002	May, June, July, Aug., Sept.	190	Valley	10050015	48 35 08	106 59 51
Willow Creek (North)	2001 & 2002	May, June, July, Aug., Sept.	480	Valley	10050015	48 34 35	106 58 53
Willow Creek (South)	2001	May, June, Aug., Sept.	190	Valley	10050012	48 08 25	106 37 36
Porcupine Creek	2001 & 2002	May, June, July, Aug., Sept.	150	Valley	10050016	48 12 29	106 22 53
West Fork Poplar River	2002	May, June, July, Aug.	240	Daniels	10060004	48 41 50	105 49 55
Wolf Creek (at Wolf Point, MT)	2002	May, June, July, Aug.	240	Roosevelt	10060001	48 05 12	105 40 37
Middle Fork Poplar River	2002	May, June, July, Aug.	280	Daniels	10060003	48 55 13	105 36 29
Butte Creek	2002	May, June, July, Aug.	150	Daniels	10060003	48 49 48	105 36 19
Sheep Creek	2002	May, June, July, Aug.	400	McCone	10060002	47 58 03	105 23 13
Redwater River	2002	May, June, July, Aug.	280	McCone	10060002	47 55 41	105 15 49
Smoke Creek	2002	May, June, July, Aug.	280	Roosevelt	10060006	48 21 32	104 44 46
Wolf Creek (nr Medicine Lake)	2002	May, June, July, Aug.	150	Sheridan	10060006	48 29 27	104 36 23
Shotgun Creek	2002	May, June, July, Aug.	200	Roosevelt	10060005	48 09 39	104 14 48
Little Muddy Creek	2002	May, June, July, Aug.	240	Roosevelt	10060005	48 07 49	104 06 46
Horse Tied Creek	2002	May, June, July, Aug.	150	Roosevelt	10060005	48 07 03	104 06 19

2.3 Routine Sampling Procedures

2.3.1 Reach Layout and Habitat Assessment

In both 2001 and 2002, streams were laid out and assessed using the Western Pilot EMAP physical habitat characterization protocols (USEPA 2000b). Total reach lengths (measured along the thalweg) were established as forty times the wetted width measured at the initial visit, or a minimum of 150 m. Each reach was then divided into 10 equally spaced subreaches, which provided a total of 11 transects perpendicular to stream flow along the entire reach (USEPA 2000b). In addition to EMAP habitat characterization, all sites were assessed using a standard MT DEQ stream reach assessment form (April 1995 version). The MT DEQ assessment form evaluates (among other things) riparian condition, but to further the riparian examination two other riparian assessment methods were employed. In 2001 the sites were assessed using the Natural Resource Conservation Service (NRCS) riparian assessment rating system (Berg and Leinard 2001) while in 2002 they were assessed using a riparian health form for Montana stream systems (Fitch et al. 2001). Both of these methods examine and assess comparable aspects of the riparian corridor.

Stream type (Rosgen 1996) was also determined at the sites. The entrenchment ratio was measured in 2001 in all streams, and the remaining components necessary for determining a Rosgen stream type (width/depth ratio, sinuosity, slope, substrate) were extracted from the EMAP physical habitat measurements. It became apparent that certain methods employed by EMAP could bias the determination of Rosgen stream types (discussed further in Section 4.0), therefore in 2002 sites were assessed at appropriately selected cross-sections within the reach using Rosgen methods in addition to the EMAP methods. In both 2001 and 2002 a laser level was employed for all Rosgen-recommended measurements of stream geomorphic characteristics.

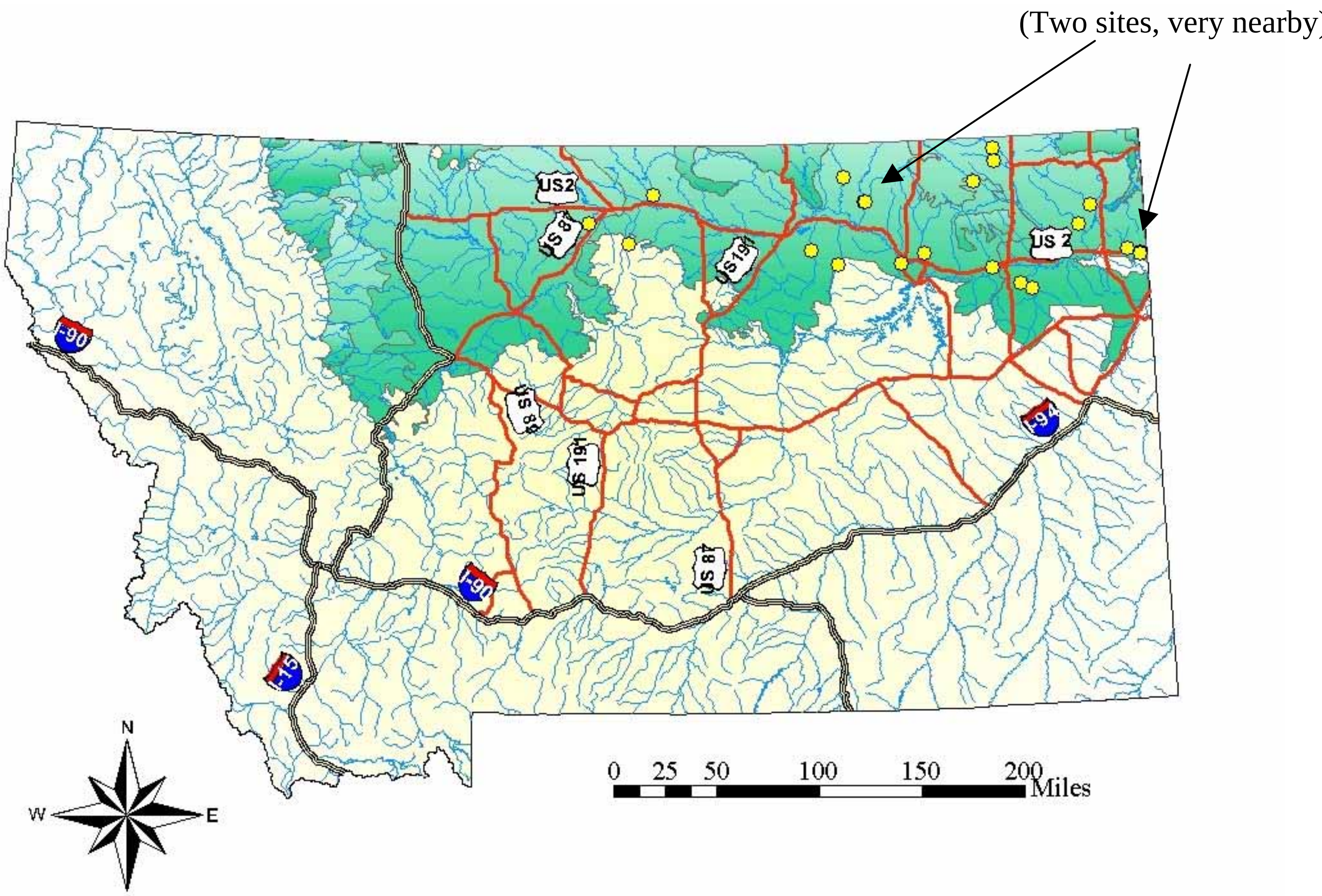


Figure 2.1 Map of site locations.

2.3.2 Chemical and Physical Water Quality Measurements

All water samples were submitted to the Environmental Laboratory of the Montana Department of Public Health and Human Services in Helena, MT, after having been collected and preserved using standard methods detailed in the project's quality assurance project plan (QAPP; Tetra Tech Inc. 2001). In 2001, each site was sampled once in June, August, and September (Table 2.2). After data from the first year of the study was reviewed, it was decided to drop sampling for dissolved aluminum and total zinc for 2002. This was due to budget considerations and also because none of the streams in which zinc was detected violated state water quality standards, and only a single sample (from Larb Cr.) exceeded the chronic (but not acute) aluminum standard (MT DEQ 1999b). In 2002 stream sites were sampled once during the months of May, June, July, and August (Table 2.2).

A number of real-time measurements were made during each field visit. Specific conductance (µS/cm @ 25^{o} C) was measured in 2001 using an Orion 150 conductivity meter or a YSI Model 85 multi-meter. The pH was measured using an Orion model 250A pH meter. Meter calibration was checked prior to departure to the field or (for pH) calibrated daily. Dissolved oxygen (DO) was measured using a YSI meter, calibrated just prior to sampling. Flow was measured using the velocity-area method (USEPA 2000b) and calculated as ft^3 sec^{-1} (CFS). Water transparency (cm) was also measured, as was water temperature and air temperature in ^{o} C (using either multi-meters or hand-held thermometers). In 2002 the only real-time field measurements taken were flow, conductivity, DO, and water temperature.

Table 2.2. Chemical and physical water quality measurements made at sites in 2001 and 2002.

Parameter	Reported Units	Analytical Method	Detection Limit
I. Laboratory Analyses			
Cations			
Calcium (Ca^{2+})	mg/L	EPA 200.7	0.02 mg/L
Magnesium (Mg^{2+})	mg/L	EPA 200.7	0.01 mg/L
Potassium (K^{+})	mg/L	EPA 200.7	0.04 mg/l
Sodium (Na^{+})	mg/L	EPA 200.7	0.02 mg/L
Anions			
Bicarbonate (HCO_3^-)	mg/L	Calculated from pH & alkalinity (APHA 1993)	na
Chloride (Cl^-)	mg/L	EPA 300.0	
Sulfate (SO_4^{2-})	mg/l	EPA 300.0	0.05 mg/L
Other			
Total Alkalinity	mg/L	EPA 310.2	1 mg/L
Specific Conductance	µS/cm @ 25 ^{o} C	SM2510B	
pH	Standard Units	EPA 150.1	
Total Suspended Solids @ 105 ^{o} C	mg/L	SM 2540-D	0.5 mg/L
Turbidity	NTU	EPA 180.1	
Aluminum, dissolved	mg/L	EPA 200.7	45 µg/L
Selenium, dissolved	mg/L	EPA 200.7	1 µg/L
Zinc, total recoverable	mg/L	EPA 200.7	1 µg/L
II. Field Measurements			
Flow	CFS		na
Dissolved Oxygen	mg/L	meter	±1% full scale
Specific Conductance	µS/cm @ 25 ^{o} C	meter	± 0.5% full scale
Water Transparency	cm		na
Water Temperature	^{o} C		na
Air Temperature	^{o}C		na

2.3.3 Nutrients

Water samples analyzed for major plant nutrients (carbon (C), nitrogen (N), and phosphorus (P); Table 2.3) were submitted to the University of Montana Flathead Lake Biological Station in Polson, MT, after having been collected and preserved using standard methods detailed in the project's QAPP (Tetra Tech Inc. 2001). In 2001 samples were collected once at each site during the months of May, June, August, and September. Field blanks were run once during each field trip, and water samples were analyzed for total persulfate nitrogen (TN), nitrate + nitrite ($NO_{2/3}$), ammonia (NH_3), total phosphorus (TP), total dissolved P (TDP), dissolved inorganic carbon (DIC), and dissolved organic carbon (DOC; Table 2.3).

Table 2.3. Major plant nutrient measured at sites in 2001 and/or 2002.

Parameter	Reported Units	Analytical Method	Detection Limit
dissolved inorganic Carbon (DIC)	mg/L	Phosphoric acid injection	0.04 mg/L
dissolved organic Carbon (DOC)	mg/L	Wet oxidation or high temp. combustion	0.04 mg/L
total Nitrogen (TN)	µg/L	Persulfate digestion	20 µg/L
nitrate + nitrite ($NO_{2/3}$)	µg/L	Cadmium reduction	0.6 µg/L
ammonia (NH_3)	µg/L	Automated phenate method	5 µg/L*
total Phosphorus (TP)	µg/L	Sulfuric acid and persulfate digestion followed by ascorbic acid	0.4 µg/L
total dissolved Phosphorus (TDP)	µg/L	Digestion followed by ascorbic acid	0.4 µg/L
soluble reactive Phosphorus (SRP)	µg/L	Direct ascorbic acid	0.3 µg/L

* As a result of high background ammonia on field filter blanks, the practical detection limit was approximately 20 µg/L.

It was noted after reviewing the data for 2001 that only 3 water samples showed detectable levels of ammonia among 44 collected, as the ammonia field filter-blank values raised the practical detection limit to about 20 µg/L (Watson, personal communication). Further, the original use for the DIC and DOC data was not realized, as sites could not be sampled as often as originally hoped. Therefore, the 2002 nutrient sampling was altered. In 2002 sites were each sampled once in May, June, July, and August for TN, $NO_{2/3}$, TP, TDP, and also soluble reactive phosphorus (SRP). As in 2001, field blanks for all measured parameter were collected during each field trip.

Both TDP and SRP were analyzed in the 2002 water samples in order to compare their values, as there is frequently confusion about which of these two dissolved P fractions is more appropriate to measure in flowing water. In both years, dissolved parameters (DIC, DOC, $NO_{2/3}$, NH_3, TDP and SRP) were analyzed from filtrate that had been passed through 0.45 µm filters in the field.

2.3.4 Floating & Benthic Algae, Macrophyte, and Phytoplankton Measurements

Combined floating & benthic algae and macrophyte samples were collected for chlorophyll *a* (Chl *a*; corrected for phaeophytins) and ash free dry weight (AFDW) analysis. Chl *a* was analyzed using a hot alcohol extraction method (Sartory and Grobbelaar 1984), while AFDW was analyzed using standard methods (APHA 1998). Samples were collected from up to11 transects (< than 11 if transects became dry) that were equally spaced along the stream reaches, following methods drawn from the Western Pilot EMAP protocols (USEPA 2000b). Transects did occasionally became dry during the course of the field season. During each visit in 2001 samples were alternately collected from the right bank, then left bank, then stream center, starting from a randomly chosen point at the downstream end of the reach. Each transect's sample was analyzed individually. In 2001 samples were collected in June, August, and September, while in 2002 they were collected in June, July and August. In 2002, due to concerns that a given location along a transect might be sampled more than once, samples were collected at each transect along the left bank in June, the right bank in July, and mid channel in August.

Samples were collected using one of three methods, as appropriate for the dominant conditions at each transect. At transects with luxuriant growth of aquatic macrophytes and filamentous algae, regardless of stream substrate, a metal hoop (710 cm^2 area) was placed on the bottom to collect all of the algae and macrophyte material within the hoop. This approach captured all of the plant material, including that which was suspended up in the water column (i.e., it vertically integrated). Each stone within the hoop was not scraped to perfect cleanliness, but rather the vast bulk of the material was consistently collected. The material was chopped, mixed, and then held frozen in dark storage until analyzed for Chl *a* and AFDW. At transects where hoop samples were collected, a visual estimate (1 meter above and 1 below the transect line around the sampling area) of the proportion (by area coverage) of macrophytes and filamentous algae was recorded. In 2002, commonly found macrophytes were collected and pressed for later identification.

At transects which had rocky bottoms (but not exhibiting luxuriant growth), periphyton was collected using a template with an internal surface area of 25 cm^2 that was placed over a representative stone from the sampling area. The template was made of an eighth-inch thick flexible foam material cut to appear like a picture frame (the internal area was equal to 25 cm^2). This method was used for diatoms, *Nostoc* films, or very short uniform growths of attached filaments. The area within the template was scraped/scrubbed clean and the material placed into a collection jar. The material was frozen and kept in the dark until analyzed for Chl *a* and AFDW.

For transects which had mud, silt, or fine-sand bottoms without luxuriant plant growth, a sediment core sample of the stream bottom was collected using a cut-off 60 cc syringe. The syringe barrel opening had a surface area of 5.7 cm^2. In deeper water, a broom handle was attached to the syringe to extend the reach of the sampler. After collecting several vertical inches of sediment within the syringe, the core sample was extruded and all sediment discarded except for the upper 1 cm (i.e., sediment from the sediment-water interface to a depth of one cm). This was then placed in a small container, frozen and stored in the dark, and later analyzed for Chl *a*. AFDW was not measured on these samples, as they likely contained organic material deposited over many years.

Some streams exhibited noticeable phytoplankton growth in the water column. In 2001, replicate water samples from different pools within stream reaches were collected from the sites with water that had a green tint. In 2002 duplicate water samples for phytoplankton were collected in a similar manner, but were collected from each stream site in July regardless of the waters' appearance. Duplicates were collected from two different pools during periods when the streams were not flowing. In both years phytoplankton water samples were filtered through 0.45 µm GF/F filters and frozen, and later analyzed for Chl *a* using a hot alcohol extraction method (Sartory and Grobbelaar 1984).

Carbon, N, and P content of benthic algae was determined in order to examine the Redfield ratio (Redfield 1958) of the stream algae community. Reach-wide filamentous-green algae composite samples and reach-wide diatom-dominated composite samples (i.e, pebble scrapes) were each analyzed for C and N, however fine-sediment composites samples (analogous to the sediment core samples) were not, as they likely contained large quantities of non-algal C and N deposited in the stream bottom. The C and N content of dried (at 105 ^{o}C) algal biomass was measured using the high temperature induction furnace method (American Society of Agronomy 1996) by the Montana State University Soils Analytical Laboratory in Bozeman, MT. In 2001, C and N samples were treated with a weak HCl solution (Froelich 1980**;** Niewenhuize et al. 1994) prior to drying, in order to remove carbonates that may have encrusted or been collected along with the algae samples. In 2002, both acid-treated and un-acidified samples were analyzed for C and N, to ascertain if the acid treatment may have influenced N content of the samples. The total P content of periphyton composite samples was analyzed using the method of Solorzano and Sharp (1980). For the same reasons described for C and N, composite sediment cores were not analyzed for total P. The C, N, and P content of the composite algae samples were then normalized to the AFDW of the analyzed algal material (e.g., mg C/g AFDW).

In 2001, alkaline phosphatase activity (APA; Sayler et al. 1979) was determined from reach wide filamentous-green algae composite samples and reach wide diatom-dominated composite samples. This method detects the presence of phosphatase enzymes, which are generally thought to indicate conditions of P limitation (Pick 1987). Due to budgetary considerations and the strong indication of N limitation in stream sites sampled in 2001, the method was not used at the 2002 sites.

2.3.5 Soft-Bodied Algae Identification and Diatom Community Metrics

Benthic samples were collected and preserved for identification of both soft-bodied algae and diatoms. All samples were preserved in the field using Lugol’s solution (MT DEQ 1999a). Samples were collected in 2001 once in June and once in late August/early September, and were grouped into one of four groups: “composite” samples, “hoop” samples, “sediment” samples, and “template” samples. Each of these sample groups was composited from up to 11 transects in the reach, according to the type of category each transect represented. All samples were qualitative in nature as the surface area of the stream bottom was not measured during collection. The “composite” samples were collected from a variety of substrates present in a reach, and included filamentous algae, macrophytes, fine sediments, and scrapings from gravel bottomed transects. “Hoop” samples were composited exclusively from transects where the hoop method was used (see Section 2.3.4 above), and were composed of filamentous algae and macrophyte clippings. “Sediment” samples were composited from sediment core transects, while “template” samples were composited from rock scrapings. In 2002 samples were collected in June, July, and August, and procedures and composite groups were identical to those described for 2001 with the exception that the “composite” samples (those combining hoop, sediment, and template-type samples together) were not collected.

All samples were submitted to *Hannaea* in Helena, MT for identification of the soft-bodied algae and diatoms. Soft-bodied algae were identified and their relative abundances recorded. Proportional counts of diatoms (minimum count of 600 valves) were made and various diatom-based metrics were calculated. A metric is a quantitative summary of some aspect of the structure and function of a biotic community that changes in some predictable way (Barbour et al. 1999). The reader is referred to Bahls (2002 and 2003) for detailed descriptions of counting methods and metrics development. All samples were supplied to *Hannaea* blind; that is, without accompanying assessments of the sites' physical or chemical conditions. The intent of this approach was to ensure an unbiased and independent assessment of the streams' conditions from a nutrient impairment perspective, based only on metrics of the diatom communities.

2.3.6 Aquatic Macroinvertebrate Identification and Community Metrics

Samples of benthic macroinvertebrate populations were collected using a 500 µm mesh D-frame kicknet and preserved in ethyl alcohol according to Western Pilot EMAP protocols (USEPA 2000b). Whenever possible, a reach wide sample and a targeted riffle sample were both collected. The reach wide sample was a composite collected from the same transects sampled for algae (unless a transect was dry) using the sampling procedure described in Section 2.3.4, paragraph one. If a riffle or riffles existed in the reach, a targeted riffle sample was collected and was comprised of eight individual kick samples collected in the riffle(s), and then composited. Sites were sampled once during each of the two field seasons. Samples were collected in June in 2001, while in 2002 they were collected in July. All samples were collected within the summer index period prescribed by MT DEQ (MT DEQ 1999a).

All aquatic macroinvertebrate samples were submitted to *Rhithron* Associates, Inc. for identification, biointegrity-metrics development, and reporting. Subsamples of at least 500 organisms were counted from each sample. The reader is referred to Bollman (2002a), Bollman (2002b), and the addendum to the 2002b report (Bollman 2003) for detailed descriptions of counting methods and metrics development. As was the case for the algal community samples, samples were submitted blind in order to ensure an independent assessment of biological condition based on the aquatic macroinvertebrate communities. The 21 sites were also ranked by quality, based on a set of MT DEQ macroinvertebrate metrics for prairie streams (Bukantis 1998) using both the composite samples and the targeted riffle samples (See Fig. 5; Bollman 2003).

2.4 Statistical Analyses

In the vast majority of cases nonparametric inferential statistics are used to draw conclusions about populations and examine relationships between parameters. Nonparametric statistics do not require the assumption that the data be normally distributed. They are often more powerful than parametric methods if the assumptions behind the parametric approaches are not true, and only slightly less effective when the assumptions behind the parametric approaches are true (Hollander and Wolfe 1973; Conover 1999). The nonparametric Mann-Whitney test is used to determine if two datasets have different medians, and the Spearman rank correlation is used to assess the strength and significance of correlations between different parameters (Conover 1999). Cluster analysis (Everitt et al. 2001) is used in Section 6.0 to place stream sites into functional groupings. Two parametric statistics, least-squares regression and the coefficient of variation (r^2), are used in a few instances when presentation of a line equation was thought to be valuable. Probabilities (p-values) less than or equal to 0.05 are considered significant in this study.

Both nonparametric and parametric descriptive statistics are used to summarize and present data. The descriptive statistics selected were chosen based on the ability of each to best convey the summarized information to the reader. Medians diminish the influence of unusually high or low values in a dataset, whereas such values are incorporated into calculations of means and can provide a very different view of the data's central tendency.

SECTION 3.0
SITE RANKINGS AND IDENTIFICATION OF REFERENCE & MINIMALLY IMPACTED SITES

3.1 Site Conditions Based on Combined Qualitative Assessments

Locating reference sites along the Hi-line was anticipated from the outset to be difficult, due to the extensive use of the area for crops and grazing and other widespread changes to area streams from hydrologic modification. It was originally expected that sites would be clustered at each end of a condition gradient. However, the sites' actual conditions, based upon their qualitative field assessment scores, were not clustered at each end but rather the bulk of the sites fell into the middle range of condition values (Fig. 3.1). The site with the lowest average condition was Willow Creek (S), while the site in the best overall condition was Clear Creek.

An average stream habitat assessment score for each site was derived by averaging scores from EPA's rapid habitat assessment form, MT DEQ's stream reach assessment form, and EMAP's riparian disturbance metric (W1_HALL; Kaufmann et al. 1999). Prior to averaging the scores of the three assessment methods, each assessment method's scoring system was normalized to a 0-100 point scale. Raw scores for EMAP's W1_HALL metric range from 0.67 (no impacts) to 3 (severe impacts; Dave Peck, personal communication), while raw scores for the other two assessment methods ranged from zero (severe impacts) to some maximum value representing no impacts.

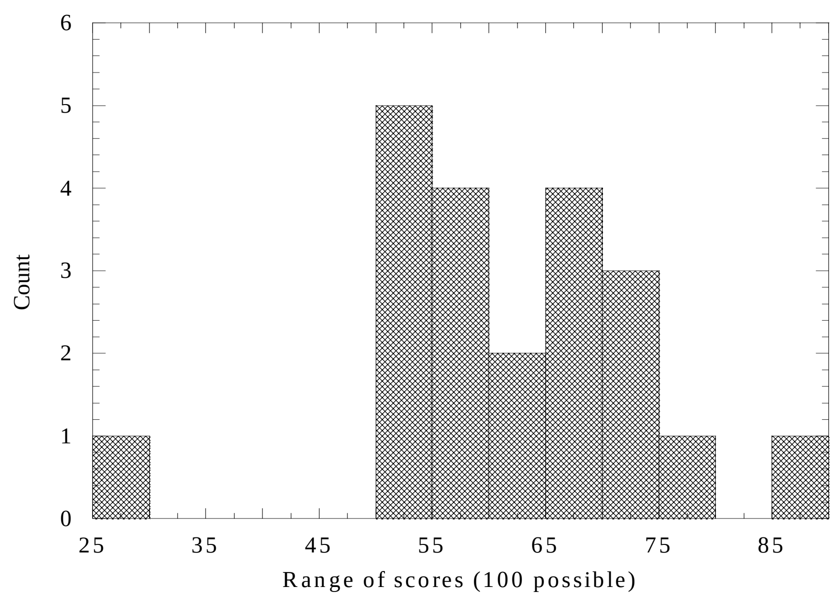


Fig. 3.1. Histogram of average stream condition for all study sites. Scores were derived from 3 assessment methods; EPA's rapid habitat assessment, MT DEQ's stream reach assessment, and EMAP's riparian disturbance metric (see text for details).

3.2 Identification of the Reference and Minimally Impacted Sites

Different means of assessing stream habitat quality and water quality can sometimes provide the investigator with contradictory results. This is especially true for prairie streams, which have been observed by MT DEQ staff (and others) to be unusually difficult to assess. This is due to the fact that most qualitative assessment procedures appear to be biased towards high gradient, forested, gravel-bottom streams (Stauffer and Goldstein 1997), and the expected biological condition of prairie stream systems, especially among the lower trophic levels, is not well described (Mathews 1988).

Given these difficulties, the technique selected to identify reference and minimally impacted sites in this study was a multiple-indicator, weight-of-evidence approach. Three categories were used to develop an overall ranking of the stream sites. These were 1. stream habitat quality based on the average of three assessment procedures (discussed in Section 3.1) 2. diatom community metrics sensitive to nutrient loading (Bahls 2003) and 3. fishery quality based on an index of biotic integrity (IBI) developed for Montana prairie stream fishes (Bramblett et al. 2003). Individual scores that each site received (ranked from best to worst) for each of these categories are shown in Figs. 3.2, 3.3, and Table 3.1.

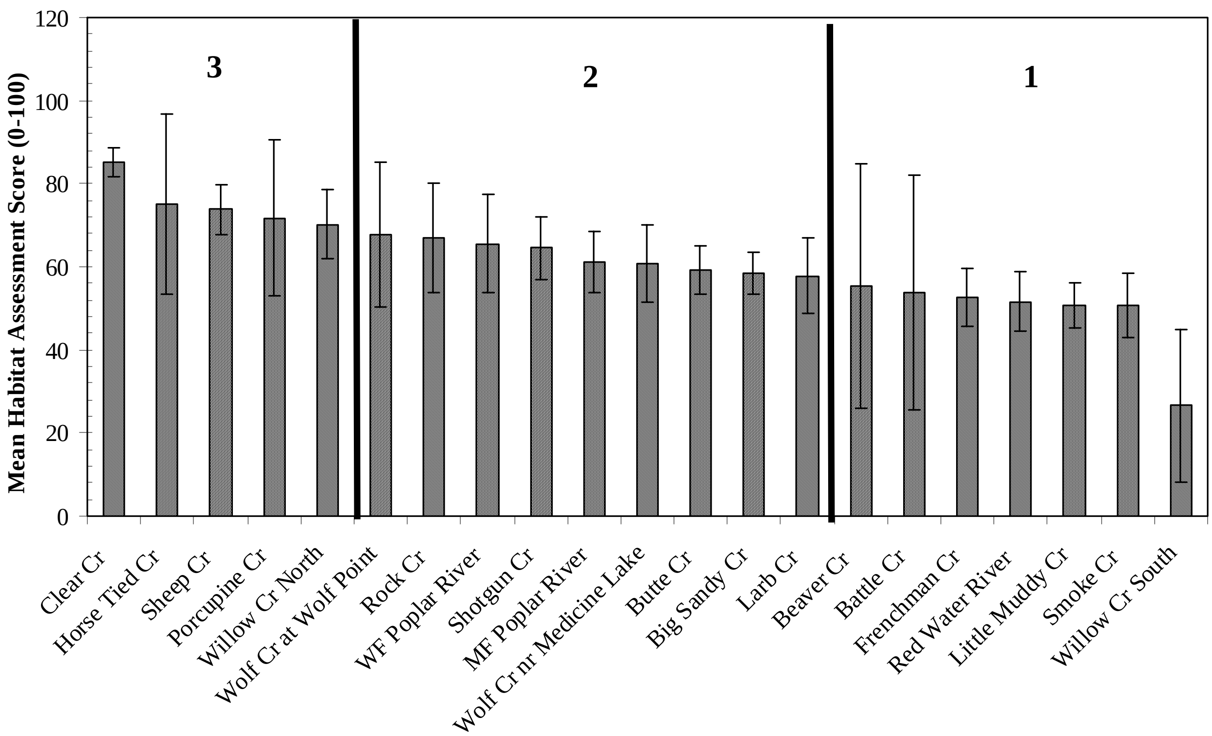


Figure 3.2. Mean habitat assessment scores based on EPA's RHA form, MT DEQ's stream reach form, and EMAP's riparian disturbance metric (W1_HALL; Kaufmann et al. 1999). All assessment methods were normalized to a 0-100 scale. Error bars equal 1 sd.

The diatom metrics used in this study were selected because they measure the diatom community's response to organic and inorganic nutrients, primarily as nitrogen and phosphorus (Fig. 3.3; Bahls 2002). Note in Fig. 3.3 that higher metric scores indicate higher nutrient loading. Inclusion of the diatoms as an evaluation category to develop final site rankings also addressed the issue of cumulative nutrient impacts, as diatom communities will have developed in response to both local impacts and impacts from upstream of the sites.

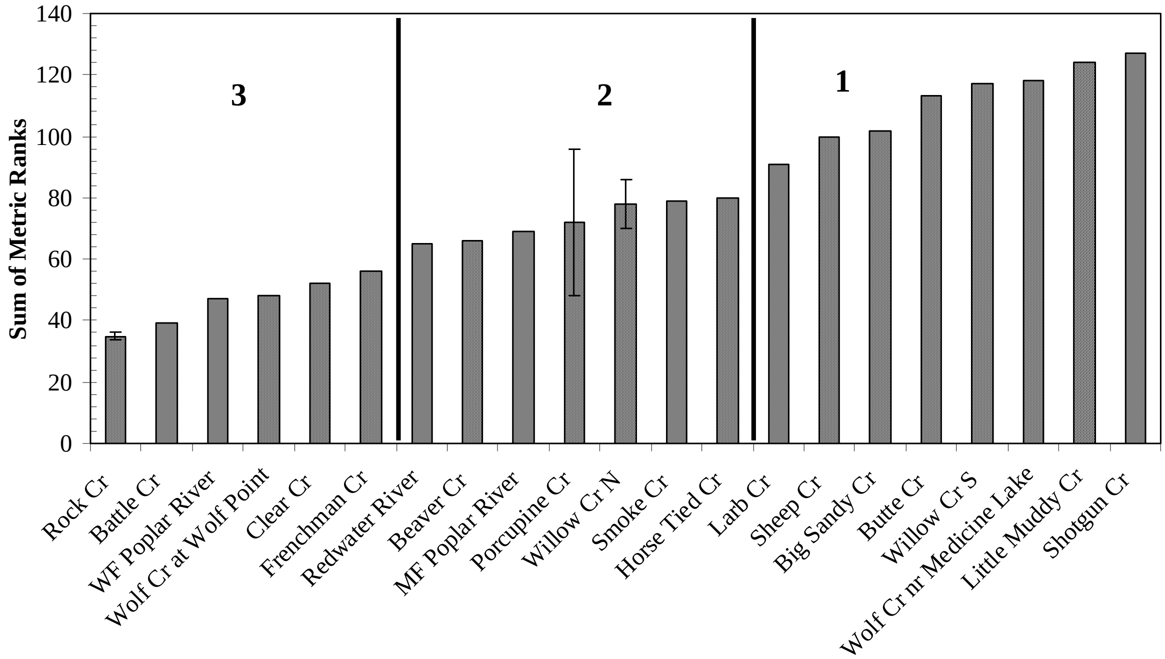


Figure 3.3. Site rankings based upon diatom community metrics, listed in order from least to most impacted by nutrients, 2001-2. Rock, Porcupine, and Willow creeks were sampled in both years; error bars are 1 standard error of the mean.

By design, seven of the present study's sites where identical to sites used in developing a fish IBI for Montana and are shown in Table 3.1. The fish IBI scores for each site were obtained (Bramblett, personal communication) and then compared to the range of IBI scores corresponding to good, fair, and poor quality sites of the Bramblett et al. (2003) study (see Figure 2 therein). Good, fair and poor site condition of sites are determined in the Bramblett et al. (2003) study using a human-influence index (HII) comprised of factors such as stream incision, stream substrate, percent fish cover, riparian cover, nutrient concentrations, and land uses in the sites' basins.

The ranked scores for each of the individual assessment categories were then divided into three ranges; the upper, middle, and lower third of the ranked distribution. If a natural break was obvious in the rankings, the break point was adjusted up or down slightly. The sites in the upper third of the range were assigned a range value of 3, those in the middle 2, and those in the lower third of the range a 1 (see Figs. 3.2, 3.3). For the fish IBI, 3, 2, or 1 (Table 3.1) corresponded to the good, fair, or poor site-condition categories described by Bramblett et al. (2003). Range values from the habitat, diatom, and fish categories were then compiled and used to develop the final rankings (Table 3.2).

Table 3.1. Range values for sites based on their fish IBI. The stream sites shown were located at the same sites used by Bramblett et al. (2003) to develop the IBI.

Site	Fish IBI score	Range Value*
Wolf Creek at Wolf Point	85	3
Willow Creek (N)	68	2
Willow Creek (S)	55	2
Butte Creek	55	2
Smoke Creek	49	1
Porcupine Creek	42	1
Big Sandy Creek	10	1

*Each site's fish IBI score was compared to the IBI scores in Bramblett et al. 2003. Median and interquartile range (in parenthesis) of the IBI scores of good, fair and poor condition sites from the Bramblett et al. study have the values shown below:

Good Condition Sites (3):	73 (67-79)
Fair Condition Sites (2):	66 (75-50)
Poor Condition Sites (1):	50 (27-61)

Table 3.2. Final determination of stream condition rankings, based on multiple assessment methods. Sites that consistantly scored in the upper third of the distribution of each assessment category are considered reference, and are shown in bold.

Study Year	Stream Site	Habitat : Range value from ranked scores	Nutrient loading based on diatoms: Range value from ranked metric scores	Prairie fish IBI: range value from IBI scores	Sum of range values	Maximum Possible	**FINAL RANKING SCORE (% of max)**	Stream Condition Group
2001	**Clear Cr**	3	3	no data	6	6	**100**	Reference
2002	Wolf Cr at Wolf Point	2	3	3	8	9	89	Minimally impacted
2002	West Fork Poplar River	2	3	no data	5	6	83	Minimally impacted
2001	Rock Cr	2	3	no data	5	6	83	Minimally impacted
2002	Horse Tied Cr	3	2	no data	5	6	83	Intermediate
2001	Willow Cr (N)	3	2	2	7	9	78	Intermediate
2001	Porcupine Cr	3	2	1	6	9	67	Intermediate
2002	Sheep Cr	3	1	no data	4	6	67	Intermediate
2001	Battle Cr	1	3	no data	4	6	67	Intermediate
2002	Middle Fork Poplar River	2	2	no data	4	6	67	Intermediate
2001	Frenchman Cr	1	3	no data	4	6	67	Intermediate
2002	Butte Cr	2	1	2	5	9	56	Intermediate
2002	Wolf Cr nr Medicine Lake	2	1	no data	3	6	50	Most Impacted
2001	Larb Cr	2	1	no data	3	6	50	Most Impacted
2001	Beaver Cr	1	2	no data	3	6	50	Most Impacted
2002	Shotgun Cr	2	1	no data	3	6	50	Most Impacted
2002	Smoke Cr	1	2	1	4	9	44	Most Impacted
2001	Willow Cr (S)	1	1	2	4	9	44	Most Impacted
2001	Big Sandy Cr	2	1	1	4	9	44	Most Impacted
2002	Redwater River	1	2	no data	3	9	33	Most Impacted
2002	Little Muddy Cr	1	1	no data	2	6	33	Most Impacted

The final ranking scores were divided into approximately the upper, middle and lower third of the ranked distribution which corresponded to three condition classes: reference & minimally impacted (comparison); intermediate, and most impacted. Natural breaks in the distribution were used to make slight adjustments; obvious breaks occurred between Willow N and Porcupine creeks, and between Butte Creek and Wolf Cr nr Medicine Lake (Table 3.2). Reference sites were considered those that consistently ranked in the upper third of all of the individual assessment categories (only Clear Creek qualified). Other sites in approximately the upper third of the final rankings were considered candidates for "minimally impacted", however Horse Tied and Willow N creeks were ultimately downgraded to "intermediate" condition because of specific impacts. Horse Tied Creek received intense and very localized nutrient loading from cattle use (this will be discussed in Section 8.0), while Willow Creek N appeared to have streambed stability problems and also had a major invasion by noxious weeds (leafy spurge, *Euphorbia esula*).

Characteristics from sites in the reference & minimally impacted condition class are used throughout the remainder of the study as points of comparison (comparison sites) to other study sites. Two of the minimally impacted sites (West Fork Poplar River and Rock Creek) were also identified as candidate reference streams in an earlier, but less extensive, study of Hi-line region streams (Table 3.2; Bahls et al. 1992).

In addition to the overall ranking process shown in Table 3.2, other variations on the ranking process were attempted. These included: the inclusion of aquatic macroinvertebrate metric scores along with the fish IBI, habitat scores and diatom metrics; ranking the streams using *only* the habitat scores and the diatom metrics; and ranking the streams using habitat scores, the fish IBI and aquatic macroinvertebrate metrics, but not the diatom metrics. In all of the variations of the ranking process, the same four streams shown as reference or minimally impacted in Table 3.2 emerged as the highest quality sites. Similarly, four of the streams in the lower third of the composite score rankings (Little Muddy, Big Sandy, Willow (S), and Shotgun creeks; Table 3.2) always appeared in the lower third of all the ranking variations. Streams in the midrange of the rankings tended to shift around somewhat. Photographs of the reference and minimally impacted stream sites are shown in Figs. 3.4 through 3.7.

Figure 3.4. Wolf Creek at Wolf Point, Roosevelt County, Montana. June 2002.

Figure 3.5. West Fork of the Poplar River, Daniels County, Montana. July 2002.

Figure 3.6. Clear Creek, Blaine County, Montana. August 2001.

Figure 3.7. Rock Creek, Valley County, Montana. July 2002.

SECTION 4.0
GEOMORPHOLOGY, SOILS, AND THE RIPARIAN AREA

4.1 Stream Classification and Morphological Characteristics of the Sites

Detailed measurements of stream morphology characteristics of each site are shown in Table 4.1. The Rosgen stream channel classification system (Rosgen 1996) was used to classify each stream site. The Rosgen system has been found to work well in other regions having low-relief terrain (Savery et al. 2001) similar to that found along the Hi-line. Three of the four top ranked streams were C channels, which are noted for well-developed floodplains and are fairly sinuous. The most highly ranked minimally impacted site (Wolf Cr at Wolf Point) was an F channel. F channels are noted for their deep incisement (they are downcut), high bank erosion potential and accelerated channel aggradation and/or degradation (Rosgen 1996). Wolf Cr at Wolf Point was observed to be developing point bars, indicating that a functional floodplain is starting to become reestablished within its confining channel (Rosgen 1996). Note that the G ("gully") stream types, known for their deep incisement, instability, and high erosion rates, tend to be clustered in the most impacted condition class or near the lower end of the intermediate condition class. Most of the streams that had a substrate D_{50} of 0.06 mm or less (silt/clay) are clustered in the most-impacted condition class (Table 4.1).

Table 4.1. Geomorphic measurements made at each site, and final stream-type classification. Numbers shown in italics were derived from EMAP data (see explanation in text).

Stream Site Name	Condition Class	Entrenchment Ratio*	Width/Depth Ratio†	Sinuosity	Water Surface Slope (%)	D_{50} of stream substrate (mm)	Rosgen Stream Type*
Clear Creek	Reference	7.91	*12.8*	1.16	1.34	*40.00*	**C4**
Wolf Cr at Wolf Point	Min. impacted	1.32	26.2	1.50	0.23	0.19	**F5**
West Fork Poplar River	Min. impacted	7.00	20.7	1.83	0.30	*40.00*	**C4**
Rock Creek	Min. impacted	2.22	24.0	1.69	0.30	13.65	**C4**
Horse Tied Creek	Intermediate	1.41	28.6	2.21	0.14	0.06	**F6**
Willow Creek (N)	Intermediate	1.18	*18.2*	1.52	0.58	*1.03*	**F5**
Porcupine Creek	Intermediate	1.98	*26.3*	1.06	0.40	*40.00*	**C4**
Sheep Creek	Intermediate	1.67	58.1	1.46	0.70	0.06	**B6c**
Battle Creek	Intermediate	1.54	*8.3*	1.09	0.38	*9.00*	**G4c**
Middle Fork Poplar River	Intermediate	7.16	22.1	1.87	0.20	0.38	**C5**
Frenchman Creek	Intermediate	1.24	*12.6*	1.07	0.31	*1.03*	**G5c**
Butte Creek	Intermediate	6.74	30.1	2.24	0.30	3.00	**C4**
Wolf Cr nr Medicine Lake	Intermediate	no data	*4.9*	1.11	0.52	*0.06*	*probably G6*
Larb Creek	Most impacted	1.15	*8.5*	1.29	0.33	*1.03*	**G5c**
Beaver Creek	Most impacted	1.28	*8.5*	1.02	0.22	*0.06*	**G6c**
Shotgun Creek	Most impacted	1.55	30.4	1.06	dry	0.06	**F6**
Smoke Creek	Most impacted	1.92	7.1	2.20	0.70	4.85	**E4**
Willow Creek (S)	Most impacted	1.4	*8.5*	1.07	0.13	*0.06*	**G6c**
Big Sandy Creek	Most impacted	1.91	*12.2*	1.04	0.11	*0.06*	**C6**
Redwater River‡	Most impacted	no data	*22.0*	1.11	0.38	*1.03*	**C5**
Little Muddy Creek	Most impacted	1.45	34.3	1.80	0.06	0.06	**F6**

* Rosgen (1996).

† Ratio calculated (per Rosgen 1996) as: bankfull width/mean bankfull depth.

‡ Entrenchment ratio was not measured at this site, but was measured at a similar Redwater River site downstream where the entrenchment ratio =7.1. The stream type classification of C5 is quite confident, based on overall site observations and the quantitative measurements that were taken.

Two physical characteristics (width/depth ratio and substrate D_{50}) in Table 4.1 have some of their values shown in italics; these were derived from EMAP data[1]. EMAP methods measure bankfull mean depth and substrate D_{50} in a manner different than that recommended by Rosgen (1996) and there was a concern that these differences could alter the identification of a Rosgen channel type. In 2002 a number of streams were selected for side-by-side measurements of both EMAP and Rosgen methodologies, the details of which are discussed in Appendix A.

It resulted that in some cases the EMAP methods could bias results and lead to the identification of a Rosgen channel type differently than if Rosgen methods had been used. Given these methodological issues, the stream types shown in Table 4.1 were very carefully determined in cases where EMAP data was employed. During multiple field visits to each stream site, the Rosgen stream type (including sediment class) was visually estimated and careful notes were taken. The final stream types in Table 4.1 were derived not only from the quantitative data shown, but also from additional field notes and observations.

4.2 Relationship between Stream Morphology and Basin Soil Structure

Originally formed under the mixed prairie of the Hi-line region were the Mollisol soils, which typically form in grasslands of drier climates that have limited leaching (Eyre 1968; Brady and Weil 2002). During the study, landowners commented that certain streams tended to be very "flashy", demonstrating periodic short-term high flows followed by periods of intermittency. To follow up on this, we looked at the relationship between the stream channel types and the soil characteristics that are dominant in each stream's respective basin. John Lhotak, graduate student with the University of Montana who collected much of the field data in 2001, compiled the soils data that were electronically available for the region.

Soil was assessed using the Soil Survey Geographic Database (SSURGO) soil data and the Soil Data Viewer, developed for ArcView 3.2 by the Natural Resource Conservation Service (NRCS). For those watersheds extending into Canada, soil information for SSURGO was not available and hence analyses were restricted to the U.S. side (Lhotak 2004). The soil-type data categories that were available included hydrologic group, depth to soil restrictions, SAR, percent clay, and permeability class. Some of these categories are closely related to one another (e.g., permeability class, hydrologic group, and percent clay). The category that was expected to be most relevant to this study was the soil permeability class. Soil permeability classes are described in the Natural Resource Conservation Service (NRCS) National Soil Survey Handbook (USDA 2002). Soil permeability is the quality of the unvegetated soil that enables water or air to move through it (USDA 2002). The NRCS has broken soils into eight different narrative classes of permeability (measured in µm/sec), ranging from very rapid (> 141 µm/sec) to

[1] In the streams sampled in 2001, sinuosity and slope were calculated using only EMAP methods (SINU and XSLOPE metrics; Kaufmann et al. 1999). In 2002 sinuosity and slope where measured in ten streams using both EMAP and Rosgen methods. In all ten streams, data from both method were similar and resulted in the same Rosgen stream type.

impermeable (< 0.01; Table 4.2). Using GIS, the proportional area of each of the eight permeability classes was determined for basins in which a study stream site was located. Site basins were delineated as combinations of 5th field HUCs that best captured the immediate drainage area of each study stream (Fig 4.1). The error within these determinations was quite reasonable, as the sum of the areas for all permeability classes in any given basin ranged from 98.7% (Horse Tied Creek) to 110.6% (MF Poplar River). A soil analysis was not completed for the Redwater River or for Sheep Creek, as the data layer for McCone County is not yet available from the NRCS. Basins that had high proportions of permeable soils were found in both the Milk and the Lower Missouri sub-major basins; Fig. 4.2 shows each site's Rosgen class and the types of soils dominant in the region.

Table 4.2. Soil permeability classes. From USDA (2002).

	Class Value	
Permeability Class	**(inches/hr)**	**(µm/sec)**
Very rapid	>20	>141
Rapid	6-20	42 - 141
Moderately rapid	2 - <6	14 - 42
Moderate	0.6 - <2	4-14
Moderately slow	0.2 - <0.6	1.4 - 4
Slow	0.06 - <0.2	0.42 - 1.4
Very slow*	0.0015 - <0.06	0.01 - 0.42
Impermeable*	0.00 - <0.0015	0.00 - 0.01

*The limit of 0.0015 approximates the limit of 1 foot/ year used by EPA and NRCS engineers for impermeable conditions for manure holding ponds.

To simplify the analysis of relationships between soil permeability and various geomorphic stream characteristics (e.g. entrenchment ratio, sinuosity, slope, etc.), the eight soil permeability classes were combined into just two groups. The first group was composed of the four "slow" soil categories classified as "impermeable" to "moderately slow", encompassing a range of soil permeabilities from 0.00 to 4.0 µm/sec. The second group was composed of the "faster" permeable soil categories whose permeability ratings ranged from "moderate" to "very rapid" (4.0 to >141.0 µm/sec; Table 4.2).

Soil permeability was found to be correlated to the entrenchment ratio of the study sites. Entrenchment is the vertical containment of a stream and the degree to which it is incised into the valley floor (Kellerhals et al. 1972), and is quantified here as the entrenchment ratio (Rosgen 1996). Fig. 4.3 shows the relationship between the "slow" soil permeability group and the entrenchment ratio of the study sites. The relationship between entrenchment ratio and the "slow" soils is significant ($P \leq 0.05$) using Spearman rank correlation ($r_s = 0.65$, $P = 0.003$). The best-fit line (based on R^2 values) to the data is curvilinear and is shown in Fig. 4.3, and suggests that soil permeability may be having a threshold effect on entrenchment ratio. This is further demonstrated in Table 4.3, where the streams have been sorted by the percent of "slow" soils in each basin. All of the streams in basins having approximately 90% or more of the "slow" soils have low entrenchment ratios (< 1.6) and are classified as F or G channels. On the other hand, all of the stream sites with high entrenchment ratios (> 2.4) are located in basins where "slow" soils represent less than 60% of the total, and are classified as C channels (Table 4.3). These data, considered along with the soil permeability-entrenchment ratio curve in Fig. 4.3, suggest that stream sites located in basins having a high proportion (>90%) of the "slow" soil types are likely to be vulnerable to entrenchment.

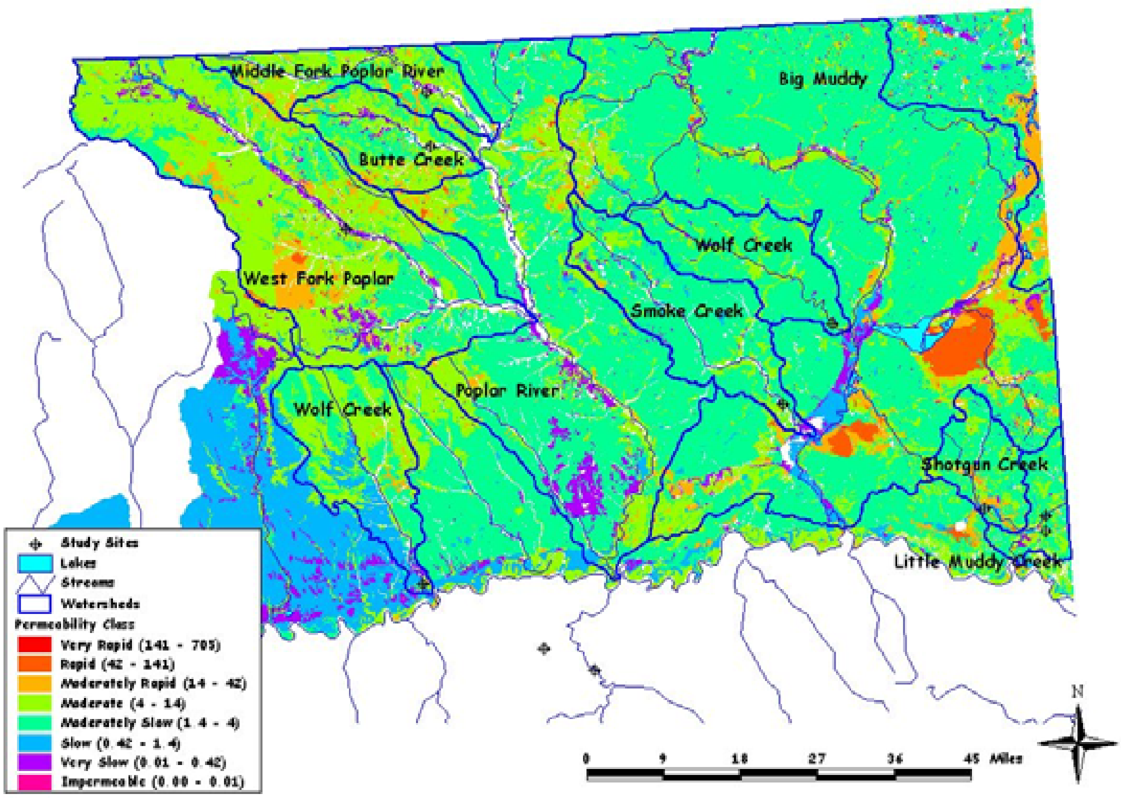


Figure 4.1. Basin delineation outlines for study stream sites. Basin delineations shown here in blue are for the Lower Missouri sub-major basin; delineated basins of a similar nature were developed for stream sites in the Milk sub-major basin. The proportion of each of the soil permeability types (NRCS 2002) was calculated for each basin.

One explanation for the patterns discussed above is that they are the result of the interaction of local soils, plant cover, and rainfall & climate patterns in the region. The regional climate is semiarid and has extreme temperature variation, which has resulted in the development of a semiarid grassland (Weaver and Albertson 1956; Covich et al. 1997). During dry periods the vegetated cover on soil can be diminished and erosion and valley degradation takes place, due to the tendency towards sporadic heavy rains and the infrequency of small, light rainstorms (Leopold 1994); such changes can dramatically affect stream channels. (Poor land use practices such as over grazing can also diminish vegetated cover of soil-more on this below.) In basins that have lost adequate plant cover and have high proportions of slow permeability soils (such as soils high in clay), water

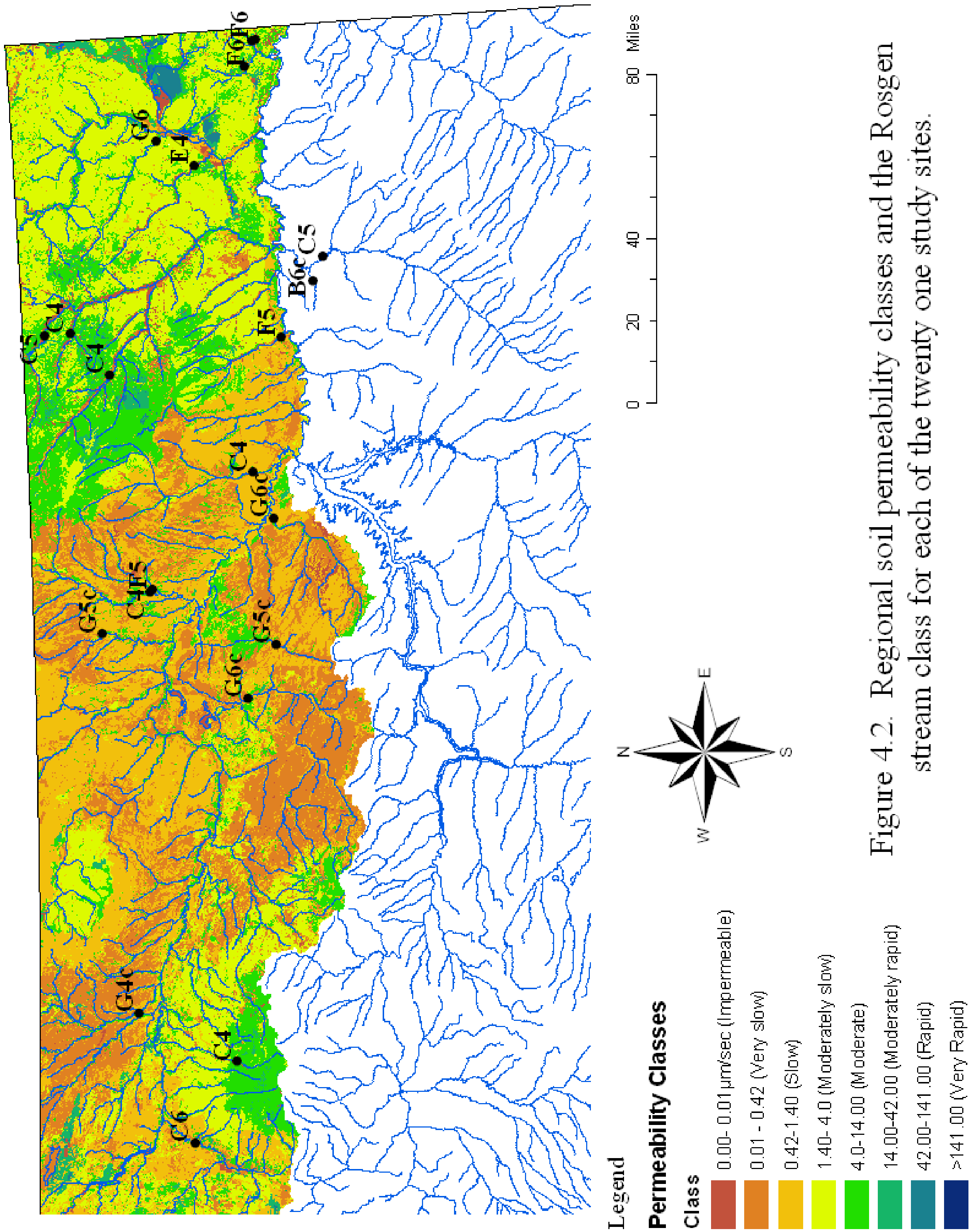


Figure 4.2. Regional soil permeability classes and the Rosgen stream class for each of the twenty one study sites.

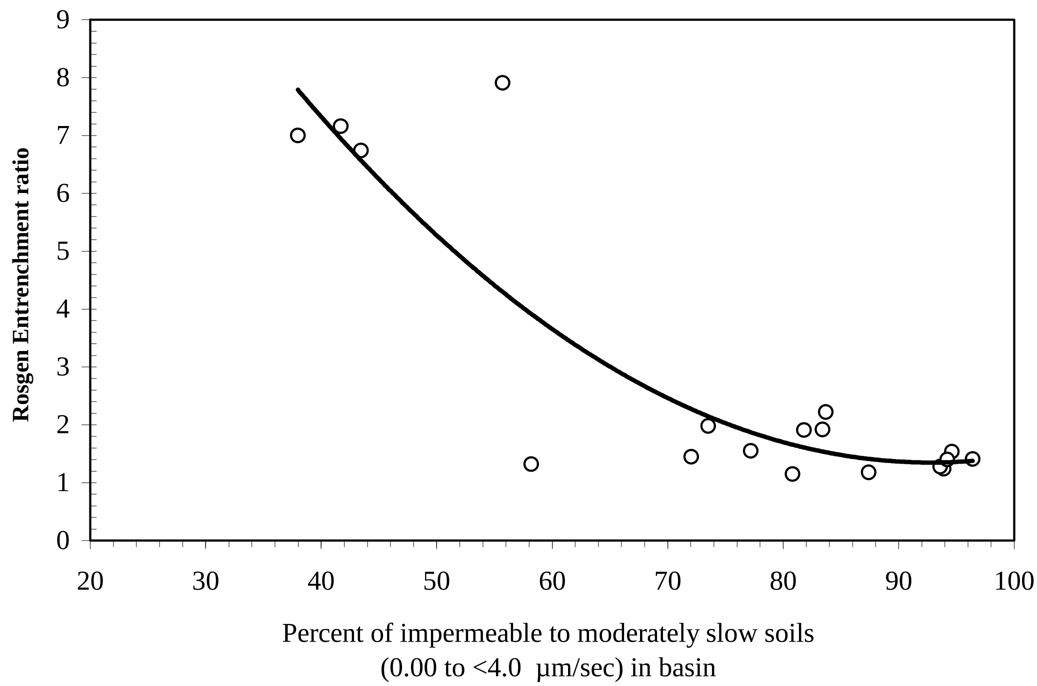


Figure 4.3. Relationship between soil permeability of stream basins and the Rosgen entrenchment ratio of study sites. Relationship is a second order polynomial ($R^2 = 0.78$).

Table 4.3. Study stream sites sorted by the percentage of "slow" permeability soils in each basin.

Stream Site Name	"Slow Soils"-sum of basin area having soils with permeability rates from 0.0 to 4.0 µm/sec (%)	"Faster Soils"-sum of basin area having soils with permeability rates from 4.0 to >141.0 µm/sec (%)	Rosgen Entrenchment Ratio	Rosgen Stream Type
West Fork Poplar River	38	71	7.00	**C4**
Middle Fork Poplar River	42	69	7.16	**C5**
Butte Creek	43	67	6.74	**C4**
Clear Creek	56	47	7.91	**C4**
Wolf Cr at Wolf Point	58	44	1.32	**F5**
Little Muddy Creek	72	30	1.45	**F6**
Porcupine Creek	74	28	1.98	**C4**
Shotgun Creek	77	31	1.55	**F6**
Larb Creek	81	19	1.15	**G5c**
Big Sandy Creek	82	27	1.91	**C6**
Smoke Creek	83	17	1.92	**E4**
Rock Creek	84	16	2.22	**C4**
Wolf Cr nr Medicine Lake	84	16		probably G6
Willow Creek (N)	87	13	1.18	**F5**
Beaver Creek	94	7	1.28	**G6c**
Frenchman Creek	94	6	1.24	**G5c**
Willow Creek (S)	94	6	1.4	**G6c**
Battle Creek	95	11	1.54	**G4c**
Horse Tied Creek	96	2	1.41	**F6**

from heavy rainfall is more apt to flow overland to the nearest channel then in basins with a high proportion of faster permeability soils. The increased flashy flows in the low permeability basins may in turn result in increased stream incisement.

Others have reported results that support this hypothesis. In a study in Michigan, 49% of the variability in flood ratio (analogous to entrenchment ratio) is explained by stream slope and the lacustrine clay content of the stream basins (Richards et al. 1996). In a study of stream runoff from paired rural basins in Illinois, the basin with clayey soils and minimal wetlands (low permeability basin) has higher runoff floods then the basin that has sandy soils and more extensive wetlands (high permeability basin; Demissie et al. 1983; Changon and Demissie 1996). Under the Hi-line region’s glacial till shale is quite common (Alt and Hyndman 1986), and shale typically weathers to clayey residues that have reduced permeability relative to other soils (Brady and Weil 2002).

Vegetation, either living or dead, must be kept on the soil to prevent raindrops from hitting bare soil and loosening the particles, which can then clog soil pores and decrease infiltration and lead to increased runoff and erosion (Weaver and Albertson 1956). The conversion of prairie lands to agriculture has generally led to decreased soil infiltration, accompanied by channel incision and headcut erosion (Prestegaard 1988). For example, in the loess hills of Iowa the breaking of the sod was followed by valley trenching to a depth of 100 ft (Hart 1957). And natural factors such a drought exacerbate these effects. Drought occurs commonly throughout the Northern Great Plains (see sub-Section 1.2) and results in several interacting factors. During the great drought in 1934, for example, a study on moderately grazed lands near Miles City revealed that up to 60% of the plants were killed back to the ground and just one percent of the original plant volume was replaced the next year (Weaver and Albertson 1956), leaving the soil exposed to much greater erosion potential. Often accompanying drought in Montana are grasshopper infestations. During a drought in 1936 near the Little Powder River, grasshoppers devoured most forms of the already-stressed vegetation (including leaves and bark of the sage brush), leaving little vegetative cover to protect the soil when the rains returned (Weaver and Albertson 1956). Thus loss of vegetative cover, regardless of the cause, leaves exposed soil vulnerable to increased erosion and higher runoff potential.

Over grazing can also lead to loss of soil permeability and increased runoff through both soil compression and vegetation loss. This phenomenon certainly has relevance along the Hi-line, where grazing is a major land use. Rauzi and Hanson (1966) showed in a carefully controlled study on mixed prairie in South Dakota that water intake rates of soil are nearly linear, with a heavily grazed watershed having the lowest water intake, a moderately grazed watershed intermediate intake, and a lightly grazed watershed the highest intake. Further, annual runoff is greatest and runoff-producing precipitation events occur more often from the heavily grazed watershed compared to the lightly grazed watershed.

All of these factors–reduced plant cover, greater direct exposure of soil, low inherent soil permeability, reduced water infiltration, and precipitation pattern changes–work together towards increasing erosion and runoff. Sudden, intense flows carried by streams in basins where the soil permeability is low can therefore lead to periodic episodes of incisement and headcut erosion. On the other had, basins having a significant proportion of faster permeability soils are more likely to absorb more of the water from heavy rainfall events, water that can then reenter the stream channels later as groundwater flow.

In conclusion, permeability characteristics of Hi-line soils are variable and once the vegetative cover is lost or diminished and the soils are exposed, streams in low soil-permeability basins are probably more vulnerable to flashy flows, incisement and headcutting than those in basins having more permeable soils. The implication of this is that if stream incisement is to be controlled, land use practices that affect the vegetative cover (e.g. grazing) and the soils' ability to absorb precipitation need to be carefully applied and monitored in basins that have high proportions of low permeability soils.

4.3 The Riparian Area

4.3.1 Habitat Types, Community Types, and Riparian Descriptions

There was a diverse array of riparian habitats observed at the twenty-one study sites. Riparian habitats ranged from those dominated by sedges and rushes but having no trees and few shrubs (West Fork Poplar River, Sheep Creek), to climax deciduous forests that were comprised of green ash, boxelder, and American elm (Horse Tied Creek). The two most common riparian ecosystems in the study sites were the "Shrub and Herbaceous

Riparian Complex" and the "Recent Riparian Complex" (pages 35 and 36, respectively; Hansen et al. 1995).

The conditions of the streams' riparian areas were assessed using one of two procedures developed in Montana (see Section 2.3.1) and are shown in Table 4.4. The riparian assessment scores in Table 4.4 did not always agree with the overall site condition rankings of the sites that were developed in Section 3.0, however riparian condition was only one small part of the assessment package that was used to derive the overall stream rankings. The riparian areas of the stream sites were classified to their habitat and community type as accurately as possible using Hansen et al. (1995). Various guides were used to identify plants, including Kershaw et al. (1998) and Little (1980). Because each site had certain unique riparian characteristics, a brief summary of each site is provided below; the habitat/community types are shown in parenthesis following the site name. Some of the comments are summarized from Lhotak (2004).

<u>*Clear Creek*</u>: (Intermittent Riparian Coulee, Yellow Willow/Beaked Sedge). This site had one of the best overall riparian conditions (note score of 95% and 'sustainable') in the study. The site had active beaver ponds and a thick riparian corridor of yellow willow (*Salix lutea*) and beaked sedge (*Carex rostrata*). Agriculture composed only about 7% of the site's upstream watershed. However this site is also the least "prairie" like in the study, as it is located in the foothills of the Bear Paw Mountains and has influences from both of the ecoregions on whose borders it is located (the Northwestern Glaciated Plains and the Middle Rockies; Omernick 2000).

<u>*Wolf Creek at Wolf Point*</u>: (Recent Riparian Complex, Great Plains Cottonwood/ herbaceous). The site was developing a new riparian area inside its incised channel. Tree and woody species in the new and on the abandoned floodplain included Great Plains cottonwood (*Populus deltoides*), willow trees (*Salix spp.*) snowberry (*Symphoricarpos spp.*), common chokecherry (*Prunus virginiana*), and currant (*Ribes spp*.). The site's relatively low score on the riparian assessment (58%) was mainly driven by the stream's incisement and by stream bank instability; however, current cattle grazing activities appeared to be very minimal. Deer appeared to graze the area moderately.

<u>*West Fork Poplar River*</u>: (Shrub and Herbaceous Riparian Complex). Located in a treeless valley, the stream's immediate riparian zone was comprised of sedges (*Carex spp*.), grasses and rushes. Snowberry and woods rose (*Rosa woodsii*) were found slightly upgradient from the sedge-dominated zone. The valley appeared to be used only for cattle grazing; grazing pressure was light to moderate with some bank trampling.

<u>*Rock Creek*</u>: (Recent Riparian Complex, Great Plains Cottonwood/herbaceous). This site had downcut the past and had an abandoned floodplain that included Great Plains cottonwood, green ash (*Fraxinus pennsylvanica*), and boxelder (*Acer negundo*). The more recent floodplain had some young Great Plains cottonwood trees but few saplings, and grazing appeared to be at least moderate. There were also dense populations of the weed species leafy spurge (*Euphorbia esula*), and a bit further upgradient from the stream American licorice (*Glycyrrhiza lepidota*) was common.

Table 4.4. Estimates of riparian habitat types and riparian condition for each of the stream study sites. The riparian condition of the sites in 2001 were evaluated using a different procedure than that used in 2002 (see text of Section 2.3.1).

Stream Name	Year	Riparian Assessment Score (%)*	Condition Rating*	Habitat Type observed‡
Clear Creek	2001	95	Sustainable	Intermittent riparian coulee (yellow willow/beaked sage)
Wolf Creek at Wolf Point	2002	58	Healthy w/ problems	Recent Riparian Complex
West Fork Poplar River	2002	70	Healthy w/ problems	Shrub and Herbaceous Riparian Complex
Rock Creek	2001	62	At risk	Recent Riparian Complex (Great Plains cottonwood/herbaceous)
Horse Tied Creek	2002	68	Healthy w/ problems	Woody Draw (green ash/common chokecherry)
Willow Creek (N)	2001	70	At risk	Recent Riparian Complex (Great Plains cottonwood/herbaceous, and sandbar willow)
Porcupine Creek	2001	93	Sustainable	Recent Riparian Complex (Great Plains cottonwood/herbaceous)
Sheep Creek	2002	100	Healthy	Shrub and Herbaceous Riparian Complex
Battle Creek	2001	52	At risk	Recent Riparian Complex (western snowberry)
Middle Fork Poplar River	2002	64	Healthy w/ problems	Shrub and Herbaceous Riparian Complex
Frenchman Creek	2001	50	At risk	Recent Riparian Complex (western snowberry)
Butte Creek	2002	64	Healthy w/ problems	Shrub and Herbaceous Riparian Complex
Wolf Creek nr Medicine Lake	2002	54	Unhealthy	Shrub and Herbaceous Riparian Complex (*closest fit*)
Larb Creek	2001	40	Not sutainable	Recent Riparian Complex (western snowberry)
Beaver Creek	2001	58	At risk	Recent Riparian Complex (boxelder/common chokecherry)
Shotgun Creek	2002	100	Healthy	Oxbox/cattail marsh
Smoke Creek	2002	44	Unhealthy	Shrub and Herbaceous Riparian Complex
Willow Creek (S)	2001	36	Not sustainable	Recent Riparian Complex (sandbar willow)
Big Sandy Creek	2001	45	Not sustainable	Shrub and Herbaceous Riparian Complex (foxtail barley and woods rose)
Redwater River	2002	69	Healthy w/ problems	Recent Riparian Complex
Little Muddy Creek	2002	45	Unhealthy	Shrub and Herbaceous Riparian Complex

*From NRCS riparian assessment form (Berg and Leinard 2001) for 2001 sites, or from Fitch et al. (2001) for 2002 sites.

‡Per Hansen et al. (1995).

Horse Tied Creek: (Woody Draw, Green Ash/Common Chokecherry). This site was the only woody draw in the study. The steep banks of the stream supported a community of Green Ash, boxelder, American elm (*Ulmus americana*), common chokecherry, snowberry, and woods rose, and appeared to be the climax community for this habitat type. Due to the stream's steep banks the heavy cattle use was concentrated mostly along individual paths that crossed the creek, or in the upland area. The creek bottom itself was heavily trampled, and cattle occasionally concentrated there.

Willow Creek (N): (Recent Riparian Complex, Great Plains Cottonwood/herbaceous and Sandbar Willow). The stream had incised in recent times, but was establishing new point bars within its channel and had good recruitment of Great Plains cottonwood seedlings and sandbar willow (*Salix exigua*). One problem with the site was the extensive invasion of the weed species leafy spurge.

Porcupine Creek: (Recent Riparian Complex, Great Plains Cottonwood/herbaceous). The site flows through areas of Flaxville gravels (Alt and Hyndman 1986), and the stream channel was comprised mainly of gravel which may have aided in the retention of its present floodplain (no incisement was noted). Age diversity of plants and general recruitement on point bars was quite good at this site. Great Plains cottonwood, narrowleaf cottonwood (*Populus angustifolia*), boxelder, and sandbar willow were common. Beaver activity was heavy. The site received fairly heavy cattle grazing, and there was moderate trampling of the stream bank. In spite of the heavy grazing, it received a score of 93% on its riparian assessment (Table 4.4).

Sheep Creek: (Shrub and Herbaceous Riparian Complex). The site scored 100% on the riparian assessment. The riparian area was comprised exclusively of sedges and grasses, while the valley upland was comprised of grasses and only used for sheep grazing. Impacts from sheep grazing appeared to be very minimal. The only woody shrubs noted were a few circular, ground-hugging junipers (not identified).

Battle Creek: (Recent Riparian Complex, Western Snowberry). Site has undergone deep incisement in recent times, as evidenced by an old legacy gallery of Great Plains cottonwoods on an abandoned floodplain at least 7 meters above the current channel. In the current riparian area there were mainly grasses, with a few willows intermixed. Recent cottonwood recruitment was not evident. Western snowberry (*Symphoricarpos occidentalis*) was present, as were disturbance-indicating species such as quack grass (*Agropyron repens*), baltic rush (*Juncus balticus*), and foxtail barley (*Hordeum jubatum*). Grazing in the current riparian area appeared to be light.

Middle Fork Poplar River: (Shrub and Herbaceous Riparian Complex). The site was located in a valley dominated by dry-land wheat production. The adjacent riparian zone was mainly grasses and sedges, with a few locations having woody species as well (snowberry and woods rose). Just beyond the bankfull channel snowberry and woods rose were common, but growth was reduced below potential due to cattle grazing. Overall, grazing appeared moderate but trampling of banks was light to moderate and not extensive.

Frenchman Creek: (Recent Riparian Complex, Western Snowberry). The stream channel was incised. The riparian zone was mainly grasses and American licorice, with some snowberry also present. Silver buffaloberry (*Shepherdia argentea*) was commonly found upgradient from the stream bank on the old floodplain. Grazing appeared to be at least moderate, with a few signs of overgrazing and trampling in the creek itself. In an abandoned oxbow of the creek a short distance downstream, an old cottonwood gallery was noted; however, no cottonwood recruitment was occurring in the current channel.

Butte Creek: (Shrub and Herbaceous Riparian Complex). Flaxville gravels were clearly evident in the stream and along eroding outside bends. No trees were visible in this drainage, but there was extensive woods rose and snowberry growth in some parts of the riparian, while other areas where mainly grasses. Sedges and rushes were common along the stream bank. Cattle use appeared moderate, and some bank trampling was present.

Wolf Creek nr Medicine Lake: (Shrub and Herbaceous Riparian Complex). The riparian area was dominated by snowberry, woods rose and grasses, with sedges found along the edge of the creek. Some parts of the stream reach were trampled and overwidened by cattle, while others were untrampled and had good vegetative cover. An instream cattle watering hole was located immediately upstream of the study reach.

Larb Creek: (Recent Riparian Complex). This stream has undergone major incisement. The riparian zone along the stream was mainly sedges, rushes and grasses, with American licorice and snowberry found a bit higher up. Grazing appeared to be at least moderate, with trampling in the stream and along the banks common. Within the study reach was an old meander of the stream that had been cutoff and was perched 3 meters above the current channel elevation. This meander had the only cottonwood tree (very old) visible in the area. This suggests that the stream may have at one time had a cottonwood gallery, as has been seen at other sites (e.g., Battle Creek).

Beaver Creek: (Recent Riparian Complex, Box-elder/Common Chokecherry). Stream has incised in the past. Older boxelder trees were found along the edge of the old floodplain, while younger boxelder, snowberry, and grasses were common in the riparian zone. Common chokecherry was not noted. Grasses provided some stream shading along banks. Grazing appeared to be light to moderate.

Shotgun Creek: (Oxbow/cattail Marsh): The only site of this riparian type in the study. The riparian zone was composed of common cattails (*Typha latifolia*) and sedges. Emergent macrophytes (e.g., duck potato, *Sagittaria latifolia*) were also very common. The stream was flowing during most of the summer, but the stream also had many wetland-like characteristics. Some cattle trampling was noted, but the overall riparian condition was excellent. Stream may have been channelized at one time in order to accommodate the hay field nearby.

Smoke Creek: (Shrub and Herbaceous Riparian Complex). This stream had little or no channel incisement. Snowberry and woods rose were present in the riparian and upland areas, but appeared to be greatly reduced due to heavy cattle grazing pressure. Most of the riparian area was grasses, and virtually all of the outside meanders of the stream were collapsing. Binding root mass protection was very poor.

Willow Creek (S): (Recent Riparian Complex, Sandbar Willow). This stream had an extremely incised and unstable channel. The riparian was mainly composed of sandbar willow and grasses on unstable, collapsing banks of fine material. Some cattle trampling, as well as deer use, were noted in the channel. From a stream bank stability perspective, this site demonstrated the poorest condition in the study.

Big Sandy Creek: (Shrub and Herbaceous Riparian Complex, Foxtail Barley and Woods Rose). The stream was only slightly incised. The riparian zone was composed mainly of sedges and grasses. Foxtail barley (*Hordeum jubatum*) was common in the riparian area, a plant noted for its tolerance to saline or alkaline waters. The stream was over-widened by cattle trampling in spots, and heavy grazing had greatly reduced the woody plant species potential. Woods rose was not noted in the reach, but was observed just downstream in an area under different management.

Redwater River: (Recent Riparian Complex). Stream has incised over time. An old gallery of Great Plains cottonwoods was located on the old floodplain about 60 m from the current channel, and elevated about 2 m above it. The current riparian was composed of sedges, grasses, woods rose and silver buffaloberry. Grazing appeared to be light to moderate, with only minor bank trampling noted.

Little Muddy Creek: (Shrub and Herbaceous Riparian Complex). The riparian zone was comprised of sedges, grasses, snowberry, current, and bulrush (*Scirpus spp.*). The stream was overwidened in parts and cattle trampling was evident. Woody shrubs appeared to be reduced from potential, and grazing was at least moderate. The stream may have been diked in the past.

Example photographs of each of the five riparian habitat & community types observed (Hansen et al. 1995) are shown below in Fig.4.4.

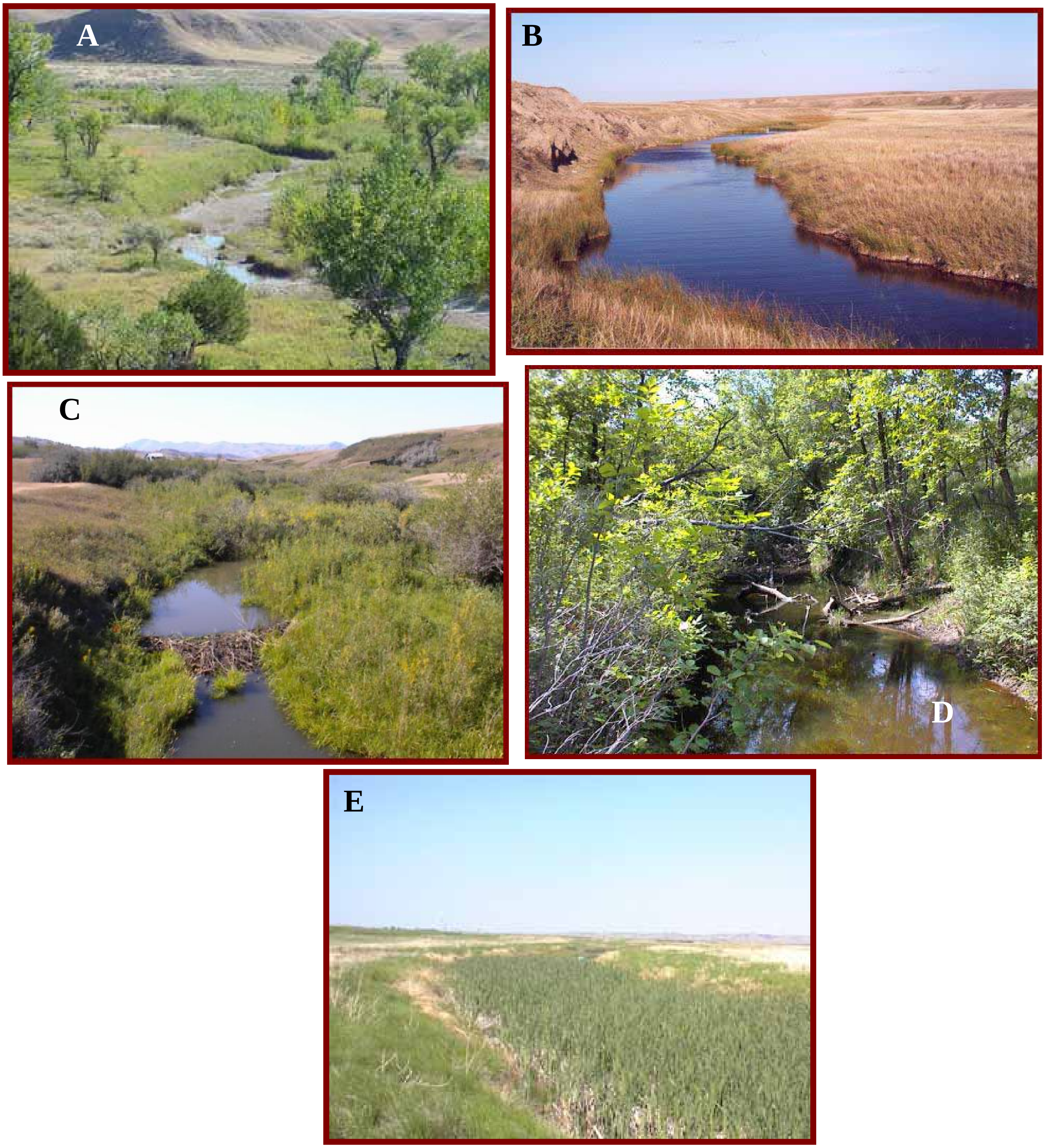


Figure 4.4. Examples of the general riparian habitat & community types found at the the sites. A. Recent Riparian Complex. B. Shrub and Herbaceous Riparian Complex. B. Intermittent Riparian Coulee. D. Woody Draw. E. Oxbow/cattail Marsh.

SECTION 5.0
PHYSICAL AND CHEMICAL WATER-QUALITY CHARACTERISTICS OF THE STREAM SITES

5.1 Physical Characteristics

5.1.1 Flow

Flow was measured whenever possible at each stream site during each visit. Nearly half of the streams ceased flowing at some point during the summer (Table 5.1), becoming a series of pools with no surface connection. The three rivers in the study (Middle Fork Poplar, West Fork Poplar, and Redwater rivers) and a number of streams maintained flow throughout the study. It is very probable that a number of the streams that were intermittent in 2001 or 2002 would run perennially during wetter climatic periods. Nevertheless, none of the study streams (perennial or intermittent) became completely dry during the summer in spite of the very hot, dry years of this drought period. A number of the stream sites were visited in early fall in order to make morphology measurements, and it was noted at that time that many were at their lowest observed water levels.

In terms of stream flow, the effect of the drought varied considerably over space and time. For example, Clear and Big Sandy creeks (in 2001) and the Redwater River (in 2002) had average summer flows an order-of-magnitude lower than their long-term summer averages (Table 5.1, far right columns). In contrast, average summer flow in the Middle Fork Poplar in 2002 (3.5 CFS) was similar to its long-term summer average (3.9 CFS), while in Rock Creek in 2002 average summer flow (2.0 CFS) was about half its long-term summer average (4.1 CFS; Table 5.1).

A number of the streams and rivers in this study originate in Canada. Study streams having their origins in Canada are Battle, Frenchman, and Rock creeks, and the Middle Fork and West Fork of the Poplar River. International agreements between Canada and the United States regulate the delivery of water to the United States. For Battle, Frenchman and Rock creeks these deliveries are governed by the Order of the International Joint Commission dated October 4, 1921 (Goos and Carswell 2002), while water deliveries for the Middle and West Forks of the Poplar River have been recommended in the 1976 "International Souris-Red Rivers Board, Poplar River Task Force"' main report (Poplar River Bilateral Monitoring Committee 2003). Fundamentally, both of these documents indicate that natural flows are to be equally divided between the two countries.

Battle Creek, Frenchman Creek, and a tributary of the West Fork Poplar River are dammed and have reservoirs in Canada. This complicates the water delivery agreements in cases where Canadian reservoirs regulate flows. For example, if during a given month Canadian use of water from Battle Creek is in excess of its share, then a delivery of equivalent volume will be delivered to the U.S. at the earliest opportunity. As a consequence, flows in these streams can vary considerably from their natural patterns. In Battle Creek in 2002, the U.S. received much less than its share throughout the spring but then received more than its share throughout the summer (Goos and Carswell 2002). Rock Creek and the Middle Fork Poplar River do not appear to have any flow-regulating structures on them on the Canadian side of the border.

5.1.2 Total Suspended Solids and Turbidity

Both total suspended solids (TSS) and turbidity varied over four orders of magnitude in the study sites (Fig. 5.1A&B). Although three of the four comparison sites had very low TSS and turbidity, one of the sites (Rock Creek) had the third highest mean TSS concentration and the fourth highest mean turbidity in the study. TSS concentrations in Rock Creek varied from 33.0 to 869.0 mg/L. Five streams stood out as having exceptionally high TSS and turbidity in the study; Rock, Willow (N), Frenchman, Larb, and Willow (S) creeks. Willow Creek (S) had especially high TSS values, with a maximum value of 2,390 mg/L. High TSS and turbidity values were probably related to a combination of the type of terrain and geology through which the streams flowed, stream morphology, and land use.

There was a significant ($P \leq 0.05$) relationship with a fairly good fit ($r^2 = 0.68$) between TSS concentrations and measured turbidity (Fig. 5.2A&B). Unlike EC-TDS relationships, there is frequently much more scatter in TSS-turbidity relationships and the correlation coefficients can be quite variable (Kunkle and Comer 1971; Costa 1977; Anderson and Potts 1987). Fig. 5.2A is shown on log scale to capture the full range of sample measurements, while Fig. 5.2B only includes samples where the TSS concentrations was between 0 and 100. The purpose of 5.2B is to show greater detail in the TSS-turbidity relationship at the lower end of the scale.

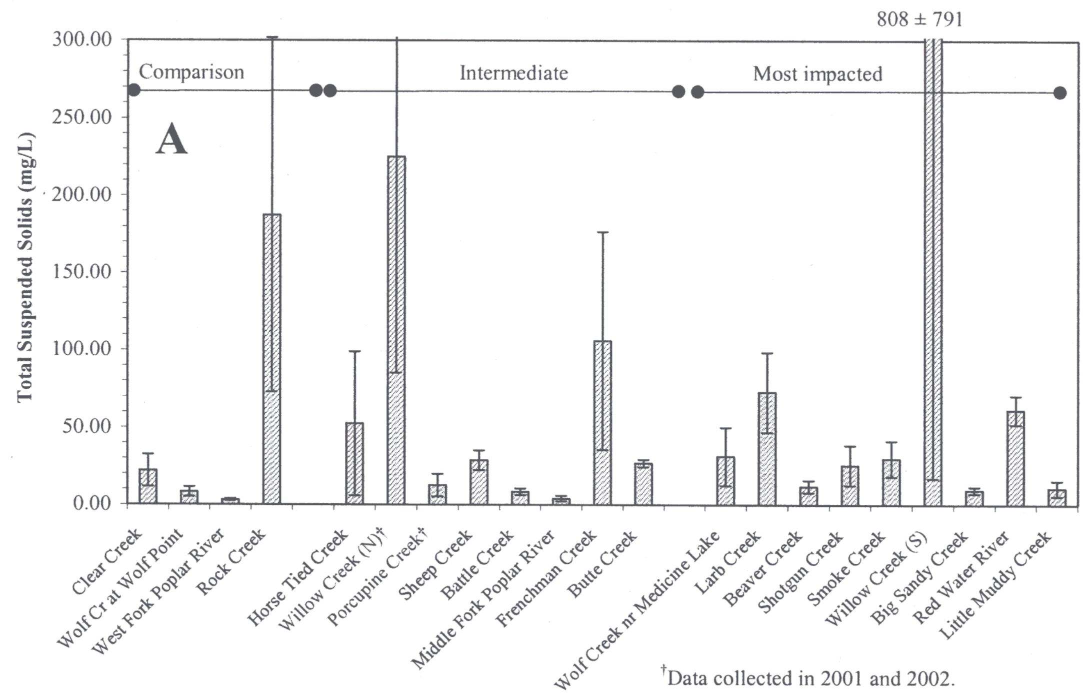
Table 5.1. Measured summertime flows in the study streams, with comparison flows for guaged streams. Flow status is based only on 2001-2002 observations.

			2001 Flow (CFS)*			2002 Flow (CFS)*			Flow at nearest USGS guage station			
Study Year	Stream Site	Condition Group	June	Aug	Sept	July	Aug	Flow status, 2001-2002	USGS Station No.	Location	Average summer flow during year sampled (CFS)†	Long-term average summer flow (CFS)‡
2001	Clear Cr	Reference	0.4	<<1	<<1	na	na	Intermittent	06142400	downstream	0.0	5.5
2002	Wolf Cr at Wolf Point	Min. impacted	na	na	na	no data	no flow	Intermittent	none			
2002	WF Poplar River	Min. impacted	na	na	na	2	1.6	Perennial	none			
01 & '02	Rock Cr	Min. impacted	1.6	no flow	no flow	0.7	0.5	Intermittent	06169500	upstream	2.0 ('02)	4.1
2002	Horse Tied Cr	Intermediate	na	na	na	<<1	<<1	Intermittent	none			
01 & '02	Willow Cr (N)	Intermediate	0.3	no flow	no flow	no flow	<<1	Intermittent	none			
01 & '02	Porcupine Cr	Intermediate	2.6	<<1	<<1	no flow	no flow	Intermittent	none			
2002	Sheep Cr	Intermediate	na	na	na	no flow	no flow	Intermittent	none			
2001	Battle Cr	Intermediate	8.8	no flow	<<1	na	na	Intermittent	06151500	downstream	0.7	22.6
2002	MF Poplar River	Intermediate	na	na	na	3.2	4.6	Perennial	06178000	upstream	3.5	3.9
2001	Frenchman Cr	Intermediate	31.7	<<1	<<1	na	na	Intermittent	06164000	upstream	5.7	26.0
2002	Butte Cr	Intermediate	---	---	---	0.2	0.7	Perennial	none			
2002	Wolf Cr nr Medicine Lake	Most Impacted	na	na	na	0.1	0.03	Perennial	none			
2001	Larb Cr	Most Impacted	no data	no flow	no flow	na	na	Intermittent	none			
2001	Beaver Cr	Most Impacted	2.7	no flow	no flow	na	na	Intermittent	06166000	downstream	0.1	38.3
2002	Shotgun Cr	Most Impacted	na	na	na	<<1	<<1	Perennial	none			
2002	Smoke Cr	Most Impacted	na	na	na	1.9	1.3	Perennial	none			
2001	Willow Cr (S)	Most Impacted	no data	no flow	no flow	na	na	Intermittent	none			
2001	Big Sandy Cr	Most Impacted	0.1	<<1	<<1	na	na	Perennial	06139500	downstream	0.9	9.6
2002	Red Water River	Most Impacted	na	na	na	2.7	1.6	Perennial	06177500	upstream	0.1	5.2
2002	Little Muddy Cr	Most Impacted	na	na	na	<<1	<<1	Perennial	none			

* <<1 indicates there was flow, but only an unmeasureable trickle.

† Calculated as the average of the mean monthly flow for July, August, and September of the year the site was sampled, as published in the USGS 2001 or 2002 Water Year report.

‡ Calculated as the average of the long-term mean flows pulished for the months of July, August, and September in the USGS 2002 Water Year report (Berkas et al. 2003). Length of record varied from station to station; but data is current through 2002.



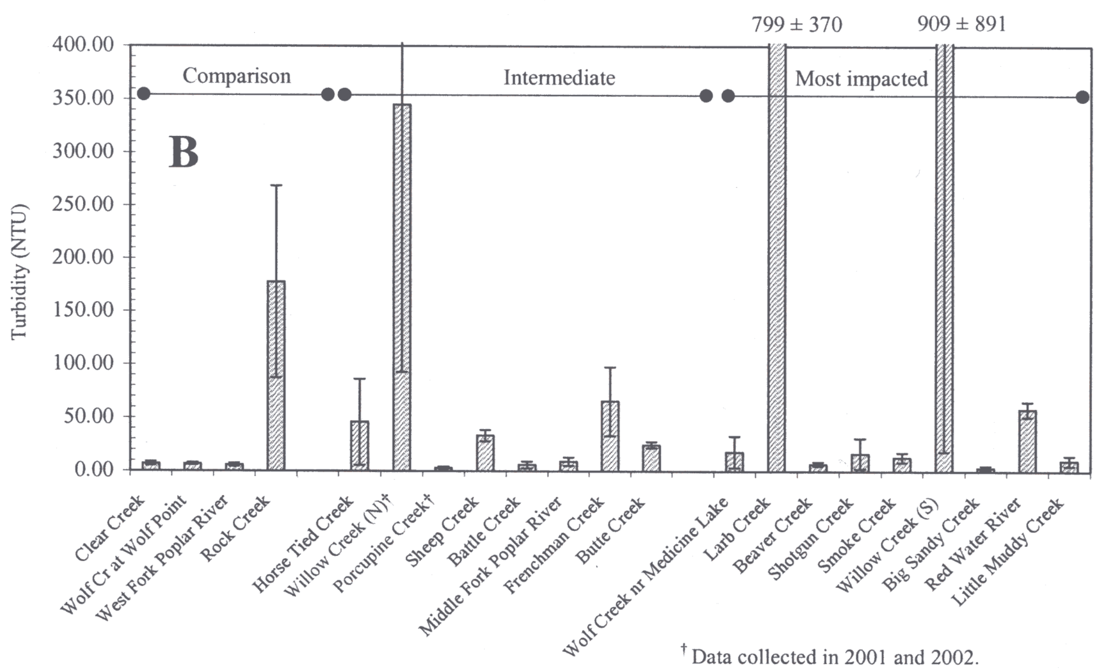


Figure 5.1. A. Mean spring and summer total suspended sediment (TSS) concentrations in the in the comparison, intermediate, and most impacted sites. B. Mean spring and summer turbidity (NTU) in the comparison, intermediate, and most impacted sites. Error bars are one standard error of the mean.

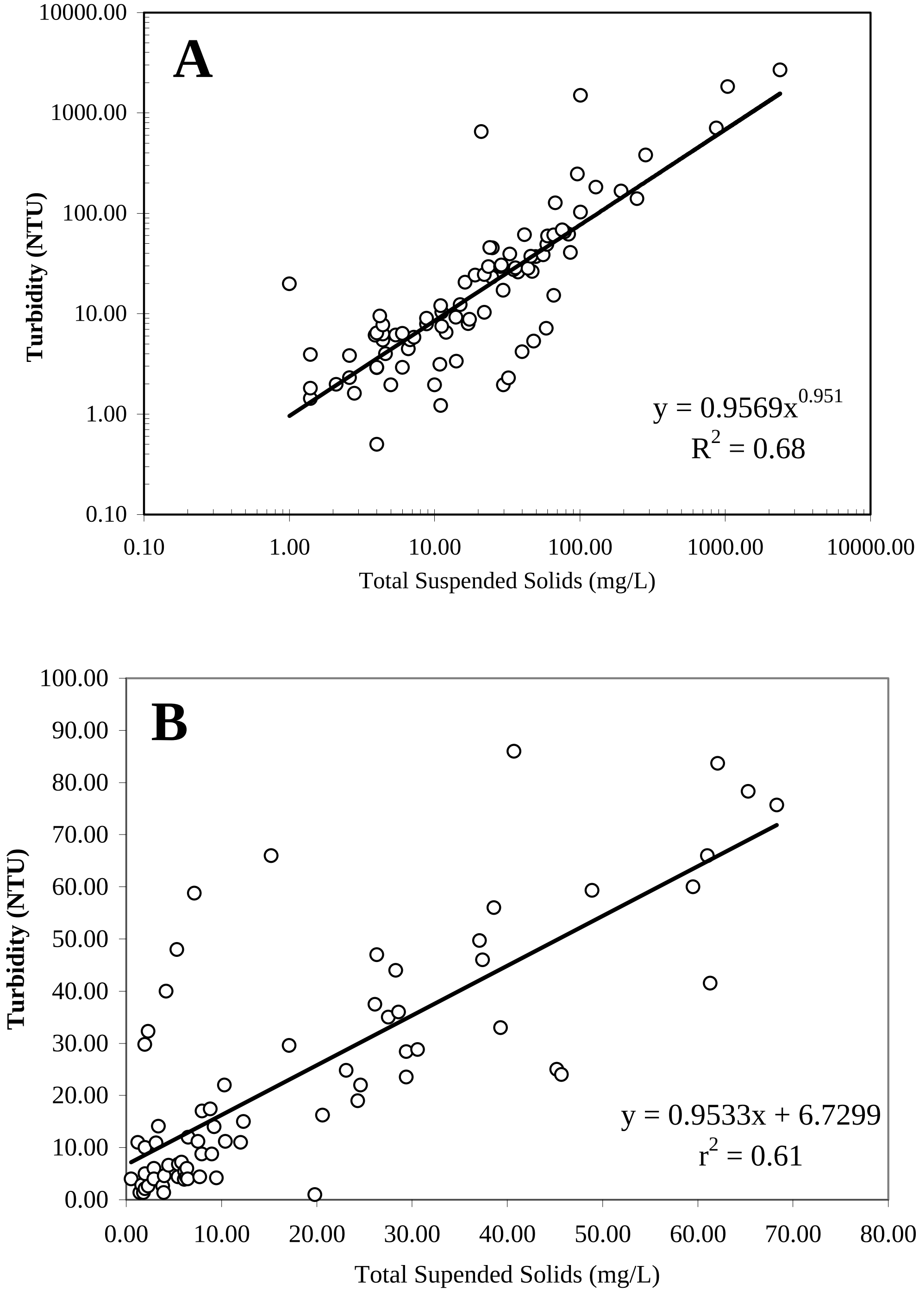


Figure 5.2. Relationship between total suspended solids (TSS) and turbidity in the study streams. Data were collected in spring and summer. A. Relationship shown on log scale, includes all measured values. B. TSS-turbidity relationship at the lower end of the scale (only data points are shown where the TSS concentration were less then 100 mg/L).

5.2 Chemical Characteristics

Waters in this study showed a wide range of chemical characteristics, reflective of the varying geology of the region. The geology is made up of upper Cretaceous marine shales, sandstones, and siltstones overlain by glacial till (Alt and Hyndman 1986). Table 5.2 shows the detail of each individual stream site's basic water chemistry. The range of values is shown in parenthesis, and means (rather than medians) were calculated as it was intended that any unusually high and low values be reflected in the descriptive statistic. Regional averages have not been calculated, as other reports that have summarized this type of data on a regional basis (e.g., Bahls et al. 1992) have been found in the past to be too general for specific watershed projects. Therefore, each stream site is shown individually in hopes that it will be more useful to the reader. It should be noted that these data were collected during two years of an ongoing dry period that began in 1998 (NWS 2004). As a result, concentrations are probably higher than what would be found during wetter periods.

The means and ranges shown were only calculated for samples where the error in the cation/anion balance was within 20%. Twenty percent error was considered a reasonable error given the high solute concentrations found in these streams (Halm, personal communication) and given that the bicarbonate concentration is estimated. All of the streams had either sodium-bicarbonate or sodium-sulfate as the dominant cation/anion pair, except for one stream (Clear Creek) that was a calcium-bicarbonate water. All of the streams were hard to very hard (high concentrations of calcium and magnesium), except for Larb Creek, which was moderately hard. The pH of all the streams was consistently alkaline and usually 8.0 or higher, as expected given the high total alkalinity of this region's waters.

Fig. 5.3A&B shows spring and summer means for electrical conductivity (EC) in the study sites. In Fig. 5.3B the streams are arranged by their sub-major basins (Seaber et al. 1987). In general, the comparison sites all had fairly low EC values (usually 1500 µS/cm) for the region as a whole. The highest EC value measured in a comparison site was 2,010 µS/cm in Rock Creek in September, 2001. 1,500 µS/cm is considered a threshold beyond which the water may have adverse effects on many irrigated crops (Ayers and Westcott 1985). Big Sandy Creek had some of the highest EC values in the study (up to 6,070 µS/cm), which are elevated due to saline seeps resulting from the crop/fallow cropping practices used in the basin (MT DEQ 2002b). There were no clear patterns in EC as one progressed from west to east across the basin (Fig. 5.3B), although some smaller basins showed distinct characteristics. For example, both the Redwater River and Sheep Creek are located in the same 4th field basin (USGS HUC No. 10060002) south of the Missouri River and have similarly high EC values. The West Fork Poplar River, Middle Fork Poplar River, and Butte Creek are all major tributaries that form the Poplar River, and all of them have similarly low EC values.

Table 5.2. Means and ranges of major ions and other water quality parameters in the study stream sites. All data were collected in the spring and summer of 2001and/or 2002.

			Anions			Cations								
Stream Site	Year	Condition Rating	Sulfate [SO_4^{2-}] (mg/L)	Chloride [Cl^-] (mg/L)	Bicarbonate [HCO_3^-] (mg/L)*	Calcium [Ca^{2+}] (mg/L)	Magnesium [Mg^{2+}] (mg/L)	Sodium [Na^+] (mg/L)	Potassium [K^+] (mg/L)	Total Alkalinity (mg/L as $CaCO_3$)	pH	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)‡	Dominant cation/anion
Clear Creek	2001	Reference	91.6 (28.9-141.0)	5.9 (5.5-6.2)	446 (349-508)	83.2 (60.2-96.0)	35.8 (23.5-43.0)	47.0 (36.2-54.3)	11.8 (10.3-13.4)	369.3 (288.0-420.0)	7.98 (7.86-8.16)	355.3 (247.1-416.8)	497 (337-592)	Calcium Bicarbonate
Wolf Cr at Wolf Point	2002	Min. impacted	195.0 (156.0-235.0)	14.5 (12.6-16.6)	568 (560-581)	24.0 (12.3-35.9)	32.4 (30.4-34.2)	220.0 (200.0-240.0)	6.7 (6.0-7.5)	494.5 (460.0-522.0)	8.70 (7.86-9.16)	193.2 (158.9-230.5)	789 (764-817)	Sodium Bicarbonate
WF Poplar River	2002	Min. impacted	137.3 (110.0-177.0)	6.1 (5.7-6.7)	610 (555-678)	18.2 (13.2-23.6)	18.6 (17.7-19.6)	220.5 (202.0-238.0)	8.2 (6.5-10.8)	510.5 (480.0-564.0)	8.57 (8.16-8.80)	121.8 (111.6-132.6)	715 (646-802)	Sodium Bicarbonate
Rock Creek†	01 & '02	Min. impacted	358.8 (84.8-818.0)	8.5 (2.0-13.9)	272 (156-391)	35.2 (22.5-42.7)	20.6 (11.7-32.8)	187.2 (61.2-340.0)	11.5 (7.5-15.6)	280.0 (128.0-580.0)	8.3 (7.22-8.64)	176.4 (104.4-240.7)	789.8 (267-1390)	Sodium Sulfate
Horse Tied Creek	2002	Intermediate	142.1 (81.8-234.0)	10.7 (2.5-25.9)	474 (444-511)	60.7 (42.2-73.9)	25.4 (21.9-27.2)	121.0 (109.0-145.0)	20.1 (9.7-32.6)	394.5 (378.0-424.0)	8.10 (7.61-8.61)	255.9 (195.6-296.5)	617 (519-771)	Sodium Bicarbonate
Willow Creek N†	01 & '02	Intermediate	591.8 (234.0-962.0)	24.3 (5.4-98.5)	125 (58-200)	53.0 (26.7-73.4)	25.7 (12.2-36.3)	205.7 (84.1-302.0)	11.9 (6.9-15.9)	126.0 (48.0-228.0)	8.09 (6.90-8.69)	238.2 (116.9-332.8)	988 (409-1463)	Sodium Sulfate
Porcupine Creek†	01 & '02	Intermediate	447.9 (396.0-501.0)	24.0 (20.8-27.4)	369 (358-398)	66.2 (53.6-79.8)	32.3 (30.0-34.6)	204.3 (194.0-218.0)	8.5 (7.7-9.4)	307.4 (288.0-332.0)	8.05 (7.44-8.59)	298.5 (262.6-341.7)	967.7 (896-1058)	Sodium Sulfate
Sheep Creek	2002	Intermediate	1562.5 (1380.0-1700.0)	15.0 (13.1-15.7)	899 (682-1177)	22.4 (15.9-28.4)	31.0 (27.4-34.4)	927.8 (736.0-1200.0)	11.1 (9.6-13.1)	779.5 (578.0-1100.0)	8.92 (8.55-9.17)	183.5 (169.1-212.6)	3037 (2689-3634)	Sodium Sulfate
Battle Creek	2001	Intermediate	622.0 (280.0-964.0)	27.0 (9.6-44.4)	203 (145-261)	61.3 (36.6-85.9)	47.3 (19.8-74.7)	199.5 (110.0-289.0)	13.7 (12.0-15.3)	168.0 (120.0-216.0)	7.99 (7.98-7.99)	347.5 (172.9-522.1)	1071 (540-1603)	Sodium Sulfate
MF Poplar River	2002	Intermediate	257.5 (220.0-297.0)	8.5 (7.3-8.7)	522 (476-599)	33.4 (16.1-54.9)	50.4 (42.2-59.7)	204.3 (184-228)	12.3 (10.5-15.2)	484.0 (480.0-496.0)	8.89 (8.00-9.38)	290.7 (219.2-356.5)	857 (794-949)	Sodium Bicarbonate

*Calculated from total alkalinity and pH (APHA 2000).

† Means and ranges were calcualted from data collected in both 2001 and 2002.

‡ Calculated as the sum of cations and anions as per APHA (1998), excluding SiO_3, NO_3, and F.

Table 5.2, cont. Means and ranges of major ions and other water quality parameters in the study stream sites. All data were collected in the spring and summer of 2001 and/or 2002.

			Anions			Cations								
Stream Site	Year	Condition Rating	Sulfate [SO_4^{2-}] (mg/L)	Chloride [Cl^-] (mg/L)	Bicarbonate [HCO_3^-] (mg/L)*	Calcium [Ca^{2+}] (mg/L)	Magnesium [Mg^{2+}] (mg/L)	Sodium [Na^+] (mg/L)	Potassium [K^+] (mg/L)	Total Alkalinity (mg/L as $CaCO_3$)	pH	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)‡	Dominant cation/anion
Frenchman Creek	2001	Intermediate	1481.3 (374.0-2110.0)	41.8 (8.3-64.6)	479 (335-576)	108.4 (65.1-131.0)	77.6 (50.2-98.4)	565.7 (140.0-825.0)	22.9 (15.8-26.8)	400.0 (280.0-480.0)	8.29 (8.24-8.32)	590.3 (369.3-732.3)	2538 (821-3504)	Sodium Sulfate
Butte Creek	2002	Intermediate	310.8 (125.0-545.0)	11.2 (8.9-13.9)	514 (438.0-631.0)	39.8 (31.3-48.7)	43.2 (34.7-51.9)	215.5 (102.0-347.0)	7.7 (6.7-10.0)	445.0 (376.0-522.0)	8.53 (7.92-8.97)	277.1 (221.1-335.3)	895 (551-1309)	Sodium Bicarbonate
Wolf Cr nr Medicine Lake	2002	Most impacted	1420.0 (1430.0-1630.0)	28.6 (19.1-43.7)	600 (519-686)	75.6 (37.3-119.0)	170.8 (110.0-214.0)	476.0 (400.0-527.0)	22.6 (15.9-37.2)	508.0 (468.0-580.0)	8.68 (8.51-9.02)	891.8 (584.8-1113.7)	2498 (2209-2748)	Sodium Sulfate
Larb Creek	2001	Most impacted	164.0 (137.0-216.0)	2.7 (2.4-3.0)	86 (73-107)	24.2 (19.7-27.6)	10.3 (7.3-13.7)	67.9 (58.0-87.1)	9.9 (8.4-12.0)	70.7 (60.0-88.0)	7.67 (7.41-7.85)	102.8 (79.3-119.8)	321 (274-405)	Sodium Sulfate
Beaver Creek	2001	Most impacted	1656.7 (1500.0-1870.0)	80.2 (72.7-88.5)	134 (110-156)	91.9 (67.2-141.0)	71.2 (60.8-84.1)	589.7 (534.0-693.0)	19.3 (17.2-20.4)	122.7 (116.0-132.0)	8.96 (8.51-9.45)	522.6 (418.2-698.4)	2582 (2332-2878)	Sodium Sulfate
Shotgun Creek	2002	Most impacted	1192.5 (1050.0-1310.0)	13.0 (9.4-16.8)	945 (913.0-969.0)	48.2 (28.5-74.1)	117.5 (109.0-136.0)	625.0 (592.0-688.0)	17.2 (13.5-23.9)	794.0 (756.0-836.0)	8.50 (8.00-8.75)	604.3 (520.1-745.1)	2490 (2333-2697)	Sodium Sulfate
Smoke Creek	2002	Most impacted	557.0 (509.0-636.0)	23.9 (16.1-34.0)	516 (488-537)	67.5 (46.9-105.0)	119.1 (89.0-166.0)	202.6 (89.4-297.0)	13.6 (10.9-18.4)	571.5 (404.0-980.0)	8.39 (7.99-8.62)	658.9 (485.3-945.8)	1327 (1032-1759)	Sodium Sulfate
Willow Creek S	2001	Most impacted	547.0 (231.0-994.0)	77.2 (3.6-174.0)	147 (34-226)	75.0 (41.8-120.0)	29.7 (16.5-49.2)	206.1 (80.2-380.0)	12.2 (10.0-15.3)	122.7 (28.0-188.0)	7.91 (7.03-8.46)	309.7 (172.3-502.2)	1021 (401-1845)	Sodium Sulfate
Big Sandy Creek	2001	Most impacted	3260	177	1633	106	260	1010	21.6	1680	9.43	1335.4	5843	Sodium Sulfate
Redwater River	2002	Most impacted	1902.5 (1630.0-2090.0)	23.3 (19.2-32.8)	752 (692-818)	50.6 (38.2-68.6)	121.3 (111.0-132.0)	864.5 (727.0-957.0)	21.0 (16.9-31.8)	649.0 (612.0-680.0)	8.65 (8.16-8.92)	625.7 (564.8-668.4)	3373 (2990-3650)	Sodium Sulfate
Little Muddy Creek	2002	Most impacted	799.0 (601.0-998.0)	15.3 (10.2-26.5)	817 (745-858)	35.9 (18.0-51.7)	54.7 (44.6-72.1)	501.0 (341.0-647.0)	17.9 (12.9-31.6)	671.5 (584.0-744.0)	8.59 (8.20-8.87)	314.9 271.8-370.8)	1827 (1447-2166)	Sodium Bicarbonate

*Calculated from total alkalinity and pH (APHA 1998).

‡ Calculated as the sum of cations and anions as per APHA (1998), excluding SiO_3, NO_3, and F.

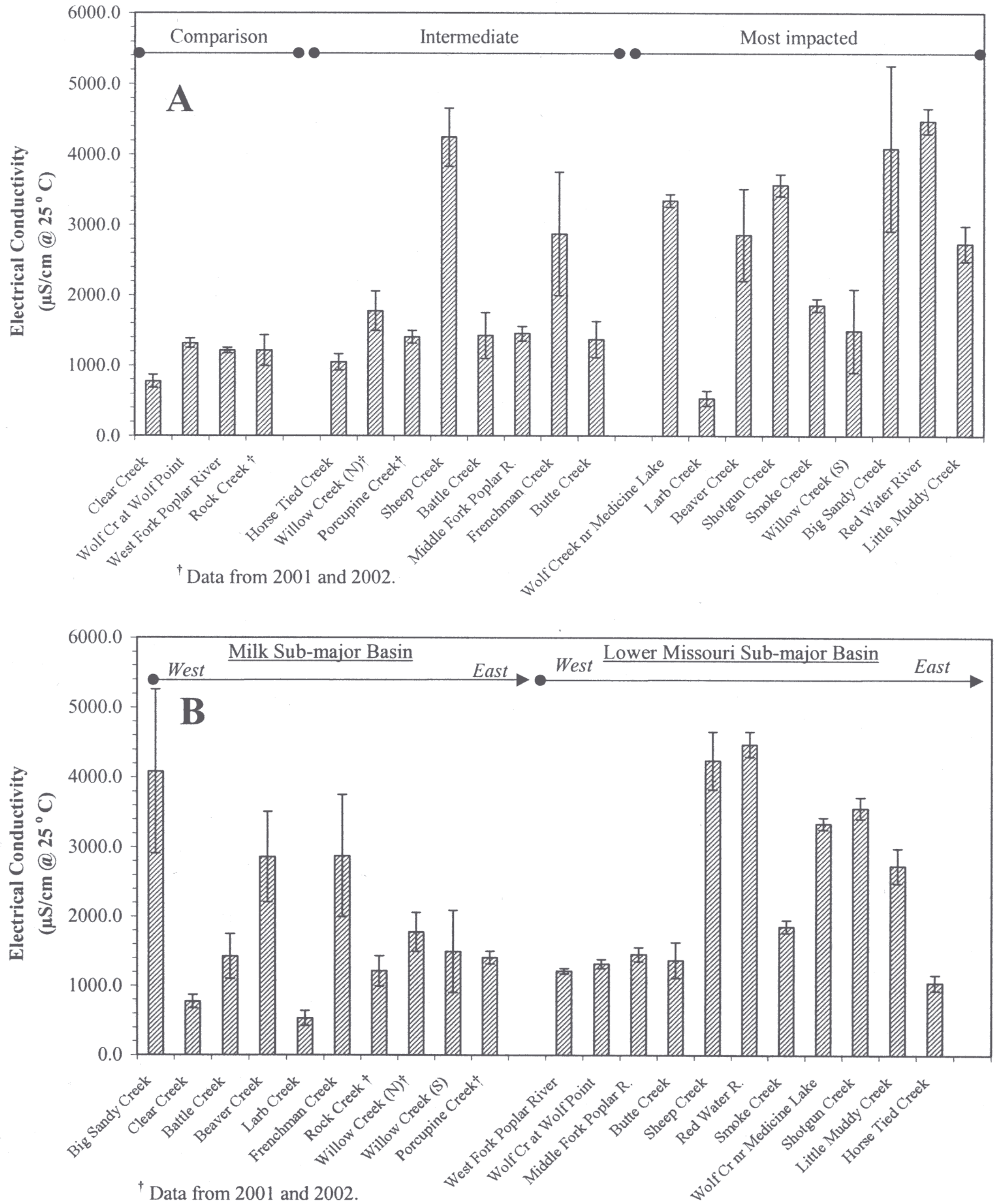


Figure 5.3. Electical conductivity in the study stream sites. Error bars are one standard error of the mean. A. Stream sites arranged by their condition ranking. B. Streams sites arranged from west to east within their Sub-major basins.

The sodium adsorption ratio (SAR) is an important index for measuring the potential hazard to irrigated soils from excess sodium in the water supply. If there is too much sodium relative to calcium and magnesium, the sodium will dominate the clay surfaces in the soil and cause swelling, causing the soil to be less permeable and diminishing salt leaching (Hansen et al. 1999). The potential impact of sodium (measured as SAR) varies as a function of the water's EC and needs to be considered in relation to the EC; the reader is referred to Hansen et al. (1999) and Ayers and Westcott (1985) for more information. SAR showed, in general, similar patterns to those of EC (Fig.5.4A&B). One notable value was the mean SAR of 30 in Sheep Creek; this is an unusually high value and even given the stream's high EC (mean: 4,242 μS/cm) use of this water for irrigation would cause at least moderate reductions in soil infiltration. Appropriately, Sheep Creek basin is used mainly for grazing sheep. EC and SAR are significantly correlated to one another but the relationship is only moderately strong (Fig. 5.5).

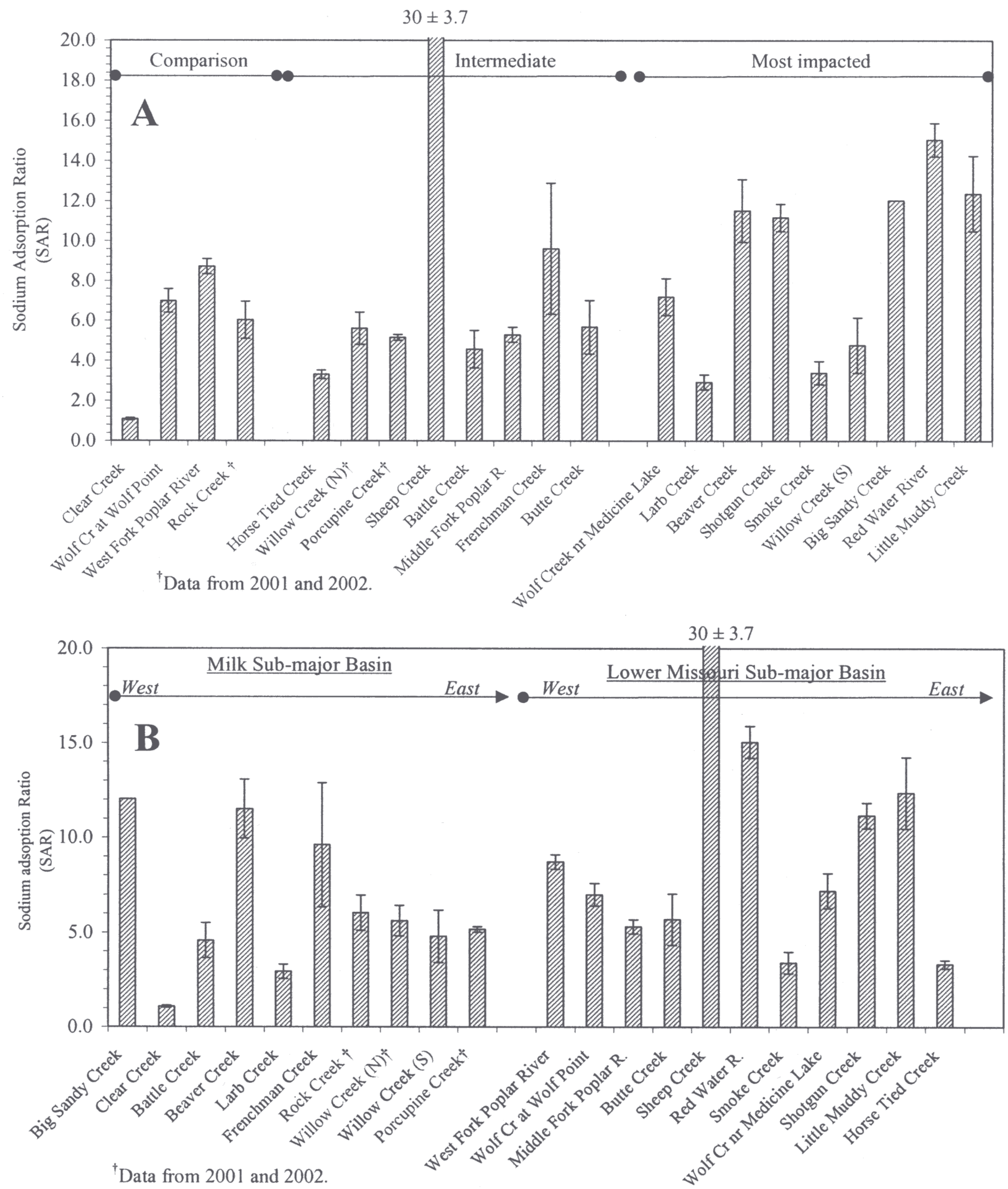


Figure 5.4. Sodium adsorption ratio in the study stream sites. Error bars are one standard error of the mean. A. Stream sites arranged by their condition ranking. B. Streams sites arranged from west to east within their Sub-major basins.

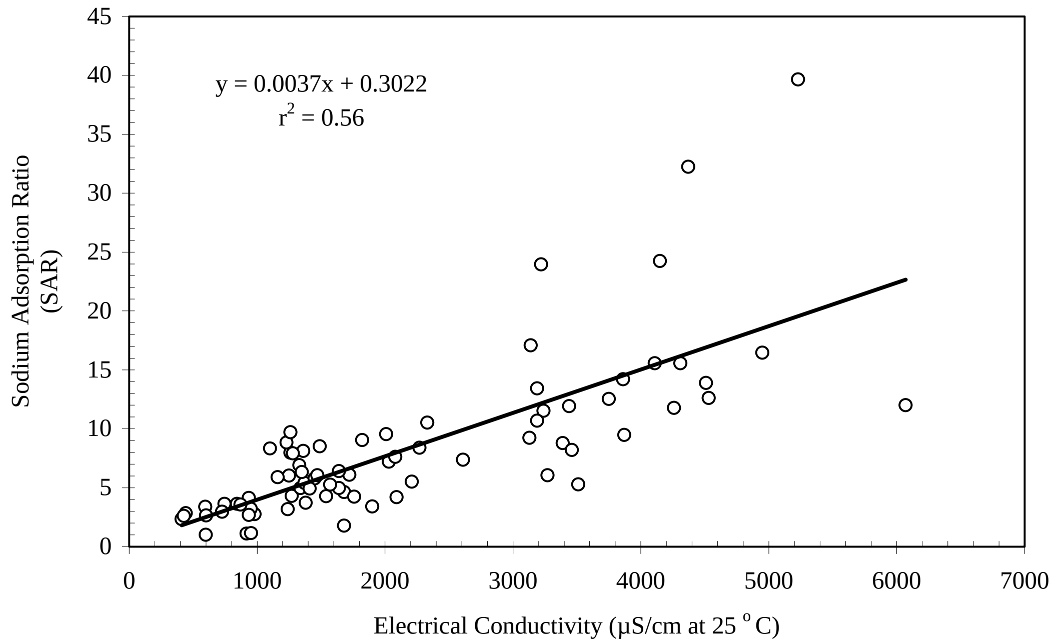


Figure 5.5. Relationship between EC and SAR in the study streams.

The relationship between EC and total dissolved solids (TDS) was also examined in the study streams. All samples were collected in spring through summer. Not surprisingly, there was a near-perfect fit between the EC and TDS (Fig. 5.6). EC measurements are less expensive then TDS measurements and can be carried out in the field. The relationship in Fig. 5.6 can be used as a general relationship to convert EC measurements to TDS for wadeable streams of the Northwestern Glaciated Plains eceregion.

The seasonal change in the streams' EC (from June through September) was also examined to see if any patterns were manifested. In 2001 it appeared that there was a consistent, steady increase in streams' EC as flows ceased and the remaining water became ponded. This pattern did not hold in 2002 however, even at sites that had in 2001 shown clear EC increases as the summer progressed (e.g., Porcupine Creek). In fact, in 2002 the EC values in some streams were slightly lower near the end of the summer then they were at midsummer, long after flows had already ceased. Apparently, other factors (groundwater influences?) had more influence on EC values then evaporative increases in salt concentration.

Selected metals and selenium were measured to determine if they might be significant stressors on aquatic life or other beneficial uses. Zinc and aluminum were only measured in 2001 and only in the Milk sub-major basin streams. Most zinc samples (82%) were below the detection limit of 1.0 µg/L, and those few that were above the detection limit did not exceed any of Montana's surface water quality standards (MT DEQ 2002a). Dissolved aluminum levels were also low, and although one sample (Larb Creek, September 2001) did exceed Montana's chronic aquatic life standard by 170%, it did not exceed the acute criteria (DEQ 2002a). Aluminum levels may be naturally elevated in Larb Creek, as there is no mining activity in the basin and the only use of the basin upstream of the study site was grazing.

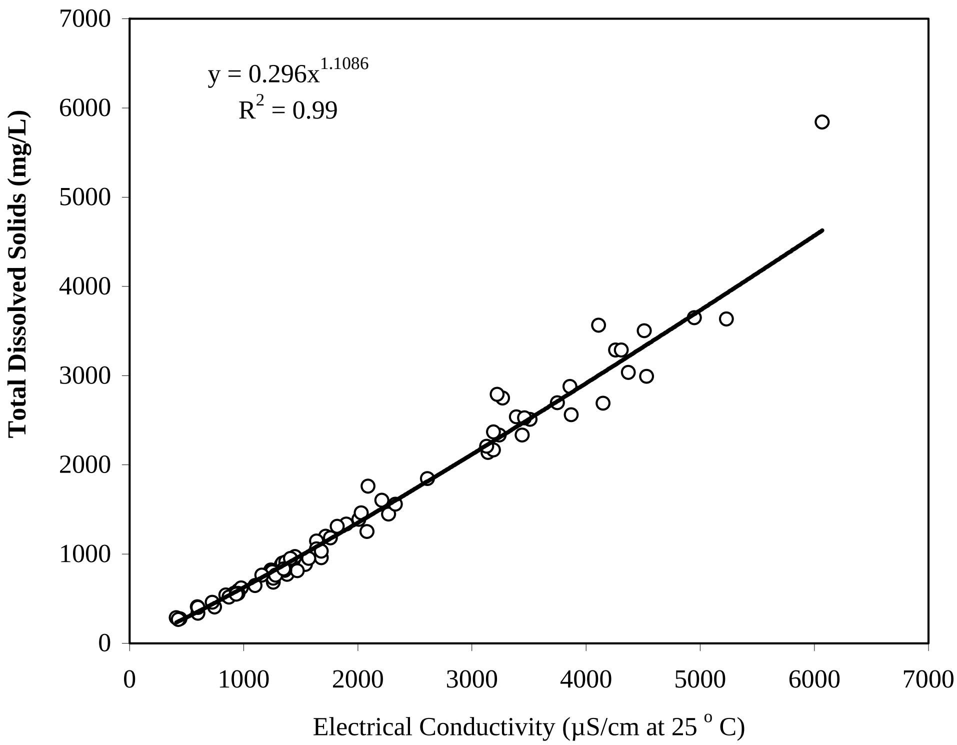


Figure 5.6. Relationship between TDS and EC in the study streams. Equation and R^2 shown are for the best fit relationship, which was a power regression trendline.

Selenium was measured in both years of the study, in study sites of both the Milk and the Lower Missouri sub-major basins. Out of 78 samples only one was above the detection limit of 2 µg/L. The single sample that was above the detection limit (3 µg/L; Wolf Cr nr Medicine Lake, July 2002) did not exceed any of Montana's water quality standards (DEQ 2002a).

SECTION 6.0 DEVELOPMENT OF USEFUL STREAM GROUPINGS FOR THE HI-LINE REGION

6.1 Cluster Analysis of Stream Sites Based on their Characteristics

Up to this point, the stream sites have been presented and discussed mainly by their position on the condition rankings developed back in Section 3.0. As the streams' physical and chemical data have been presented, it should have become apparent that in spite of the fact that all the streams are within a similar geographic region there is a tremendous diversity of intrinsic characteristics (e.g., riparian type, entrenchment ratio, dominant cation and anion, etc.). One of this project's goals was to locate and describe reference and minimally impacted sites, sites whose characteristics could then be used for comparative purposes. However, a comparison site is only useful if it has sufficient similarity to the stream against which it is being compared. Since there are demonstrable differences between the stream sites in the present study, it is appropriate to try to group streams with similar characteristics together.

Classification or grouping schemes are neither right nor wrong; they are either useful or not useful. From the MT DEQ's point of view, a useful grouping scheme would be one that clusters streams together that have similar ecological structure and function, and which could be expected to respond to stressors in a similar manner. Four guiding principles were used when selecting the characteristics used to create the stream groups:

1. The ultimate groups should make intuitive sense.

2. Given a choice, it is better to lump streams together then to split them apart. (An approach that ultimately assigns each stream to its own unique group is not useful).

3. The groups should be developed using stream characteristics that have been shown to be important influences on aquatic life, but which are at the same time either natural or essentially permanent stream features.

4. The approach taken should not create unique stream groups due to a degraded characteristic (e.g., elevated nutrients, high salinity), if there is a reasonable potential that the degraded characteristic can be improved.

There was an array of stream characteristics to choose from. These included flow, water chemistry (e.g. TDS, dominant cation, dominant anion, nutrients), stream morphology (e.g. entrenchment, width/depth ratio), and measures of the riparian habitat. Note that this list only contains physical, chemical, and riparian habitat attributes of the streams, and does not include any of the aquatic biological qualities. It was assumed that the aquatic biological community develops largely in response to the physical, chemical, and riparian habitat framework provided by the stream, and therefore any grouping scheme should be based on those characteristics.

Although riparian habitat is a biological attribute, it is included among the physical and chemical stream characteristics because the riparian corridor has a major influence on several important in-stream characteristics such as light availability, water temperature, streambank stability, and water storage (Sweeney 1993). A number of quantifiable measures (metrics) of the riparian area were collected in this study using EMAP methods. These riparian metrics were reviewed and compared to the general riparian descriptions detailed in Section 4.0 in an effort to select a few key metrics that could distinguish and quantify the overall riparian types. Two EMAP metrics, 'mean % canopy density mid-stream' and 'riparian woody cover' (XCDENMID and XCMGW, respectively; Kaufmann et al. 1999), were ultimately selected. XCDENMID and XCMGW effectively captured distinct differences that were observed in the density and structure of the riparian types, clearly distinguishing in particular two riparian types (those of Horse Tied and Shotgun creeks) which were each unique in this study. The XCDENMID metric is the riparian canopy shading measured with a spherical densitometer (Lemmon 1957) at the midchannel water surface, and is therefore a direct measure of the shading quality of the riparian area[1]. The XCMGW metric measures the presence and relative density of woody vegetation, which was shown in sub-Section 4.3 to be quite variable among stream sites and therefore a distinguishing riparian attribute.

One of the most significant morphological features of a stream is its entrenchment, which is the vertical containment of a stream and the degree that it is downcut into the valley floor (Kellerhals et al. 1972). Not only is stream entrenchment important, it may also be a permanent or at least long-term feature of the Hi-line region's streams. This is because alluviation (the process by which incised gullies refill with sediment) is slow and may take hundreds of years, whereas erosion and downcutting is fast and can occur during periods of less then 50 years (Leopold 1994).

Salinity can influences aquatic life communities (Klarich and Regele 1980; Moreno et al. 2001), but may not be a particularly good choice as a grouping characteristic since salinity degradation (i.e. excess salinity) is a quality that has potential for improvement, as was shown to be the case for Big Sandy Creek (MT DEQ 2002b).

Cluster analysis (a multivariate statistical method; Everitt et al. 2001) was used to develop various stream groups using the characteristics listed above. Various grouping schemes were generated using combinations of the characteristics, and each scheme was considered in light of the grouping principles. As suspected, salinity and dominant cation/anion were not found to be useful inclusions in the grouping schemes as they both produced a small "high salinity" group that included Big Sandy Creek, which has already

[1] The mean % canopy density mid-stream (XCDENMID) metric was useful for grouping streams in this study because the focus was on lower order wadable streams, none of which were particularly wide (maximum wetted width 11 m). This metric may not be as useful in wider, higher order streams, as the midchannel locale could be far enough from the bank as to be unable to capture even taller trees. In such cases the bank canopy density metric (XCDENBK) may be more applicable.

been shown to have human–caused elevated salinity (and therefore this approach violated principle No. 4). In the end, the most intuitively correct and apparently useful grouping scheme was developed from three variables derived from two major stream characteristics; incisement and riparian type. Incisement was quantified using the entrenchment ratio, while the riparian was quantified using the two EMAP metrics XCDENMID and XCMGW. The dendrogram[2] in Fig. 6.1 below shows the groupings.

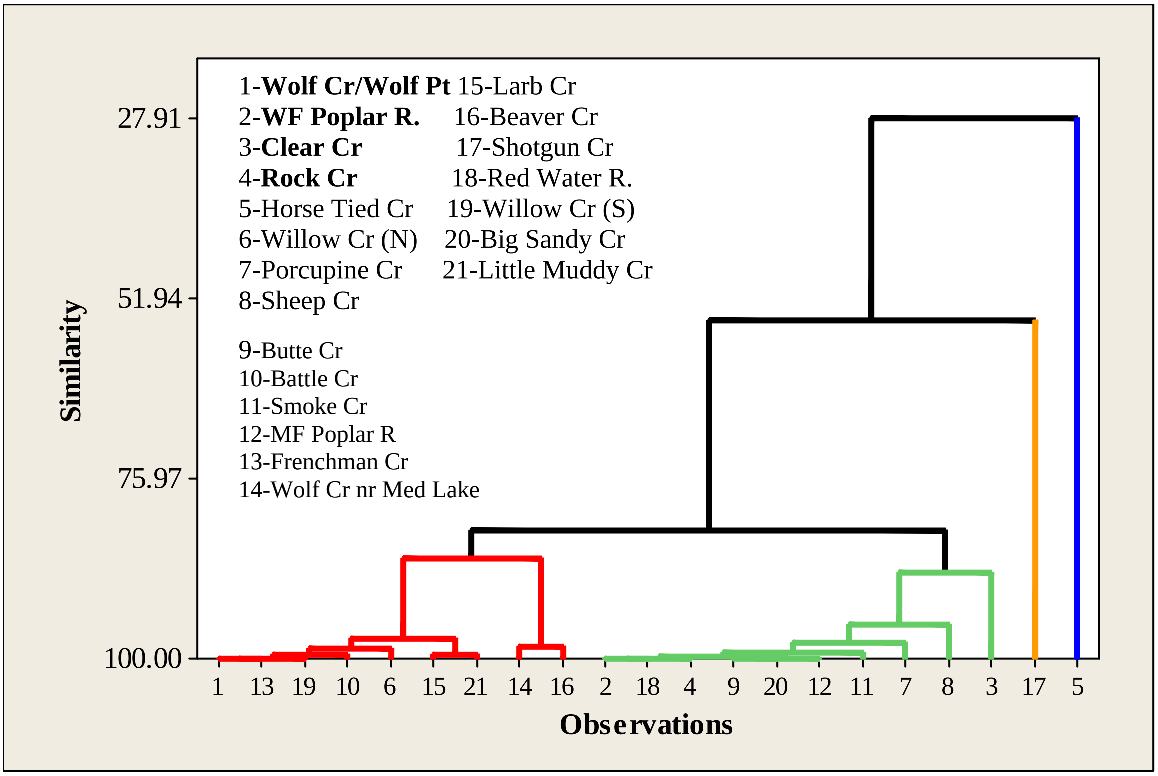


Figure 6.1. Dendrogram showing similarity of the 21 stream sites in the study, based on their entrenchment and two measures of the riparian community. The dendrogram was generated using standardized units, average linkage and squared Euclidean distance (Everitt et al. 2001). The comparison sites are printed in bold.

The scheme created two major groups (shown in red and green; Fig. 6.1) and identified two stream sites that did not fit into either group. The two major groups in Fig. 6.1 can be summarized as follows:

<u>Red group</u>: This group is comprised of entrenched, Rosgen G and F stream sites that had generally low levels of canopy density (0-14.4%) and low to intermediate (0-58%) woody vegetation density. The riparian habitat types represented are either the "Recent Riparian Complex" or the "Shrub and Herbaceous Riparian Complex" types (see Fig.

[2] A dendrogram that clearly differentiates groups has small distances in the far branches of the tree (those at the very bottom of Fig. 6.1) and large differences in the near branches (those near the top of Fig. 6.1). In Fig. 6.1, there are clear differences between the red group, green group, and the two individual sites (note the longer length of the near branches compared to the shorter lengths of the far branches).

4.4). It contains two somewhat distant members on the right side, Wolf Cr nr Medicine Lake (No. 14) and Beaver Creek (No. 16). These two stream sites had higher woody vegetation densities then any other members of the Red group, but were otherwise similar in other respects.

Green group: This group is comprised of the moderately to slightly entrenched streams (Rosgen B, C, and E classes) that had low canopy density (0-4.4%) and low to moderate (0-35%) woody vegetation density. The riparian habitat types represented were either the “Recent Riparian Complex” or the “Shrub and Herbaceous Riparian Complex”. The group has one notable outlier, Clear Cr (No. 3). Clear Creek had a much higher canopy density then most other sites (25.4%) in this group, as well as a unique riparian habitat type (Intermittent Riparian Coulee). And although not used as part of the grouping scheme, Clear Creek had a unique macrophyte population (see sub-Section 8.2.2), a relatively higher gradient, and was the only stream in the group with calcium-bicarbonate water chemistry.

Others: The dendrogram in Fig. 6.1 shows Horse Tied Creek (No. 5) and Shotgun Creek (No. 17) as separate from all of the other sites. Horse Tied Creek had a ‘Woody Draw’ riparian type, while Shotgun Creek was an ‘Oxbow/cattail Marsh’ (see Fig. 4.4). Both stream sites had very high canopy densities (60.0 % and 63.0 %, respectively) but completely contrasting woody vegetation. Horse Tied Creek had the highest woody vegetation cover in the study (83.0%), while Shotgun Creek had among the lowest (0.0 %). Horse Tied and Shotgun creeks should each be considered unique cases in this study, for which there are no other members.

6.2 Use and Application of the Groups

In Table 3.2 of Section 3.0, each stream site was presented in order of its overall condition ranking, a convention followed through the report to this point. The remainder of this report will deal with the biological and nutrient characteristics of the streams. In the remaining sections, data will be presented ordered by the cluster groups shown in Fig. 6.1, in order to demonstrate how the physical framework of the stream sites influences the biological communities found within them.

Caution should be used when using data from Clear Creek as a point of comparison to other prairie streams. Although the cluster analysis includes it with other members of the Green Group, it has a number of very unique attributes such as its water chemistry, gradient, and location in the foothills of the Bear Paw Mountains that make it something of an outlier from the main Green Group cluster. Caution should also be used when applying these groups to streams over long periods of time. It has been demonstrated that stream channels go through natural geomorphic evolution (Rosgen 1996), this being particularly true for incised channels such as those in the Red Group. Over time, a stream that is currently a G or F channel may develop a new floodplain within the incised gulley and become a C or E channel. This evolutionary process has apparently already occurred in Rock Creek. When a channel-type change of this type occurs, the stream site should be re-evaluated to determine if it would not be better as a member of the Green Group, or perhaps some yet-to-be-described group.

SECTION 7.0
NUTRIENTS (NITROGEN AND PHOSPHORUS)

7.1 Nutrient Limitation as Indicated by Benthic Algal C:N:P Ratios, Water Column Nutrient Concentrations, and Alkaline Phosphatase Activity

The data strongly suggest that N is the major limiting nutrient in these streams. This conclusion comes from three lines of evidence.

The first line of evidence comes from the C, N and P content of the algae themselves. A direct way to ascertain algal nutrient limitation is to measure algal cellular nutrient concentrations and then compare the values to the "Redfield" ratio. When phytoplankton algae are growing at maximal rates, they develop cellular C, N, and P concentrations in the ratio of 41:7:1 (by weight; also reported as a molar ratio of 106:16:1); this ratio is commonly referred to as the Redfield ratio (Redfield 1958; Goldman et al. 1979). Deviations from these ratios are indicative of either N or P limitation. High N:P ratios indicate P is limiting growth, low N:P ratios suggests that N is limiting, and high C:N ratios indicate that N is limiting (Healey 1973b; Healey and Hendzel 1980; Kahlert 1998; Hillebrand and Sommer 1999; and others).

The acid-rinsing procedure (described in Section 2.3.4) was found to be the best way to prepare the C:N samples. (A detailed methodological comparison of acid vs. non-acid washed samples is discussed in Appendix B.) Therefore, only samples that received acid rinsing are shown. There were analytical problems with the P analysis procedure in 2001, so only data for C and N content of the algal cells (which was analyzed separately) are shown. The data were examined by considering the slope of the regression line relative to the Redfield ratio, or slope (Fig. 7.1). The slope of the C vs. N regression line in Fig 7.1 is 15.4, much higher than the slope of the "idealized" line (6.6) at the Redfield ratio (note that these are molar ratios). Kahlert (1998) suggests that for benthic algae a molar C:N ratio greater 11 indicates N limitation, while Hillebrand and Sommer (1999) suggest that for periphyton a molar C:N ratio greater then 10 indicates N limitation. Either way, the results in Fig. 7.1 indicate that in 2001 the algae were N limited.

In 2002 data were again examined by regression analysis, but this time include the P content of the cells as the analytical problems of 2001 had been resolved (Fig. 7.2; values molar). Taken together, these analyses indicate that N was the most limiting nutrient. The slope of the C vs. N regression line (Fig. 7.2A) is 7.9, slightly higher than the slope of the "idealized" slope of 6.6 at the Redfield Ratio but not quite as high as the value (about 10) that has been suggested to indicate N limitation (Kahlert 1998; Hillebrand and Sommer 1999). Essentially, Fig. 7.2A suggests that N is optimal for the algae or starting to become limiting. On the other hand, both the P vs. C and the N vs. P regressions have slopes lower (often much lower) than Redfield (Fig. 7.2B & C). Both Kahlert (1998) and Hillebrand and Sommer (1999) suggest that molar N:P ratios lower than about 12

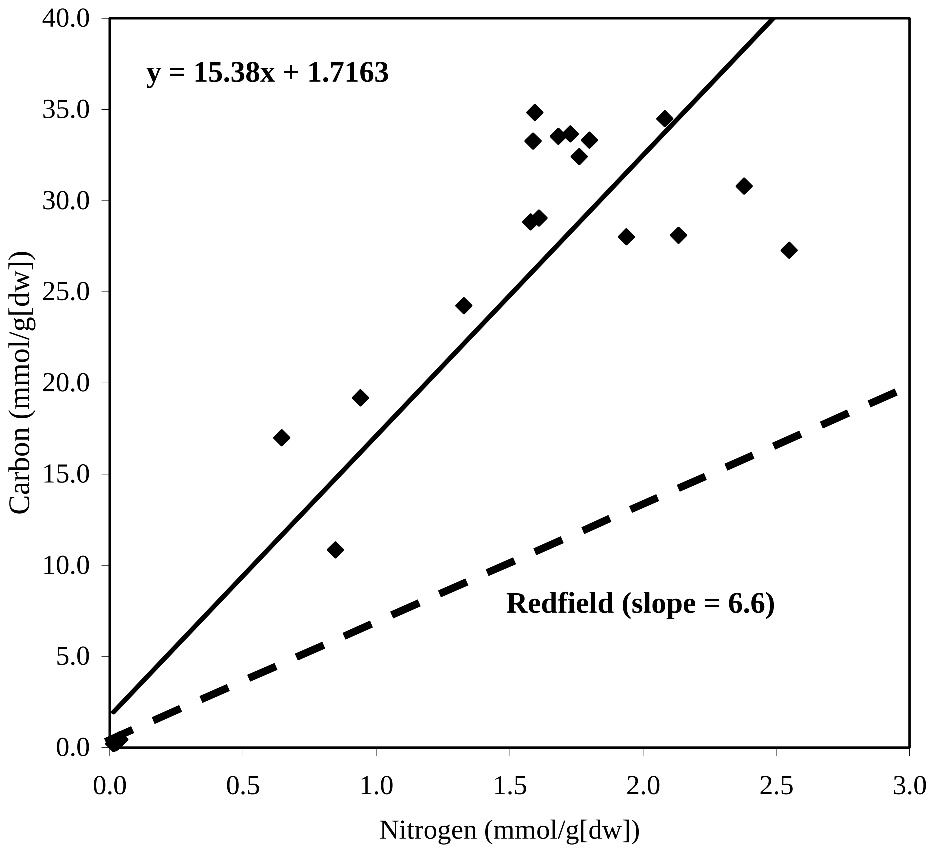


Figure 7.1. Ratio of benthic algae cellular nitrogen and carbon content. Slope of line of samples is 15.4 and is compared to the Redfield ratio (dotted line).

indicate N limitation, therefore the N:P ratio of 5.8 in Fig. 7.2C, taken together with the results of 7.2A, indicates N limitation in these algae.

Almost all of the nutrient ratios were high in 2002 (note the high Y-intercepts in Fig. 7.2), suggesting that there was a fair amount of detrital C in the samples (Allan 1995). This is supported by observation of the collected material, which frequently had some proportion of dead algal material in it especially as the season progressed toward fall. Furthermore, filamentous algae (e.g., *Cladophora*, *Spirogyra*), which composed the bulk of many samples (Table 8.2), tend to have higher cellular C content then diatom algae (Kahlert 1998) and this would result in higher ratio values when C is involved.

Taken together, the data in Figs. 7.1 and 7.2 indicate that the algae were N limited. An independent scientist with expertise in this area also reviewed the data and concluded that it indicated N limitation in these streams (Cotner, personal communication).

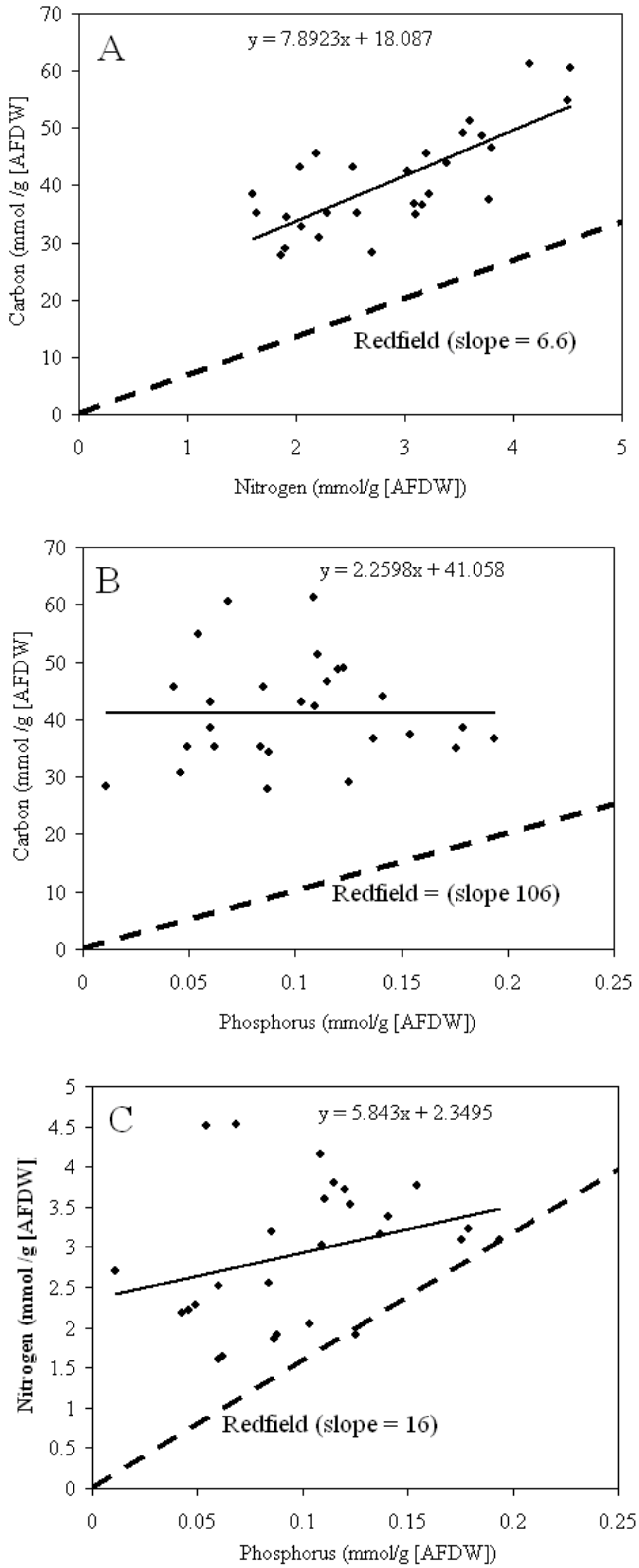


Figure 7.2. Comparison of benthic algal nutrient ratios in this study to the Redfield ratio (dotted line). Regression equations for the data points (acid-treated samples only) are found at the top of each graph. A. Carbon vs. nitrogen. B. Carbon vs. phosphorus. C. Nitrogen vs. phosphorus.

The second line of evidence comes from the study's water-column nutrient data. Water-column nutrient concentrations can be used as surrogates for algal cellular concentrations (Schindler 1977; Watson et al.1990; Lohman and Priscu 1992; Chessman et al. 1992). Stream water N:P ratios can influence algal cellular N:P ratios (Stelzer and Lamberti 2001), but aquatic plants also have the ability to influence the dissolved N and P concentrations of the water column via uptake and release of N and P, and therefore water-column ratios ultimately may come to resemble cellular ratios (Redfield 1958; Redfield at al. 1963). Just as was the case for cellular ratios, water N:P ratios lower than Redfield indicate N limitation. The N:P ratio of the dissolved fractions of inorganic N and P ($NO_{2/3}$ and TDP or SRP) in the present study's water samples ranged from zero to 7.91 (by weight), with a median of 0.27 (Table 7.1). These are *very* low water-column N:P ratios, as the by-weight Redfield ratio is 7.2, and strongly suggest that N is limiting growth of the aquatic plants. And it should be noted that the approach taken in Table 7.1 is conservative, in that the ratios shown are always the *highest* that could be calculated given the data. For example, $NO_{2/3}$ was frequently below the detection limit of 0.6 µg/L but in such cases 0.6 was used to calculate the N:P ratio. And whenever possible SRP rather than TDP data was used, which will almost always result in a higher N:P ratio.

The last line of evidence comes from a small amount of alkaline phosphatase activity (APA) data. Sayler et al. (1979) reported that APA activity levels of benthic rock algae scrapings from an uncontaminated stream in Tennessee range from 87,000 to 268,000 nmol g^{-1} (dry weight) hr^{-1}. APA activity values for benthic rock scrapings in this study only ranged from 273-2,602 nmol g^{-1} (wet weight) hr^{-1}. Converting this study's sample values to dry weight (not possible as that data was not obtained) would raise these values slightly, but certainly not into the range reported by Sayler et al. (1979). APA is produced by both algae and bacteria (Jones 1972), and low APA activity indicate that the organisms have sufficient soluble P available (Pick 1987). This study's streams had much lower APA activity then the stream in Tennessee, which suggests P is not (or much less) limiting in these streams.

In summary, all lines of evidence indicate that N is the major limiting nutrient in these streams.

Table 7.1. Soluble N:P ratios (by weight) in the water column of the study stream sites. All samples from all sites are shown. The by-weight Redfield N:P ratio is 7.

Stream Site	Collection Date	Soluble N:P ratio*†	Stream Site	Collection Date	Soluble N:P ratio*†
Clear Creek	5/30/2001	2.05	Frenchman Creek	5/31/2001	0.05
Clear Creek	6/22/2001	0.01	Frenchman Creek	6/24/2001	0.02
Clear Creek	8/21/2001	0.00	Frenchman Creek	8/23/2001	0.03
Clear Creek	9/20/2001	0.01	Frenchman Creek	9/22/2001	0.06
Wolf Creekat Wolf Point	5/18/2002	0.27	Battle Creek	5/30/2001	0.03
Wolf Creekat Wolf Point	6/13/2002	0.35	Battle Creek	6/22/2001	0.03
Wolf Creekat Wolf Point	7/17/2002	0.11	Battle Creek	8/21/2001	0.04
Wolf Creekat Wolf Point	8/14/2002	0.21	Battle Creek	9/21/2001	0.06
WF Poplar River	5/19/2002	0.96	Butte Creek	5/19/2002	2.87
WF Poplar River	6/16/2002	0.14	Butte Creek	6/16/2002	0.23
WF Poplar River	7/23/2002	0.08	Butte Creek	7/22/2002	0.07
WF Poplar River	8/11/2002	0.11	Butte Creek	8/11/2002	0.21
Rock Creek	6/25/2001	0.03	Wolf Creek nr Medicine Lake	5/19/2002	0.78
Rock Creek	6/25/2001	0.04	Wolf Creek nr Medicine Lake	6/14/2002	0.50
Rock Creek	8/25/2001	0.03	Wolf Creek nr Medicine Lake	7/21/2002	0.25
Rock Creek	9/22/2001	0.15	Wolf Creek nr Medicine Lake	8/12/2002	0.43
Rock Creek	5/18/2002	1.50	Larb Creek	6/23/2001	*no P data*
Rock Creek	6/17/2002	1.07	Larb Creek	8/22/2001	0.03
Rock Creek	7/24/2002	0.08	Larb Creek	9/23/2001	0.05
Rock Creek	8/15/2002	0.50	Beaver Creek	5/31/2001	0.02
Horsetied Creek	5/19/2002	0.46	Beaver Creek	6/23/2001	0.05
Horsetied Creek,	6/14/2002	0.17	Beaver Creek	8/22/2001	0.02
Horsetied Creek	7/20/2002	0.01	Beaver Creek	9/23/2001	0.02
Horsetied Creek	8/13/2002	0.09	Shotgun Creek	5/19/2002	0.62
Willow Creek (N)	5/31/2001	0.02	Shotgun Creek	6/14/2002	0.02
Willow Creek (N)	6/24/2001	0.02	Shotgun Creek	7/19/2002	0.02
Willow Creek (N)	8/23/2001	0.02	Shotgun Creek	8/13/2002	0.02
Willow Creek (N)	9/22/2001	0.03	Smoke Creek	5/19/2002	0.43
Willow Creek (N)	5/18/2002	0.33	Smoke Creek	6/14/2002	0.42
Willow Creek (N)	6/17/2002	0.66	Smoke Creek	7/21/2002	0.97
Willow Creek (N)	7/24/2002	0.28	Smoke Creek	8/12/2002	0.78
Willow Creek (N)	8/15/2002	7.91	Willow Creek (S)	6/25/2001	*no P data*
Porcupine Creek	6/1/2001	0.35	Willow Creek (S)	8/24/2001	0.03
Porcupine Creek	6/22/2001	0.03	Willow Creek (S)	9/21/2001	0.02
Porcupine Creek	8/24/2001	0.04	Big Sandy Creek	5/30/2001	0.01
Porcupine Creek	9/21/2001	0.02	Big Sandy Creek	6/21/2001	0.02
Porcupine Creek	5/18/2002	0.16	Big Sandy Creek	8/20/2001	0.01
Porcupine Creek	6/17/2002	0.55	Big Sandy Creek	9/20/2001	0.02
Porcupine Creek	7/23/2002	0.04	Red Water River	5/18/2002	0.48
Porcupine Creek	8/14/2002	0.34	Red Water River	6/13/2002	0.47
Sheep Creek	5/18/2002	0.41	Red Water River	7/17/2002	0.20
Sheep Creek	6/13/2002	0.41	Red Water River	8/14/2002	0.33
Sheep Creek	7/19/2002	0.17	Little Muddy Creek	5/19/2002	0.71
Sheep Creek	8/14/2002	0.79	Little Muddy Creek	6/14/2002	0.09
MF Poplar	5/19/2002	0.08	Little Muddy Creek	7/20/2002	0.02
MF Poplar	6/16/2002	0.02	Little Muddy Creek	8/13/2002	0.01
MF Poplar	7/22/2002	0.13			
MF Poplar	8/11/2002	0.04			

*Soluble N is the NO_{2+3} concentration. In cases where the sample was below the method detection limit (0.6 ug/L), 0.6 is used to calculate the ratio. Ammonia was below the detection limit in all but 3 sample and was not used to calculated soluble N.

† Soluble P values used are TDP (2001 data) or SRP (2002 data).

7.2 Nitrogen and Phosphorus Concentrations in the Stream Water

Summary nutrient data for each stream site are shown in Table 7.2 (page 64) and are arranged by the grouping scheme (Red, Green, or Other) developed in Section 6.0. There are a number of notable features in Table 7.2. Total N ranged from 236.6 µg/L (Wolf Cr at Wolf Point) to 5,575.7 µg/L (Horse Tied Creek). Average $NO_{2/3}$ was low in all of the streams, and 45% of the $NO_{2/3}$ samples were below the detection limit of 0.6 µg/L[1]. In contrast, SRP was always detectable in the water and ranged from 1.2 (Wolf Cr nr Medicine Lake) to 254.0 µg/L (Shotgun Creek). Total P ranged from 15.8 (Wolf Cr at Wolf Point) to 2,791.2 µg/L (Willow Cr {S}). The Red and Green groups both had about the same mean TN and TDP concentrations, but the Red group's mean $NO_{2/3}$, TP, and SRP concentrations were more than twice those of the Green group (Table 7.2).

The data summarized in Table 7.3 show the mean and range of nutrients for the comparison sites vs. the entire dataset. Concentration ranges for all parameters were greater in the complete set of sites then in the comparison sites, although some of the median values were quite similar (e.g., $NO_{2/3}$, TP, and TDP). As mentioned above, many of the $NO_{2/3}$ concentrations were below the method detection limit of 0.6 µg/L; however, an accurate median could be calculated for $NO_{2/3}$ since the total number of non-detects was < 50% of the dataset (Helsel and Hirsch 1992). The TN concentration of the comparison sites and of the complete dataset are shown in box and whisker plots in Fig. 7.4A&B. (An explanation of the box and whisker plots used here and elsewhere in this report is shown in Fig. 7.3.) TN concentrations tended to be lower in the comparison sites than in the complete data set, although the difference is not quite significantly different (p-value = 0.06) based on the Mann-Whitney test for difference in medians. There was much more overlap in TP concentrations between the complete dataset and the comparison sites, and the medians were not found to be significantly different (p-value = 0.77; Figs. 7.5A&B). Interestingly, the median TP of the comparison sites was slightly higher than that of the complete dataset.

The median and interquartile range of N and P concentrations for the Red and Green groups are shown in Figs 7.6 and 7.7. In spite of the groups' morphological and riparian habitat differences, their median TN and TP concentrations are virtually identical. Note that this approach to summarizing the data, which shows the two groups having similar median TP values, contrasts with the *mean* TP values for the Red and Green groups in Table 7.2 which shows that the Red group's mean TP is more than twice that of the Green group. Medians (such as those in Figs. 7.6 and 7.7) diminish the influence of unusually high or low concentration values in a dataset, whereas such values are incorporated into calculations of means and provide a very different (sometimes skewed) view of the data's central tendency.

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[1] Overall data quality for $NO_{2/3}$ and other nutrient parameters was good. Field filter blanks were almost always below the detection limit of the different analyses, with the exception of ammonia (discussed in Section 2.3.3), and TDP. TDP typically had concentrations of about 2.0 µg/L above the method detection limit of 0.4 µg/L. Replicate laboratory analyses (repeated measures) and duplicate field samples were well within the QC limit (<10% relative standard deviation) of the analysis plan, with the exception of (again) a few cases for ammonia.

Time vs. concentration plots were generated to assess (visually) whether or not there were any obvious temporal changes in nutrient concentrations over time. No clear temporal patterns (e.g., increasing or decreasing concentrations during the growing season) for nutrients could be discerned in the comparison sites or in the study sites as a whole. Inter-annual variation for sites sampled in both 2001 and 2002 was also very high and no trends were noted. It should be borne in mind that the time period of the study (2 years) was probably too short to discern trends, however interested readers are referred to Appendix C where a number of time vs. concentrations plots will be found.

Table 7.3. Median and range of concentrations of nitrogen and phosphorus measured in the comparison sites and in all study sites, 2001 and 2002. The median is shown, followed by the range in parenthesis.

	Comparison sites		All study sites	
Parameter	Number samples	Median and range (μg/L)	Number samples	Median and range (μg/L)
Total Nitrogen (TN)	20	712.9 (236.5-1,952.6)	94	979.0 (180.4-5,575.7)
Nitrate + Nitrite ($NO_{2/3}$)	20	0.8 (0.0-89.1)	93	0.8 (0.0-318.1)
Total Phosphorus (TP)	20	98.9 (15.8-381.8)	94	82.9 (15.8-2,791.1)
Total Dissolved Phosphorus (TDP)	20	30.9 (9.2-256.8)	92	29.7 (5.2-335.1)
Soluble Reactive Phosphorus (SRP)*	12	4.7 (1.7-28.6)	56	26.9 (1.2-254.0)

*SRP data collected in 2002 only.

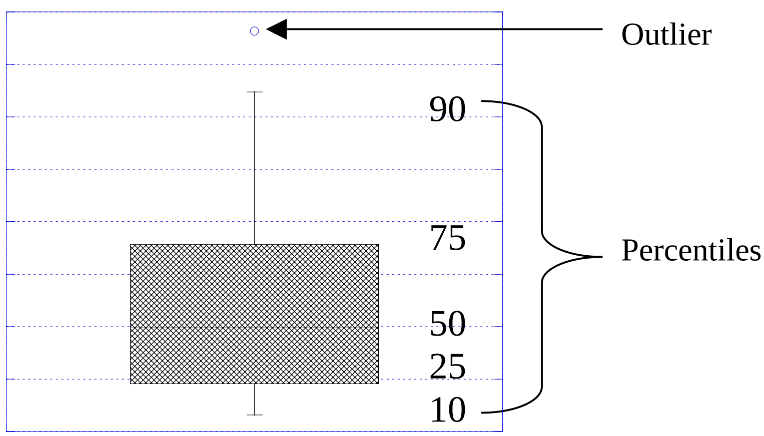


Figure 7.3. Definitions for symbols in box plots of Figure 7.4, 7.5, and elsewhere in this report. Outliers are values found outside the 90th (or 10th) percentile, but are sed as part of the calculation of the box and whisker plot.

Table 7.2. Means and ranges of nutrients measured in the study streams, 2001-2002. All data were collected in spring and summer.

Stream Site Name	Year	Condition Rank	Total N (µg/L)	NO_2+NO_3 (µg/L)†	Total P (µg/L)	Total Dissolved P (µg/L)	Soluble Reactive P (µg/L)
RED GROUP STREAMS							
Wolf Cr at Wolf Point	2002	Min. impacted	584.7 (236.6-1141.5)	0.6 (all 0.6)	52.5 (15.8-124.2)	18.0 (9.2-29.5)	3.0 (2.2-5.2)
Frenchman Creek	2001	Intermediate	1056.0 (782.4-1476.6)	0.9 (0.6-1.4)	132.0 (50.5-241.0)	24.9 (20.4-31.9)	not collected
Willow Creek (S)	2001	Most impacted	1109.7 (838.6-1296.7)	0.6 (all 0.6)	983.8 (74.6-2791.2)	22.4 (19.4-25.4)	not collected
Battle Creek	2001	Intermediate	780.1 (689.0-842.7)	0.6 (all 0.6)	29.5 (18.3-44.6)	16.3 (10.7-21.0)	not collected
Willow Creek (N)*	01 & '02	Intermediate	891.0 (481.0-1506.9)	44.3 (0.6-318.1)	170.7 (60.3-545.5)	44.4 (20.9-100.4)	23.5 (1.8-40.2)
Larb Creek	2001	Most impacted	846.3 (752.7-901.6)	3.4 (0.6-7.4)	900.7 (164.6-1860.5)	143.0 (45.8-240.2)	not collected
Little Muddy Creek	2002	Most impacted	1303.2 (1087.6-1565.9)	3.0 (1.5-7.1)	155.8 (92.2-224.7)	102.0 (44.0-155.6)	67.8 (9.9-136.6)
Wolf Cr nr Medicine Lake	2002	Most impacted	1224.6 (862.3-1482.3)	1.9 (0.6-2.9)	44.9 (28.2-60.3)	22.3 (16.1-30.2)	5.2 (1.2-11.6)
Beaver Creek	2001	Most impacted	977.8 (959.0-994.2)	0.7 (0.6-1.1)	94.6 (54.8-165.4)	31.4 (23.5-39.0)	not collected
		Red Group Means:	**965.8**	**10.7**	**238.3**	**42.9**	**24.9**
GREEN GROUP STREAMS							
WF Poplar River	2002	Min. impacted	1062.8 (618.3-1952.6)	1.9 (0.6-4.0)	55.6 (30.4-62.2)	38.2 (21.9-67.1)	8.6 (4.2-15.7)
Redwater River	2002	Most impacted	1059.7 (970.1-1221.1)	0.6 (all 0.6)	96.8 (74.9-115.1)	17.6 (5.2-23.5)	1.8 (1.3-3.0)
Rock Creek*	01 & '02	Min. impacted	981.4 (633.4-1428.8)	5.1 (0.6-30.6)	129.1 (50.7-202.4)	33.8 (21.0-82.0)	10.4 (1.8-28.6)
Butte Creek	2002	Intermediate	1651.2 (1118.1-2565.6)	6.2 (2.0-16.6)	116.3 (82.6-167.6)	55.8 (33.6-79.8)	17.4 (5.8-39.0)
Big Sandy Creek	2001	Most impacted	1709.1 (1009.6-2266.5)	0.6 (all 0.6)	91.2 (27.8-215.8)	52.0 (24.4-107.1)	not collected
MF Poplar River	2002	Intermediate	1235.2 (970.4-1962.3)	1.6 (0.6-2.8)	89.8 (59.6-130.7)	69.6 (42.9-112.7)	33.6 (7.8-73.2)
Smoke Creek	2002	Most impacted	695.6 (537.6-861.9)	1.5 (0.6-2.9)	36.3 (31.5-40.4)	15.9 (14.1-19.0)	2.2 (1.4-3.7)
Porcupine Creek*	01 & '02	Intermediate	626.2 (180.4-2920.3)	1.4 (0.6-6.4)	34.7 (22.2-56.4)	18.1 (9.0-31.5)	5.8 (1.8-14.9)
Sheep Creek	2002	Intermediate	1580.4 (1414.2-1840.5)	1.7 (1.1-2.6)	113.8 (84.5-139.8)	30.5 (26.2-36.2)	5.0 (1.4-8.4)
Clear Creek	2001	Reference	550.6 (454.0-706.9)	23.1 (0.6-89.1)	235.2 (57.4-381.8)	164.3 (43.4-256.8)	not collected
		Green Group Means:	**1048.2**	**4.5**	**97.8**	**45.3**	**10.6**
OTHERS							
Shotgun Creek	2002	Most impacted	1998.5 (1278.0-3263.7)	5.1 (1.3-11.0)	229.0 (64.1-519.9)	177.3 (51.4-335.1)	129.3 (17.9-254.0)
Horse Tied Creek	2002	Intermediate	2034.5 (504.2-5575.7)	2.0 (1.7-2.4)	253.3 (50.6-590.1)	137.6 (29.2-267.6)	62.7 (5.1-208.1)

*Means and ranges shown are for the complete 2001 and 2002 dataset.

† Nearly half of all the $NO_{2/3}$ data was less than the detection limit of 0.6 µg/L. Means and ranges shown were calculated using 0.6 in cases where the sample value was < 0.6 g/L.

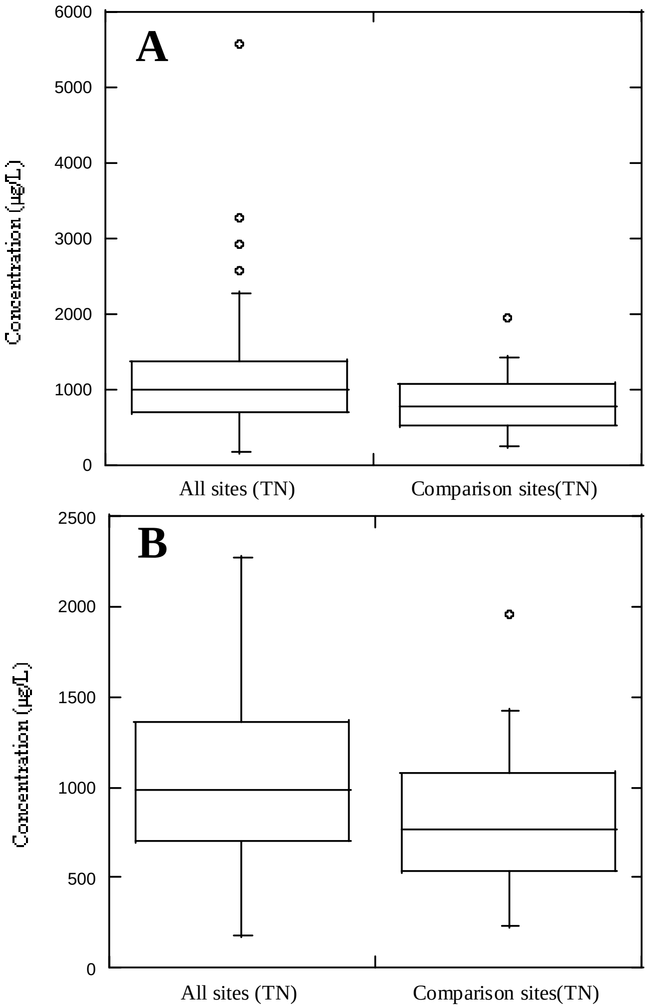


Figure 7.4. TN concentration in the study sites. A. All study sites and the comparision sites, showing all outliers. B. All study sites and the comparison sites, but Y axis scale reduced to better show interquartile range.

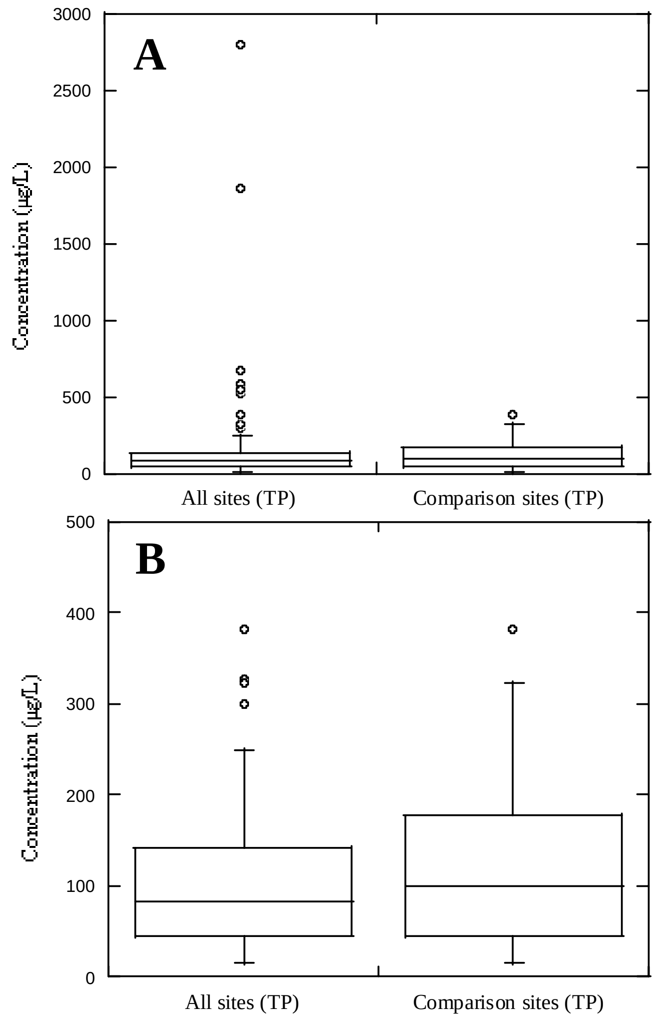


Figure 7.5. TP concentrations in the study sites. A. All sites and the comparison sites, showing all outliers. B. All sites and the comparison sites, with the Y axis scale reduced to better show interquartile range.

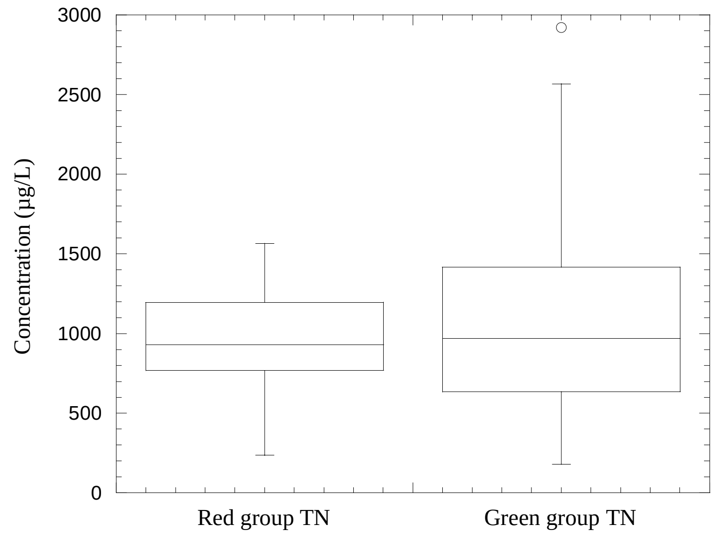


Figure 7.6. TN concentration of the streams in the Red and Green groups. All outliers are shown.

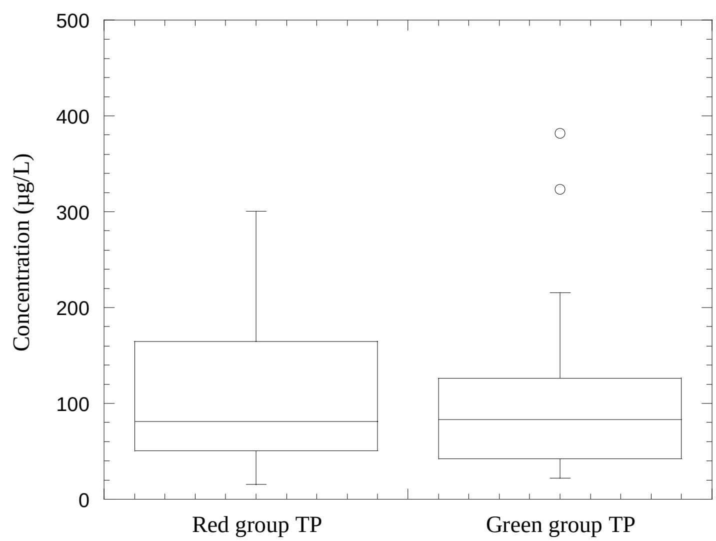


Figure. 7.7. TP concentration in streams of the Red and Green groups. Not all outliers shown; Y axis scale has been reduced to better shown interquartile range.

7.3 Soluble Reactive P (SRP) vs. Total Dissolved P (TDP)

In 2002 we compared measurements of SRP and TDP in the study streams. These two analyses are frequently the most common techniques used to determine dissolved P in water samples, and each measures somewhat different forms of P. Direct colorimetry of filtered (0.45 µm) samples is used to analyze for SRP and is largely a measure of available orthophosphate. TDP analysis is undertaken on samples that have been filtered, digested with a persulfate oxidation, and then measured using direct colorimetry (APHA 1998). The additional digestion step for TDP samples is intended to liberate dissolved organically bound P that may be present, therefore for any given water sample TDP concentrations should always be at least equal to SRP, and are frequently higher.

There is a strong and significant relationship ($r_s = 0.90$; p-value = 0.000) between TDP and SRP using Spearman rank correlation (Fig. 7.8), indicating the consistency of the two methods. To try to ascertain which of the two dissolved P measures may be better suited to prairie streams, a water quality measurement was selected that might act as a surrogate for dissolved organically bound P. TSS is probably the best surrogate available in this study, as increasing TSS in the streams (from higher flows and associated erosion) might co-occur with higher dissolved organic fractions in the water. It could therefore be expected that increasing TSS would correspond to higher organic P fractions and in turn higher TDP concentrations. However, there was no significant relationship between TSS and TDP, or for that matter between TSS and SRP (Fig. 7.9A & B). Given these results, it appears that TDP and SRP are probably both good methods to measure dissolved P in prairie streams. If nutrient limitation is of interest and dissolved N:P ratios are analyzed, it should be noted that if the soluble P fraction used is TDP the ratios will likely be lower than if SRP had been used.

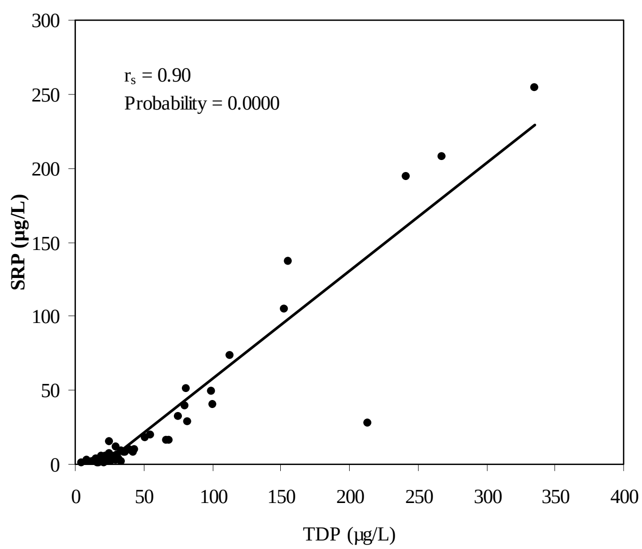
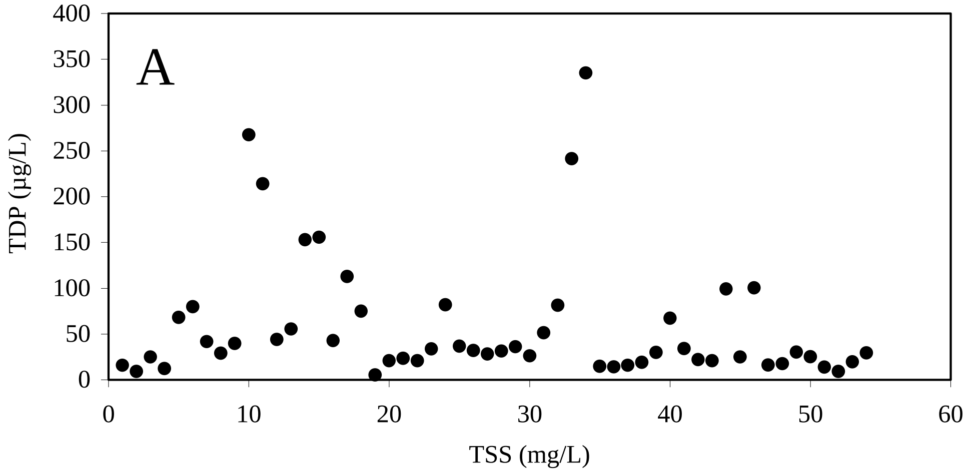


Figure 7.8. Relationship between total dissolved phosphorus (TDP) and soluble reactive phosphorus (SRP) in study streams, 2002.



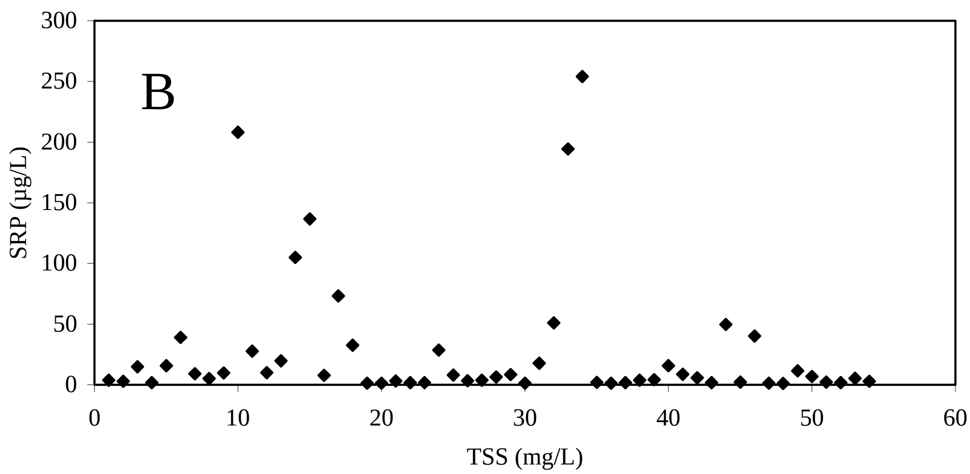


Figure 7.9. Relationship between different forms of soluble P and TSS in the study streams. A. TSS and TDP. B. TSS and SRP.

SECTION 8.0
BENTHIC & FLOATING ALGAE, MACROPHYTES, AND PHYTOPLANKTON

8.1 Quantitative Measurements

8.1.1 Benthic & Floating Algae, Macrophytes, and Phytoplankton

A total of 474 individual quantitative samples of aquatic plants were collected in the study streams (Table 8.1). Core and template samples were considered 100% algae, as these sampling methods were designed such that macrophytes were not included. For each hoop sample, during collection a visual estimate was made of the proportion of macrophytes vs. filamentous (floating & benthic) algae. The estimates were based on the areal coverage each plant-type contributed to the sample. Most samples collected in the study were either hoops or cores (total of 90%; Table 8.1), indicating that floating & benthic filamentous algae and macrophytes (hoop samples) and fine-sediment algal films (core samples) were widespread in Hi-line streams, whereas rocky substrates with diatom growth (template samples) were not.

Table 8.1. Summary of sampling efforts for floating & benthic algae and macrophytes.

Stream Site Name	Months sampled	Number samples (hoop)	Number samples (template)	Number samples (core)	Site totals
RED GROUP STREAMS					
Wolf Creek at Wolf Point	July, Aug	16	0	0	16
Frenchman Creek	Aug, Sept	0	4	18	22
Willow Creek (S)	Aug, Sept	0	0	19	19
Battle Creek	Aug, Sept	18	0	0	18
Willow Creek (N) †	July, Aug, Sept	0	7	29	36
Larb Creek	Aug, Sept	0	0	19	19
Little Muddy Creek	July, Aug	20	0	1	21
Wolf Creek nr Medicine Lake	July, Aug	21	0	0	21
Beaver Creek	Aug, Sept	0	0	22	22
Red Group Totals:		**75**	**11**	**108**	**194**
GREEN GROUP STREAMS					
West Fork Poplar River	June, July, Aug	14	17	3	34
Redwater River	July, Aug	17	3	1	21
Rock Creek†	July, Aug, Sept	0	9	24	33
Butte Creek	July, Aug	16	4	0	20
Big Sandy Creek	Aug, Sept	22	0	0	22
Middle Fork Poplar River	July, Aug	16	3	1	20
Smoke Creek	June, July, Aug	16	3	2	21
Porcupine Creek †	July, Aug, Sept	38	0	0	38
Sheep Creek	July, Aug	9	0	3	12
Clear Creek	Aug, Sept	21	0	0	21
Green Group Totals:		**169**	**39**	**34**	**242**
OTHERS					
Shotgun Creek	July, Aug	10	0	12	22
Horse Tied Creek	July, Aug	2	0	14	16
	Grand Totals:	**256**	**50**	**168**	**474**

† Data collected in 2001 and 2002.

The term "total aquatic plant biomass" used in this Section refers to the combined biomass of floating & benthic algae and macrophytes, measured as mg Chl *a*/m^2. Total aquatic plant biomass ranged from zero (Larb Creek) to 1,994 (Middle Fork Poplar River) mg Chl *a*/m^2. Summertime means and standard errors for total aquatic plant biomass at each site are shown in Fig. 8.1A, along with estimates of the average summertime proportion of macrophytes vs. floating & benthic algae in the samples (Fig. 8.1B). Estimates of the biomass of macrophytes vs. algae were made for each site by multiplying the mean total aquatic plant biomass (mg Chl a/m^2) by the mean visually estimated proportion of each plant type (either macrophyte or algae; Table 8.2). These values were then converted to weight values by multiplying the result by the g AFDW/mg Chl *a* ratio specific to each plant type. The last step was performed so that the estimate could better account for the variable Chl *a* per unit weight that was measured in the two different aquatic plant types.

There are several interesting features in Fig. 8.1A&B. Note that over half of the stream sites in the Red group have average total aquatic plant biomass under 20 mg Chl *a*/m^2, and none of the samples from these five sites contained macrophytes (field notes indicate that rare patches of macrophytes were observed in Beaver Creek). And the site with the highest average total aquatic plant biomass in the Red group, Wolf Cr at Wolf Point, is also the only minimally impacted site for that group. On the other hand, all of the Green group streams had on average more than 20 mg Chl *a*/m^2 total aquatic plant biomass, usually much more, and all of the samples had a sizeable proportion of macrophytes except for Rock Creek. (Rock Creek exhibited macrophyte patches scattered throughout the reach, but none ended up in any of the samples.) Note also that 3 out of the 4 reference and minimally impacted sites are found in the Green group.

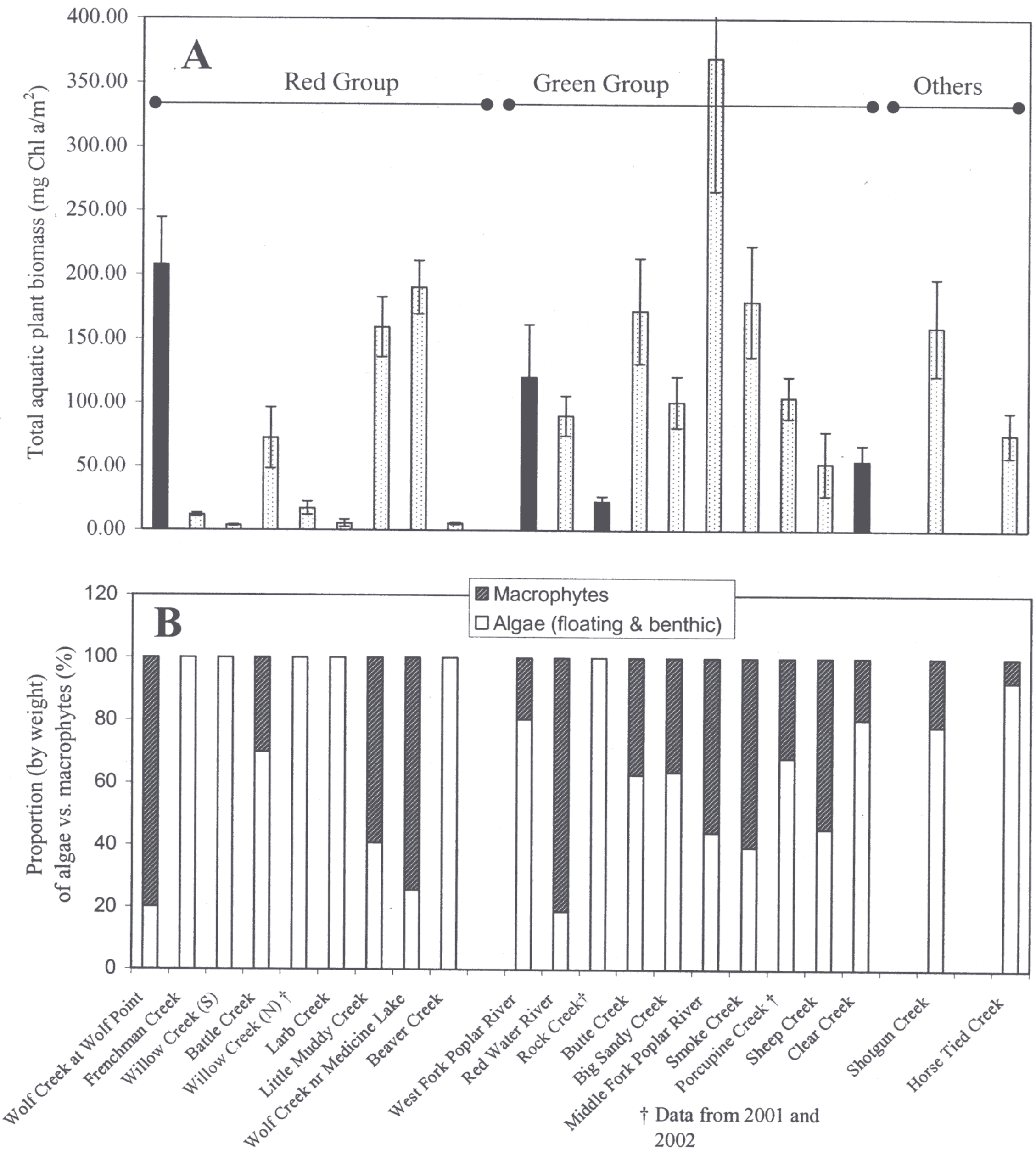


Figure 8.1. A. Average summer (June to September) total aquatic plant biomass in each of the study streams. Error bars are one standard error of the mean. Comparison sites are shown in black. B. Average summer proportions of algae (floating & benthic) and macrophytes in each study stream site. Values are estimates (see text) and are based on weight.

Table 8.2. Average of visually-estimated proportions of algae (floating and benthic) and macrophytes in each study site.

Stream Site Name	Year	Mean of estimated algal proportions in samples	Mean of estimated macrophyte proportions in samples
RED GROUP STREAMS			
Wolf Creek at Wolf Point	2002	13	87
Frenchman Creek	2001	100	0
Willow Creek (S)	2001	100	0
Battle Creek	2001	58	42
Willow Creek (N) †	01 & '02	100	0
Larb Creek	2001	100	0
Little Muddy Creek	2002	29	71
Wolf Creek nr Medicine Lake	2002	17	83
Beaver Creek	2001	100	0
GREEN GROUP STREAMS			
West Fork Poplar River	2002	78	22
Redwater River	2002	12	88
Rock Creek †	01 & '02	100	0
Butte Creek	2002	50	50
Big Sandy Creek	2001	51	49
Middle Fork Poplar River	2002	32	68
Smoke Creek	2002	28	72
Porcupine Creek †	01 & '02	56	44
Sheep Creek	2002	33	67
Clear Creek	2001	71	29
OTHERS			
Shotgun Creek	2002	68	32
Horse Tied Creek	2002	88	12

† Means based on data collected in both 2001 and 2002.

Because total aquatic plant biomass is comprised of algae and macrophytes mixed together, the proportional contribution to biomass by algae alone could only be estimated, as shown in Fig. 8.1. In order to have a more accurate measure of floating and benthic algal biomass in the streams, samples in the dataset that were (based on field notes) at least 90% algae were identified, and are shown in Fig. 8.2. (Only 4 samples of the 244 used to generate Fig. 8.2 were less then 100% algae.) It was decided that at least three all-algae samples would have to have been collected in order for a mean and standard error to be shown for a site in Fig. 8.2.

Floating & benthic algal Chl *a* ranged from a minimum of 0.0 mg/m^2 in Larb Cr to a maximum of 651.0 mg/m^2 in Shotgun Cr. In general, the Green group had higher densities of floating and benthic algal Chl *a* than did the Red group. Fig. 8.3 shows box plots of floating & benthic algal Chl *a* in the comparison sites. The median algal Chl *a* in the comparison sites was 19.7 mg/m^2, the mean 43.6 mg/m^2. Ninety percent of the algal Chl *a* values measured in the comparison sites fell between 6 mg/m^2 and 106 mg/m^2.

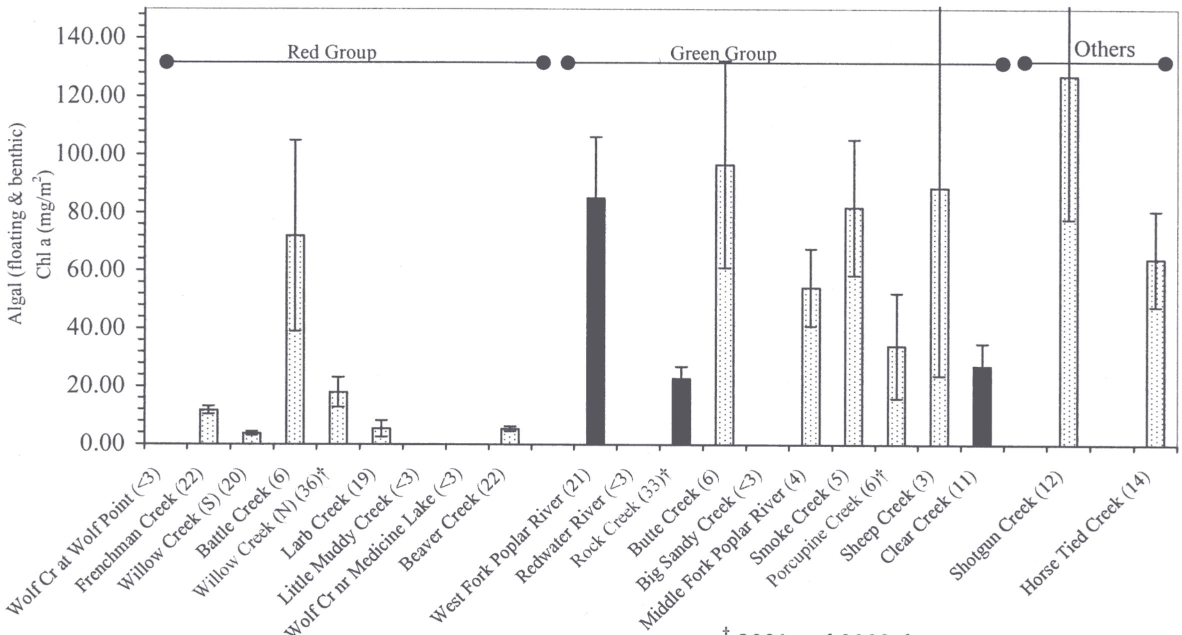


Figure 8.2. Mean summer algal (floating & benthic) Chl *a* in the study sites. Error bars are one standard error of the mean. Only samples that were 90-100% algae (based on field notes) are shown; number of such samples per site is shown in parenthesis following the site names. If a stream site had less than 3 all-algae samples, data has not been shown. Comparison sites are shown as black bars.

Figure 8.4 and Fig. 8.5 show visually estimated streambed coverage by macrophytes or filamentous algae (both floating & benthic), respectively. Like Fig. 8.1A&B, Fig. 8.4 shows that 4 out of 9 of the Red group streams had no or only trace macrophyte cover on the stream bottom, whereas all of the Green group streams had at least 5% coverage and usually much more. Average coverage of the stream bottom by filamentous algae ranged from zero to 30% in all the sites with one exception, Big Sandy Creek (Fig. 8.5). Big Sandy Creek had 68% streambed coverage by filamentous algae and was a major outlier in this regard. It is very likely that Big Sandy Creek is polluted with excess N (most probably nitrate) from saline seeps, as saline seeps are a problem in the basin (MT DEQ 2002b) and nitrate can be highly elevated in saline seeps (Brown et al. 1982; SD DENR 2004). This topic will be explored further in sub-Section 8.1.2.

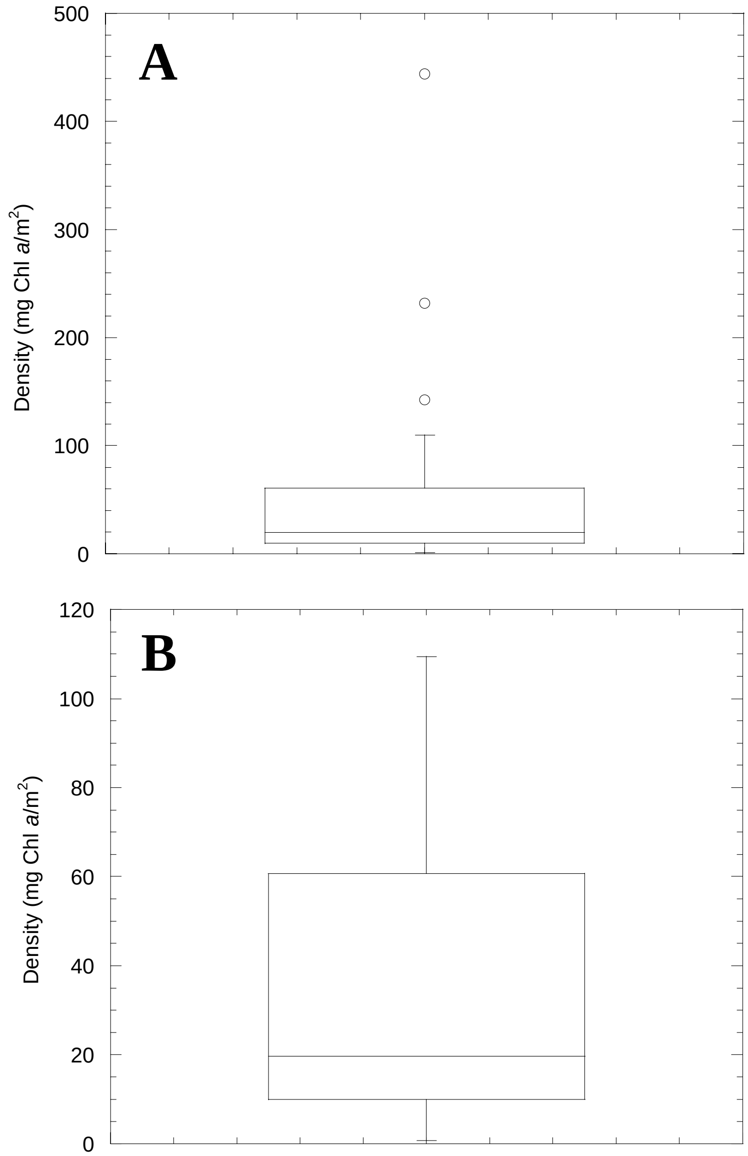


Figure 8.3. Boxplots of floating & benthic algal Chl *a* in the comparison sites. A. Showing all outliers. B. Y axis scale reduced to better show interquartile range.

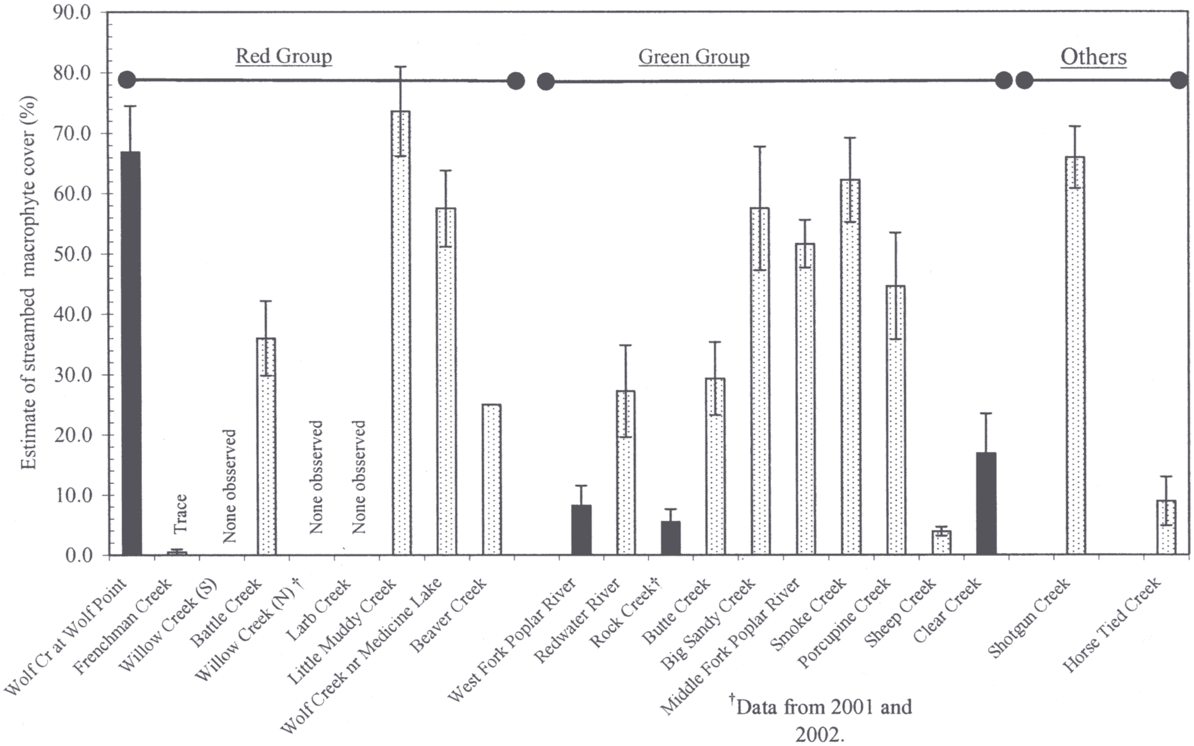


Figure 8.4. Average summer (June through September) stream bottom coverage by macrophytes in the study sites. Coverage was based on the EMAP habitat metric XFC_AQM (Kaufmann et al. 1999), derived from visual estimates of areal coverage by macrophytes. Error bars are 1 standard error of the mean. Bars shown in black are the comparison sites.

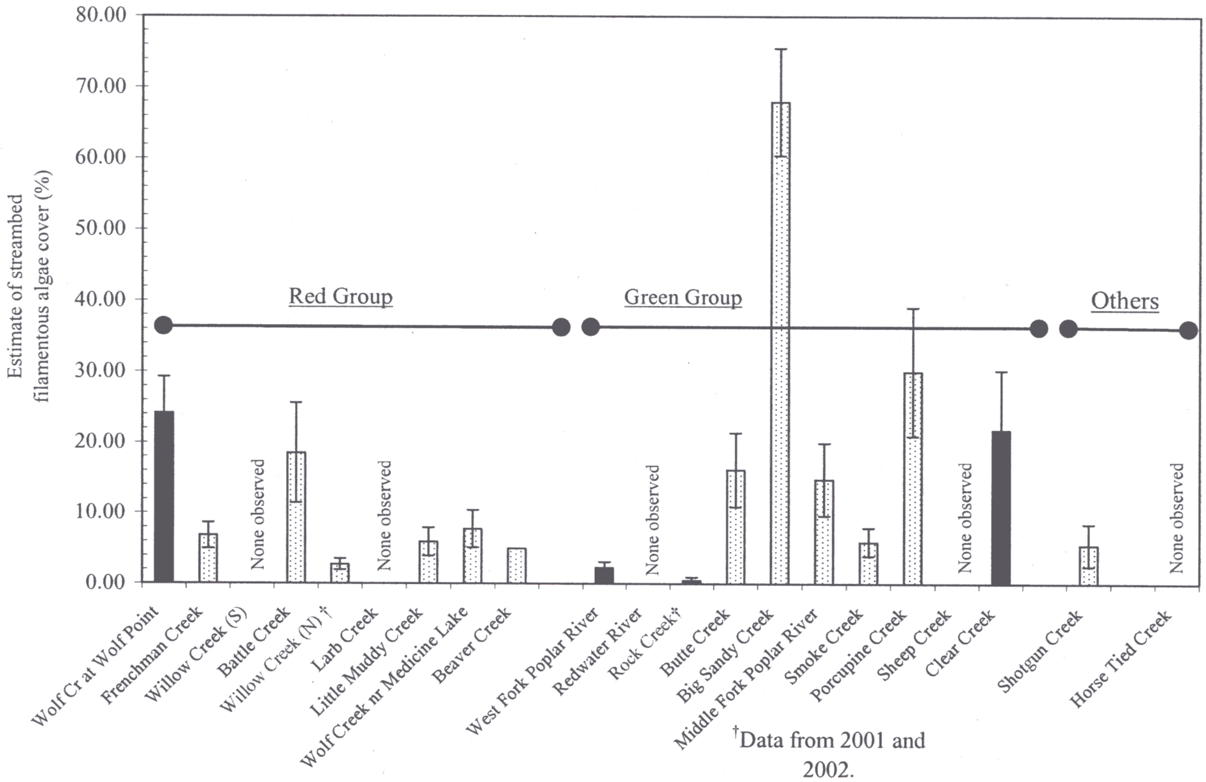


Figure 8.5. Average summer stream bottom coverage by floating & benthic filamentous algae in the sites. Coverage was based on the EMAP habitat metric XFC_ALG (Kaufmann et al. 1999), derived from visual estimates of areal coverage by filamentous algae. Error bars are 1 standard error of the mean. Bars shown in black are comparison sites.

Phytoplankton biomass (quantified as stream water Chl *a*) was also measured in most of the streams, although not as frequently as was floating & benthic algae. Phytoplankton Chl *a* concentrations in the study ranged from 0.4 µg/L (West Fork Poplar River) to 515.8 µg/L (Horse Tied Creek). The summertime means and standard errors for phytoplankton for each stream site are shown in Fig. 8.6. No clear difference is evident between the Red and Green groups, although there is a reasonable explanation for the unusually high values in Horse Tied Creek and Wolf Creek nr Medicine Lake. Both of these streams were intermittent during the summer, and each was directly impacted by heavy cattle use. Horse Tied Creek is a deeply incised channel that cattle would walk into and then remain for some time. The low-volume, isolated pools in the stream bottom then received large amounts of cattle excrement and urine, which doubtless provided the nutrient stimulus for the high phytoplankton biomass. Similarly, the site on Wolf Creek nr Medicine Lake was just downstream from and connected to a major cattle watering hole in the stream, which regularly attracted large numbers of cattle to drink.

Finally, an examination was made of the relative contributions of floating & benthic algae *and* phytoplankton algae to total Chl *a*, on an area basis (Fig. 8.7). As was the case in Fig. 8.2, only samples that were 90-100% floating & benthic algae were selected to match with the phytoplankton data. Mean stream depth (from EMAP habitat data) was used to integrate phytoplankton concentrations to units common with the floating & benthic algae (mg/m^2; see explanation in the figure). The two plant communities, measured as Chl *a* on an area basis, could then be summed to provide total algal Chl *a* at each site.

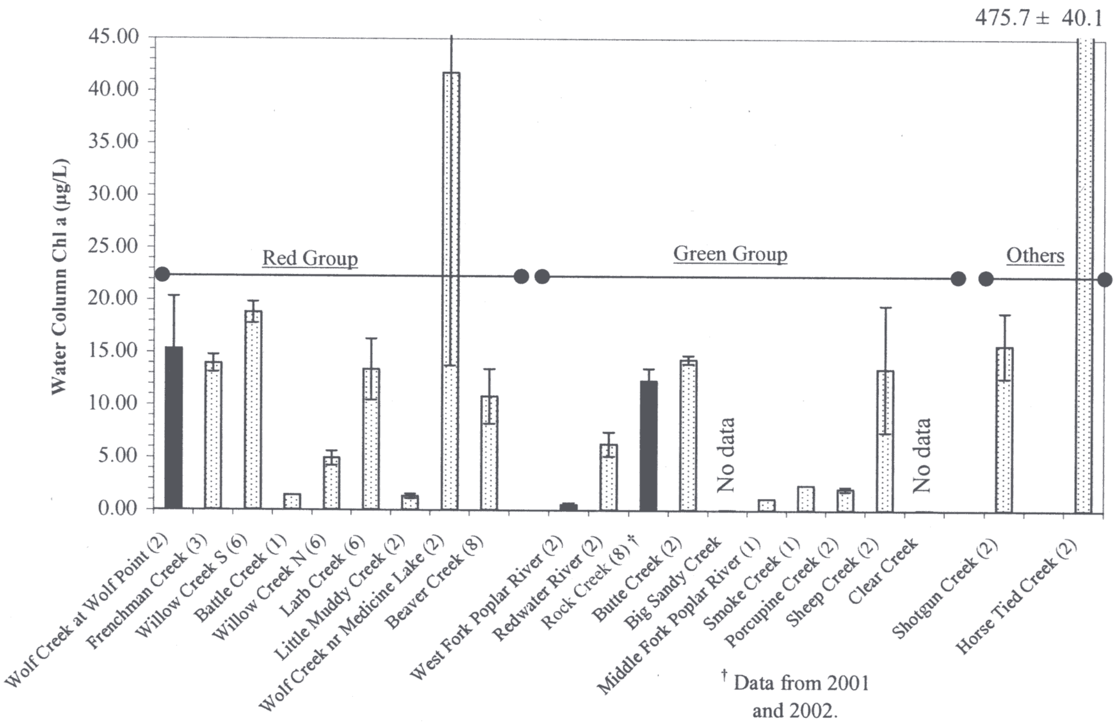


Figure 8.6. Summertime water-column Chl *a* concentrations in the study sites. Error bars are one standard error of the mean. Number of collected samples is shown in parenthesis following each site name. The comparison sites are shown in black.

One of the more interesting features of Fig. 8.7 is the relative contribution of phytoplankton to total stream Chl *a* in Red group vs. Green group streams. Larb, Beaver, Willow N and Willow S creeks are all incised stream channels (Rosgen F or G stream types) with fine-sediment bottoms. Conditions in these streams appear to discourage benthic plant development and as a result, the phytoplankton represent a major proportion (in most cases over half) of the streams' total algal Chl *a* density. These data support the notion that when unstable stream bottom conditions discourage benthic plant development, resources (e.g. N and P, among other things) are simply utilized by a different community of aquatic plants (the phytoplankton). Others have observed competition between benthic plants and phytoplankton with ensuing shifts in the dominant plant community (Phillips et al. 1978; Coffaro and Bocci 1997; Nelson and Lee 2001).

In summary, the streams of the Red group had lower total aquatic plant biomass, lower density of floating & benthic algae, fewer sites with macrophytes, and higher relative proportions of phytoplankton relative to streams of the Green group. Big Sandy Creek (of the Green group) had 68% of its stream bottom covered by filamentous algae and was a clear outlier from the rest of the study sites, where coverage was on average less than 30%. Wolf Creek nr Medicine Lake and Horse Tied Creek each had average phytoplankton Chl *a* concentrations (41.8 and 515.8 µg/L, respectively) far higher than the other streams in the study, where Chl *a* concentrations ranged from 0.4 to 27.2 µg/L. In the comparison sites the mean floating & benthic algal Chl *a* density was 43.6 mg/m^2, the median was 19.7 mg/m^2, and ninety percent of the samples fell between 6 mg/m^2 and 106 mg/m^2.

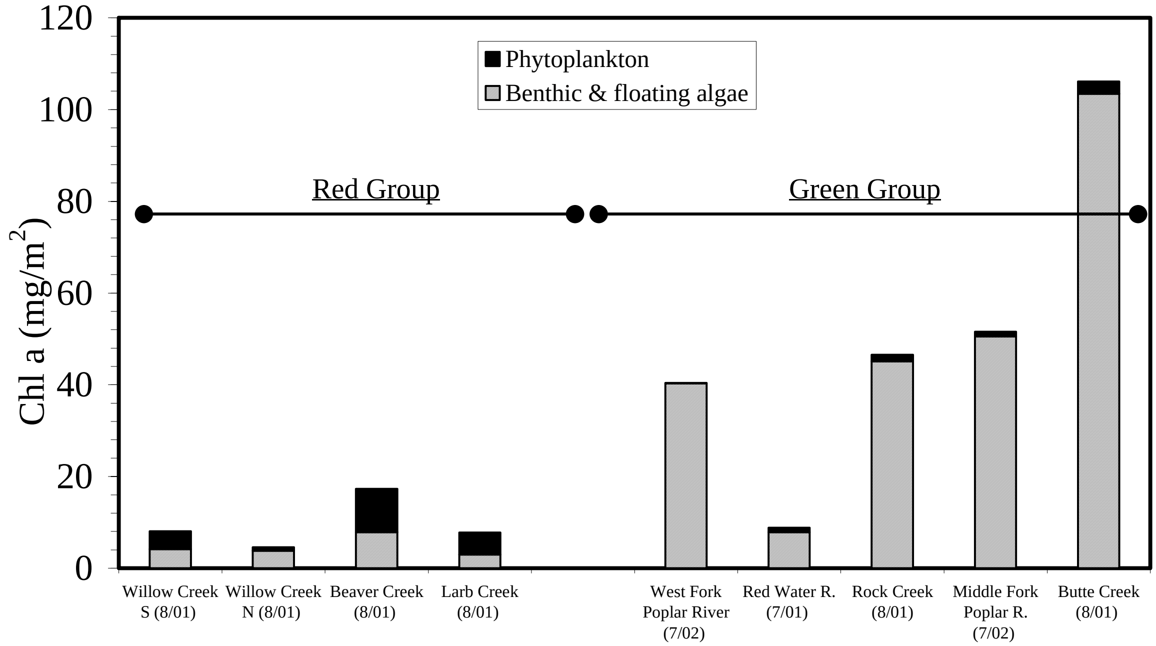


Figure 8.7. Chl *a* contribution from benthic & floating algae and from phytoplankton in selected sites. Areal phytoplankton Chl *a* for each site was calculated as the product of the mean water column Chl *a* concentration (mg/m^3) and the mean water-column depth (m) of the site's transects. Dry transects were not included. Only sites are shown where water depth, floating & benthic Chl *a*, and phytoplankton Chl *a* were all collected on the same day (sampling dates shown following site names).

8.1.2 Analysis of Variables that Affect Aquatic Plant Biomass

In this sub-Section variables will be examined that were likely to have had some influence on algae and other aquatic plants in these streams. Rather then take a “data dredging” approach, in which many potential causal variables are examined but as a result there is a greatly increased likelihood of concluding that insignificant relationships are significant, variables have been selected that have been shown in other studies to be important in regulating aquatic plant life (see Wetzel 1975; see Stevenson et al. 1996). These variables, along with the rationales for their selection, are shown in Table 8.3.

Table 8.3. Selected variables that were likely to affect benthic algal and aquatic plant biomass.

Independent Variable	Data Form	How Measured	Reason Selected
Physical Variables			
Light availability	Quantitative	Bank edge and midchannel canopy densiometer measurements, per Western EMAP protocols.	Light is a major factor controlling plant growth.
Stream sediment D_{50}	Quantitative	100-count of sediment particles within stream reach, per Western EMAP protocols.	Sediment substrate influences rooting capability and can influence benthic aquatic plant community structure. D_{50} is the mean particle size of the reach.
TSS and turbidity	Quantitative	Labaratory measument of water samples.	Direct measure of suspended sediments in stream. Can impact light availability and is related to stream channel stability.
Rosgen entrenchment ratio	Quantitative	At representitive x-section within the stream reach using a laser level.	Quantitatively defines vertical containment of a stream or river. May act as a surrogate measure of the influence of scouring flows.
Flow status	Qualitative	Observation of flow status spring through summer 2001 & 2002.	Influences many physical and chemical characteristics of streams.
Chemical Variables			
Electrical conductivity (EC)	Quantitative	Combination of field and laboratory measurements of water samples.	Different algal and aquatic plant species have different sensitivities to salinity, which can be measured as EC.
Total N	Quantitative	Labaratory measument of water samples.	Streams were shown to be N limited.
Nitrate + Nitrite	Quantitative	Labaratory measument of water samples.	Streams were shown to be N limited.

Note that among the nutrient variables in Table 8.3, P was not analyzed. The stream sites have already been thoroughly demonstrated to be N limited in Section 7.0, therefore significant regressions between P and benthic plant biomass would surely be spurious.

A total of fifty correlations were examined using the Spearman rank test (Table 8.4). Data from all streams sites were used to run the tests. Among the variables tested, channel D_{50}, TSS, turbidity, entrenchment ratio, flow, and $NO_{2/3}$ concentration produced significant (p-value $\leq$ 0.05) relationships.

The variables that produced the greatest number of significant relationships were TSS and turbidity. The patterns of the correlations for these variables (positive or negative) relative to the plant communities in question are very telling. In all cases, TSS and turbidity were negatively correlated with forms of aquatic plant life (macrophytes, benthic algae) associated with the stream bottom (Table 8.4; Fig 8.8). Rosgen entrenchment ratio was positively correlated to both total aquatic plant biomass and floating & benthic algal density, indicating that more incised channels had less bottom-associated plant biomass than un-incised channels. The significant correlations between TSS & turbidity and stream water Chl *a* may not be meaningful, because phytoplankton are part of the TSS and turbidity measurements and so these may simply be autocorrelations.

The correlations generally support the idea that unstable stream bottoms associated with certain kinds of channel morphology and flow patterns discourage benthic plant development, and therefore available resources may shift to and are used by phytoplankton. Both high TSS and high turbidity are found in streams that have more erosion and so TSS and turbidity are probably acting as surrogates for stream channel stability. As the entrenchment ratio of streams decreases (the channels are more incised), high flows are increasingly contained within the channel due to reduced floodplain access and in the case of F and G channels this can result in intense channel scouring, as their floodplains are practically non existent (Rosgen 1996). So, streams that experience more intense scouring flows (like Red group streams) tended in this study to have reduced benthic plant populations and increased phytoplankton populations. Others have shown that high flows are an important factor that control benthic aquatic plant communities (Biggs and Close 1989; Biggs 1995; Peterson 1996).

Table 8.4. Spearman rank correlations & probabilities between aquatic plant variables and potential controlling variables. The Spearman Rank correlation for each pair is shown (plus or minus indicates if correlation was positive or negative) followed by the probability in parenthesis. Significant correlations (probability ≤ 0.05) are shown in bold.

	Dependent Variable				
Independent Variable	Mean summer total aquatic plant biomass (mg Chl a/m^2)	Mean summer algal Chl *a* (floating & benthic) (mg/m^2)	Mean summer phytoplankton (µg Chl *a* /L)	Stream bed cover by macrophytes (%)	Stream bed cover by filamentous algae (floating & benthic) (%)
Physical Variables					
Mean bank canopy density (%)	-0.02 (0.924)	+0.31 (0.240)	+0.02 (0.939)	-0.03 (0.900)	-0.29 (0.211)
Mean mid-channel canopy density (%)	+0.02 (0.920)	+0.31 (0.242)	+0.19 (0.463)	+0.10 (0.671)	-0.15 (0.524)
D_{50} of stream substrate (mm)	+0.01 (0.956)	+0.11 (0.698)	**-0.51 (0.031)**	-0.18 (0.429)	+0.23 (0.323)
Mean spring and summer TSS (mg/L)	**-0.56 (0.008)**	-0.44 (0.080)	**+0.49 (0.037)**	**-0.62 (0.002)**	**-0.59 (0.005)**
Mean spring and summer turbidity (NTU)	**-0.49 (0.026)**	-0.41 (0.112)	**+0.50 (0.034)**	**-0.63 (0.002)**	**-0.72 (0.002)**
Rosgen entrenchment ratio	**+0.50 (0.024)**	**+0.52 (0.039)**	-0.38 (0.135)	+0.30 (0.195)	+0.23 (0.339)
Flow status (perennial or intermittent)†	+0.42 (0.060)	+0.37 (0.164)	**-0.49 (0.037)**	+0.22 (0.344)	-0.12 (0.612)
Chemical Variables					
Mean spring & summer EC (µS/cm @ 25^o C)	+0.13 (0.574)	+0.00 (0.989)	+0.21 (0.428)	+0.38 (0.095)	+0.09 (0.711)
Median spring and summer total N (µg/L)	+0.06 (0.793)	+0.27 (0.322)	+0.42 (0.084)	+0.11 (0.627)	-0.21 (0.354)
Median spring and summer $NO_{2/3}$ (µg/L)	+0.25 (0.267)	**+0.50 (0.048)**	+0.21 (0.411)	+0.05 (0.848)	-0.25 (0.283)

† Qualitative variable. These were converted to dummy variables for correlation with the dependent variables.

In Table 8.4 phytoplankton Chl *a* is negatively correlated to flow (meaning perennial streams had less phytoplankton then intermittent streams) and this may be related to competition from other aquatic plants. All of the perennial streams (e.g. WF Poplar, MF Poplar, and Redwater rivers) had growth of macrophytes, whereas a number of the intermittent streams did not (Table 5.1; Fig. 8.4). Phytoplankton blooms were often observed in the remaining pools of intermittent streams when other aquatic plants were not present.

One of the nutrient variables ($NO_{2/3}$) was positively correlated to mean summer floating & benthic algal density (Table 8.4; Fig. 8.9), which supports the finding that N was the limiting nutrient to algae as discussed in Section 7.0. Nitrate (NO_3) is a dissolved inorganic form of N that is readily taken up by algae (Wetzel 1975), and therefore should be more closely correlated to algal biomass than total N. Case in point, Big Sandy Creek's unusually high filamentous streambed cover (68%; Fig. 8.5) is most likely the result of N-limited algae utilizing a rich nitrate source. Saline seeps (which frequently develop where dryland crop-fallow practices dominate) are a problem in Big Sandy Creek's basin, and nitrate is often highly elevated in saline seeps, sometime to greater than 1 g/L (Brown et al. 1982; Nimick and Thamke 1998; SD DENR 2004).

It is not surprising that there were only insignificant correlations between $NO_{2/3}$ and the aquatic plant variables 'total aquatic plant biomass' and 'streambed cover by macrophtyes', since both of these aquatic plant variables measure macrophyte biomass in some manner. Macrophytes can take up nutrients with their roots and so are not wholly dependent upon water column nutrients (Barko and Smart 1986), and are frequently found to be poorly correlated to N and P concentrations (Carr and Chambers 1998). It is somewhat more surprising that significant correlations were not found between $NO_{2/3}$ and phytoplankton or filamentous algae streambed cover.

Variations in EC have been shown to affect periphyton biomass in plains streams (Giorgi et al. 1998) and algal & macrophyte species composition in arid ephemeral streams (Moreno et al. 2001), however it was evidently not an important causal variable controlling total aquatic plant biomass in these streams.

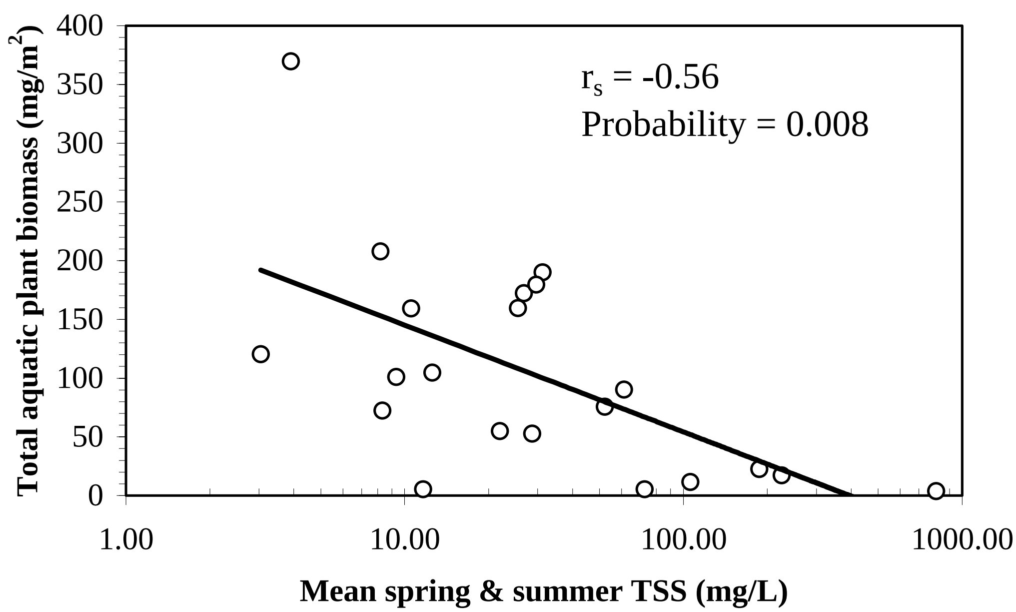


Figure 8.8. Relationship between mean spring & summer TSS concentration and mean summer total aquatic plant biomass. Line shown is the best-fit least-squares regression line.

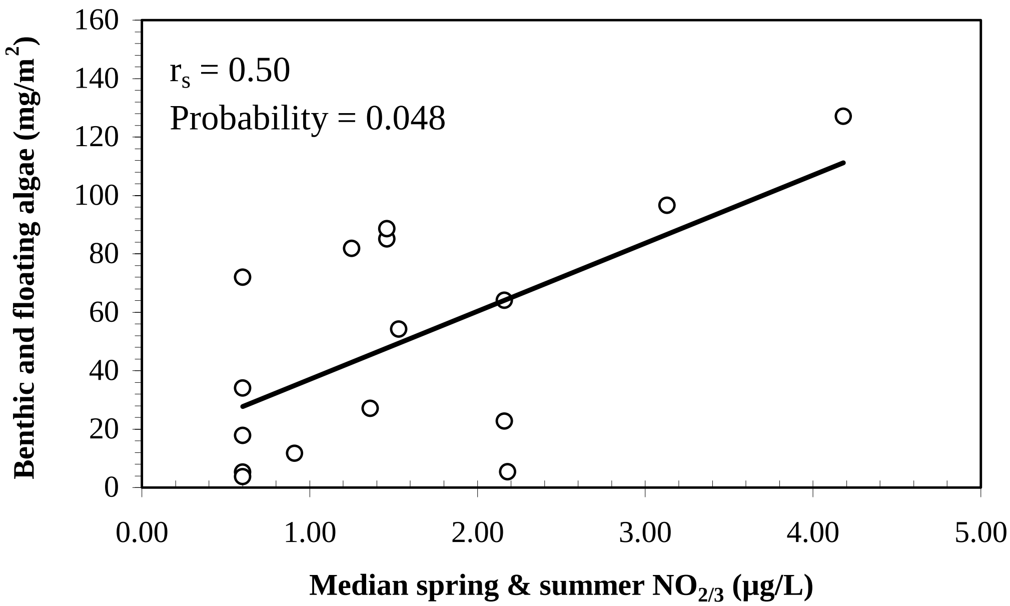


Figure 8.9. Relationship between mean summer floating & benthic algae and median spring and summer NO2/3 concentration. Line shown is the best-fit least-squares regression line.

In summary, aquatic plant communities in these streams are probably most influenced by the degree of vertical channel incisement and the resulting return frequency of scouring high flows. Entrenched streams like those of the Red group can have severe scouring events during high flows, and Red group streams were found to have lower total aquatic plant biomass, lower density of floating & benthic algae, fewer sites with macrophytes, and higher relative proportions of phytoplankton compared to the slightly- to moderately-entrenched Green group streams. Supporting this scenario were significant negative correlations between TSS & total aquatic plant biomass, TSS & macrophyte streambed cover, and TSS & filamentous algae streambed cover (i.e., streams with higher TSS concentrations had lower floating & benthic plant biomass). Floating & benthic algal biomass was positively correlated to spring and summer $NO_{2/3}$ concentrations, a finding that supports those of Section 7.0 that showed that N (rather than P) was the most limiting nutrient for algae. The latter correlation also indicates that N was an important regulator of algal density (along with TSS/entrenchment) in these streams.

8.2 Qualitative Measurements

8.2.1. Summary of Diatom and Soft-Bodied Algae Analyses

In both 2001 and 2002, qualitatively collected benthic samples of soft-bodied and diatom algae were provided to *Hannaea* for analysis. These samples were provided blind, in that no indication of the sites' conditions was provided nor were any other supporting data (nutrient concentrations, water quality, etc.) provided. Complete results for both years of the study can be found in Bahls (2002 and 2003), and only the most important findings will be summarized here; therefore, the reader is referred to the original reports for greater detail.

Six diatom metrics were used to evaluate streams in both years of the study. These were: % Nitrogen Autotrophs; % Eutrophic Diatoms; % Epithemiaceae; % Nitrogen Heterotrophs; Polysaprobous Diatoms; and the Pollution Index. These metrics were selected because each of them measures the diatom community response to either inorganic or organic nutrients (primarily N and P). The overall indication from these metrics was that the streams were N-limited, which is in agreement with the findings of Section 7.0 of the present study.

The reports also concluded that among the three general methods used to collect benthic samples (hoop, core, and template), the sediment cores (also referred to as "mud plugs" in the reports) were the best for assessing nutrient availability in prairie streams. This was because the sediment core samples represented depositional habitats that accumulated diatom taxa from all of the other habitat types found in the streams. Template and hoop samples, on the other hand, represented habitats where intense competition for nutrients was occurring and this likely skewed the metric values calculated from these sites.

Non-diatom algae were also identified. Among the green algae, *Cladophora*, *Spirogyra* and *Rhizoclonium* were the three most common. These three and a few other filamentous algae species comprised the bulk of the streams' overall algal biovolume, and were by far the greatest contributors of algal biovolume in the hoop samples. This finding is in agreement with the findings of a long-term study of a northern prairie stream in North Dakota, in which *Cladophora* and *Spirogyra* were found to be the most prominent macroalgal species (Neel 1985). Neel (1985) also notes that *Spirogyra* thrived in quiescent pools, a fact which was also observed in the present study in Clear Creek.

Algae of the Cyanophyta Division (cyanobacteria, also called blue-green algae) were also identified, and one of the most interesting findings was that more than 50% of the most abundant cyanobacteria in the study streams had heterocysts. Heterocysts are specialized cell features formed uniquely by this algae group that fix atmospheric N into a form (ammonia) that can be used by the cells (Bold and Wynne 1985), and are formed when inorganic N in the water is in short supply. The large proportion of cyanobacteria possessing heterocysts is further evidence supporting the case for N-limitation in these streams.

The diatom metrics generated a considerable spread of scores among the sites, which indicated significant differences in nutrient availability among the streams. This spread of scores was very useful in the reference site selection and ranking process undertaken in Section 3.0. Others have also found diatom metrics to be useful in segregating streams along a disturbance gradient (Hill et al. 2000). In seeming contrast, a recent study that evaluated a diatom index of biotic integrity (IBI) found that in prairie streams diatoms were not a particularly good predictor of conditions along a human disturbance gradient (Bramblett et al. 2003). However, that study's goals differed considerably from those of the present study. The present study was aimed mainly at examining nutrient impacts, and therefore the metrics selected were picked because they best addressed nutrient availability. On the other hand, the Bramblett et al. (2003) study selected a number of diatom metrics that were intended to create a metric battery. Each metric in the battery was thought to be responsive to different stressors – nutrients included – among an array of different human disturbances. There is only one diatom metric (% Nitrogen Heterotrophs) in the final Bramblett battery that is common to the present work, although there are four other metrics (% Nitrogen Autotrophs, % Eutrophic Diatoms, Polysaprobous Diatoms, and the Pollution Index) used in the present study that were eliminated in the Bramblett study because they were thought to be redundant. And although the diatom IBI did not work very well overall, their study states that a number of *individual* diatom metrics worked quite well. In fact, the % Nitrogen Heterotrophs metric showed a positive correlation to instream $NO_{2/3}$ concentrations (Bramblett et al. 2003).

In summary, the qualitative algae data in the two reports (Bahls 2002 and 2003) were a very valuable contribution to the study. The "nutrient loading based on diatoms" rankings (Table 3.2), derived from the 6 diatom metrics mentioned above, were a key component used to select the final comparison sites. Further, these data independently confirmed that N-limitation was quite universal in these streams but more importantly, confirmed this from a completely different analytical angle. The diatom-metric and soft-bodied algae data that indicated N- limitation in the streams were based on known life history characteristics of the algae groups, whereas N-limitation was identified in Section 7.0 of this study using a completely different, quantitative approach. The convergence of results from two independent approaches is strong evidence that the conclusion is correct.

8.2.2 Macrophytes in the Study Streams

Although not a primary objective of the study, macrophytes were identified in a number of streams in 2002, and also during subsequent samplings in 2003 that were conducted as part of a separate study. Identification was undertaken by staff of the University of Montana, or by myself using Muenscher (1944). In all, macrophytes were collected and identified in seven of the stream sites (Table 8.5). Three of the streams in the study had no macrophytes of any kind (Willow N, Larb, and Willow S creeks). Clear Creek had a flora uncommon to any of the other stream sites.

Macrophytes were intensively studied and sampled in the Poplar River basin in the early 1980's in response to water quality concerns on the EF Poplar River associated with a coal burning power plant that was being developed in Canada (Klarich 1982). One macrophyte species, *Myriophyllum spicatum* L. var. *exalbescens* (also referred to simply as *Myriophyllum exalbescens*; Muenscher 1944), was found to be supra-dominant throughout the north-central and north-eastern areas of the Poplar River drainage. It was found to prefer lower current-velocity areas whereas *Potamogeton pectinatus* preferred areas where faster water currents predominated (Klarich 1982). These habitat preferences may in part explain why *Potamogeton pectinatus* was found in the perennially running Middle and West Forks of the Poplar River, whereas it was completely absent from the intermittent Wolf Creek at Wolf Point.

Table 8.5. Macrophytes identified in selected study streams.

Stream Site Name	Site Condition Rating	Macrophytes Species Identified	Common Names	Field Notes
Wolf Creek at Wolf Point†	Min. impacted	*Myriophyllum exalbescens; Zannichellia sp.* ‡	Watermilfoil; Horned pondweed	Growth heavy, in pools
West Fork Poplar River*	Min. impacted	*Myriophyllum exalbescens; Hippuris vulgaris; Potamogeton pectinatus*	Watermilfoil; Mare's tail; Sago pondweed	Growth sparse to moderate
Clear Creek†	Reference	*Ranunculus aquatilus; Veronica americana*	Water buttercup; American brooklime	Fairly heavy growth
Rock Creek †	Min. impacted	*Potamogeton pectinatus*	Sago pondweed	Growth sparse
Battle Creek†	Intermediate	*Myriophyllum exalbescens*	Watermilfoil	Growth heavy
Middle Fork Poplar River	Intermediate	*Myriophyllum sp; Potamogeton pectinatus; Potamogeton richerdsonii*	Watermilfoil; Sago pondweed; pondweed	Growth fairly heavy, in glides and pool margins.
Shotgun Creek	Most impacted	*Sagittaria latifolia*	Duck potato	Abundant emergent macrophyte in stream

*Samples collected in 2002 and 2003.

† Samples collected in 2003.

‡ Difficult to ID, plant may also have been *Najas sp*. or *Ruppia sp*. However an earlier, very intensive study of macrophytes in this basin found only *Zannichellia palustrus* (Klarich 1982).

SECTION 9.0
AQUATIC MACROINVERTERBRATES

Aquatic macroinvertebrate samples were collected once at each site during each of the two field seasons. Samples were collected in late June in 2001 and in late July in 2002. All samples were submitted to *Rhithron* Associates, Inc. for identification and calculation of biometrics, from which three reports were generated (Bollman 2002a; Bollman 2002b; Bollman 2003). Nonparametric multidimensional scaling (a graphical ordination process) was used to examine relationships between different macroinvertebrate assemblages. Only the most important findings from the reports, particularly those from the addendum to the 2002b report (Bollman 2003), will be discussed below and interested readers are referred to the original reports for greater detail. Due to time constraints the aquatic macroinvertebrates communities were not assessed in relation to the Red and Green groups developed in Section 6.0; however, work is ongoing in this area (including additional sampling in summer 2004) and will be reported on at a later date.

The reports indicate that the timing of sample collection may have a major effect on the species composition of samples, and in turn on conclusions about water quality. The taxonomic composition of the late-June 2001 samples was quite distinct from the late-July 2002 samples. Because in each year of the study sampling mainly took place in one or the other of two sub-major basins (the Milk in 2001 and the Lower Missouri in 2002), without closer examination of the data one might have concluded that the taxonomic differences were related to the difference in geography. However, three sites located in the Milk sub-major basin that were sampled in both years of the study had taxonomic composition in 2002 that looked like those of Lower Missouri sites samples in 2002. (Similarly, their 2001 assemblages looked much like assemblages from the other 2001 sites.) This implied that differences between the two years of the study were related to seasonal differences rather then environmental differences between the two basins. Evidently the four-week difference between sampling events of 2001 vs. 2002 played a huge role in shaping the macroinvertebrate populations that were found.

Nevertheless, there were a few geographically distinct characteristics of the macroinvertebrate communities. For example Butte Creek, the Middle Fork Poplar River, and the West Fork Poplar River had similar aquatic macroinvertebrate populations that differed distinctly from the other streams in the Lower Missouri sub-major basin. Samples from the Milk sub-major basin contained tubificid worms in 95% of the sites, whereas streams of the Lower Missouri basin only had these organisms in 25% of the sites. Tubificid worms thrive in silt-bottomed streams and this condition is probably related to geomorphic characteristics of the Milk basin, where more clayey, low permability soils predominate (see Section 4.0).

The two types of macroinvertebrate sampling procedures used in this study were the reach-wide composite and the targeted-riffle composite. Many of the organisms that were commonly found in the reach-wide composite samples were tolerant species, those organisms that inhabit and thrive in a wide range of habitat and water quality conditions. Reach-wide samples were collected from multiple habitat types that included riffles, pools, glides, macrophyte beds, etc. Some of these habitat types were such that only tolerant species could live there, due to conditions such as heavy siltation, anoxic sediments, etc.

There were clear differences between the aquatic macroinvertebrate assemblages collected in reach-wide vs. targeted-riffle samples. The targeted-riffle samples were much more likely then the reach-wide samples to be dominated by a single taxa. From 40-70% of the targeted riffle samples were dominated by a single taxa, while only 9% of the reach-wide samples were. Species richness (an important component of biometrics as it measures community diversity) was much higher in the reach-wide samples when compared to the targeted-riffle samples.

A study evaluating two macroinvertebrate indexes of biotic integrity (IBI's) for prairie streams of Montana indicates that riffle samples, rather then pool-only samples, are better predictors of conditions along a human-disturbance gradient (Bramblett et al. 2003). The study also states that the riffle macroinvertebrate IBI was most strongly driven by its response to stream substrate (e.g., sedimentation). Bollman (2003) also concludes that targeted riffle samples could address sedimentation issues, whereas reach-wide composite samples could not. However both studies note that most Montana prairie streams do not have riffles, and therefore the riffle IBI's scope of usefulness is quite limited; both studies go on to state that further work should be undertaken to improve the macroinvertebrate indicators for pool or reach-wide composite samples.

Bollman (2003) concludes that reach-wide composite samples, appropriately timed, could yield a great deal of information about stream habitat and water quality. Corixids (water boatman) were a dominant taxa in some streams in 2001, while odonates (dragonfly larvae) were a dominant taxa in 2002, probably as a result of sample collection timing. Both taxa have a diversity of species in this region and both taxa have been used in the past to gauge water quality. Odonate species have been shown to have associations with salinity and physical habitat structure, while corixids have certain thermal and water quality preferences. One caveat to the use of reach-wide composite macroinvertebrate samples is that they do not address fine sediment deposition as a potential impact. But this is where targeted riffle samples work well, and a sampling protocol specific to prairie streams could certainly be designed to include both sampling methods.

If sampling events could be timed to capture late-stage odonate nymphs and adult corixids, it may be possible to further develop relationships between macroinvertebrates and both water quality and habitat parameters. As stated by Bollman (2003) "the metric battery recommended by MT DEQ does not return consistent results for targeted riffles and reach-wide composites…" and "Many of the metrics included in the battery seem to have a lack of relevance to the type of waterbodies characteristic of these regions."

The take home message from these results is that, as assessment tools in Montana prairie streams, macroinvertebrates metrics need more development before they can be relied upon consistently to assess water quality and habitat.

SECTION 10.0
ALGAE AND NUTRIENT CRITERIA DEVELOPMENT

10.1 Comparison of Study Results to EPA's Criteria Development Guidance

According to guidance provided by the EPA, it is generally expected that nutrient concentrations in reference streams will have a distribution with a central tendency around a lower value than the population of streams as a whole (USEPA 2000a). This is demonstrated in Fig. 10.1 (from page 96, USEPA 2000a). EPA considers reference "relatively undisturbed stream segments that can serve as examples of the natural biological integrity of a region" (USEPA 2000a). One approach to determining appropriate nutrient concentrations suggested by EPA is as follows. If the value at the 75th percentile of the reference-sites distribution (histogram) has a close overlap with the value at the 25th percentile of the whole population's histogram, then the appropriate criteria for the nutrient in question will be a concentration falling between the 75th percentile of the reference sites and the 25th percentile of the whole population (Fig. 10.1). This approach assumes that there are sites within the whole population that are polluted with excess nutrients, and therefore by selecting a nutrient value towards the lower end of the whole-population concentration histogram, water quality and beneficial uses will be protected. In the example shown in Fig. 10.1 the appropriate nutrient value selected for this group of fictitious streams would be 5.5 µg/L.

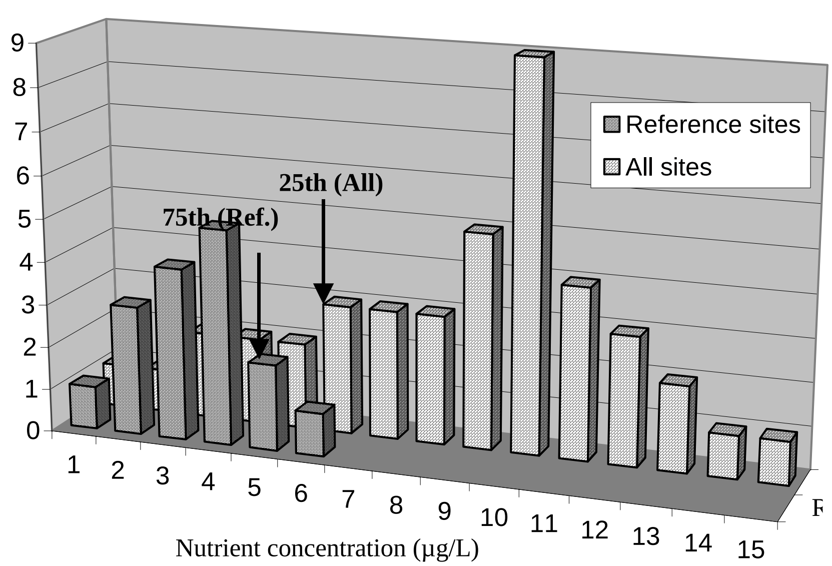
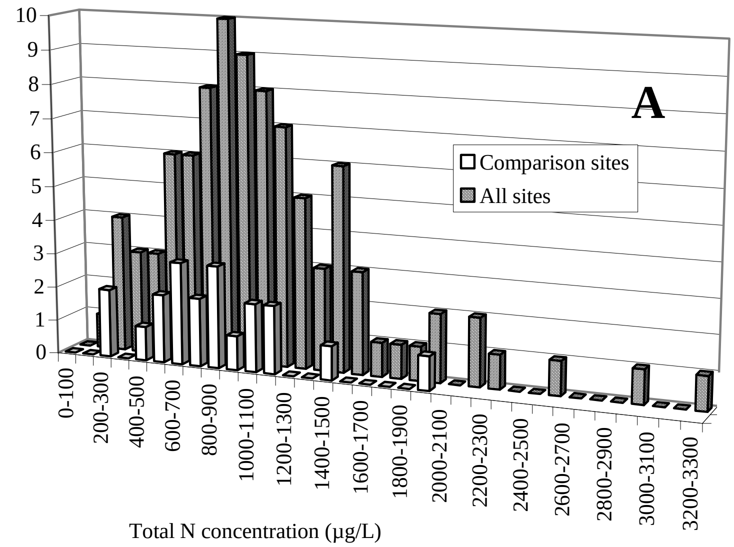


Figure 10.1. Idealized relationship between nutrient concentrations in reference sites and in all sites of a population. Redrawn from USEPA (2000a). In this example, the nutrient value falling between the 75th percentile of the reference sites and the 25th percentile of all the sites is approximately 5.5 µg/L, and would be considered an appropriate criteria for this group of streams (USEPA 2000a).

Data for total N and total P from this study were plotted as histograms in the same manner as that shown in Fig. 10.1, and are shown in Fig. 10.2A&B, respectively. The histograms clearly show a number of characteristics of these data. Both the total N and total P concentration histograms for the complete set of sites are skewed to the right, as are the histograms of the comparison sites. This is especially true for total P. The total P histogram for the comparison sites and the all-sites total P histogram do not show a close overlap at their 75th and 25th percentiles (Fig. 10.2B); on the contrary, their distributions are similar. Separation of the comparison sites total-N histogram from that of the all-sites histogram is slightly greater than that observed for total P (Fig. 10.2A). Given that the nutrient distributions in Fig. 10.2A&B do not differentiate the comparison population from the whole population, application of the EPA approach cited above would not be appropriate. But regardless of the distribution of reference sites relative to the population as a whole, what it really comes down to is one's confidence in the quality of the reference sites. If one's confidence is high that the sites are truly in a "reference" condition, than it should not matter how their histogram compares to that of the overall population of sites. This issue will be examined further in the next sub-Section (10.2).



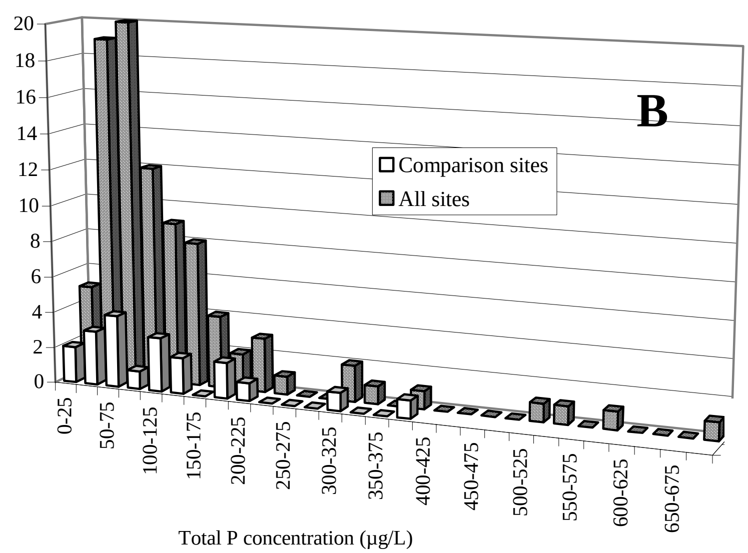


Figure 10.2. A. Total N concentration histograms for the comparison sites, and for all sites in the study. One outlier (Horse Tied Cr; 5,575.7 µg/L) in the all-sites histogram not shown in order to improve resolution of the figure. B. Total P concentration histograms for the comparison sites and for all sites in the study. Two order-of-magnitude outliers (Willow {S} and Larb creeks, 2,791.1 and 1,860.5 µg/L, respectively) not shown to improve figure resolution.

10.2 Some Thoughts on the Quality of Reference Sites, Comparison Sites, and Percentile Distributions

At this writing MT DEQ is developing a guidance document on how to interpret its narrative water quality standards. The draft guidance document defines "reference" as those sites that are essentially in an undisturbed, natural condition. "Minimally impacted" sites are those found where the activities of man have made small changes to the completeness of the structure and function of the biotic community and the associated physical, chemical, and riparian habitat conditions, but all numeric water quality standards are met and all beneficial uses are fully supported. Both reference and minimally impacted sites can be used to describe reasonably attainable conditions and therefore can be used as comparison sites.

It appears that MT DEQ's reference and minimally impacted definitions both fall within EPA's more general definition of reference ("relatively undisturbed stream segments that can serve as examples of the natural biological integrity of a region"; USEPA 2000a). But to avoid confusion and to adhere to the draft definitions MT DEQ is working under, the term "reference" has been and will continue to be used in this report only in conjunction with nearly-pristine sites. In this context the term "comparison sites" is more appropriate to use, as is can incorporate both reference *and* minimally impacted sites. In using MT DEQ's draft definitions, only one site in the present study is considered reference (Clear Creek) while the rest of the comparison sites (Wolf Cr at Wolf Point, WF Poplar River, and Rock Creek) are rated as minimally impacted.

If one were very confident that a group of comparison sites were truly in a reference (i.e., nearly pristine) condition, then a nutrient criteria based on such a group could reasonably be set to the higher side of the groups' nutrient-concentration distribution than if one were less confident in the overall condition of the sites. The choice of which percentile to use could be based on a "confidence in the quality of the sites" rating, as demonstrated in Fig. 10.3. In this context "quality" is meant to mean how close the sites' conditions comes to reference, and how applicable the sites are. (One could have a group of reference sites from high mountain streams but obviously they would not be very applicable to prairie streams.) Depending upon how confident one is in the quality of the comparison sites, be it either very high, good, fair, or low, the criteria selected would correspond to either the 90th, 75th, 50th or 25th percentile of the sites' nutrient concentration distribution. Obviously a criteria set at the 25th percentile indicates little confidence in the comparison sites and probably should be used as the last resort. The approach is protective because as one's confidence in the quality of the comparison sites diminishes, lower and lower (more stringent) nutrient concentrations are selected. At the "very high" confidence level, meaning of course that one is confident that the sites are in a reference condition, a nutrient concentration at the 90th percentile (or perhaps 95th) could be selected and so only rare outliers of the comparison sites would exceed a criterion set at that level (Fig. 10.3).

This approach can be applied to the present study. As shown in Table 3.2 of Section 3.0, only one of the comparison sites in the present study is considered reference and the rest of them are rated as minimally impacted. Given that a quarter of the data is from reference and three quarters is from minimally impacted sites, a rating of no better than "good" for the group as a whole is probably justified. Based on the concept shown in Fig. 10.3, "good" equates to the 75th percentile and so nutrient concentrations at the 75th percentile of the comparison-sites distribution would be appropriate preliminary criteria. After better regional reference sites have been located and more data collected, confidence in the quality of the comparison sites will probably go up and therefore the percentile value selected could correspondingly increase.

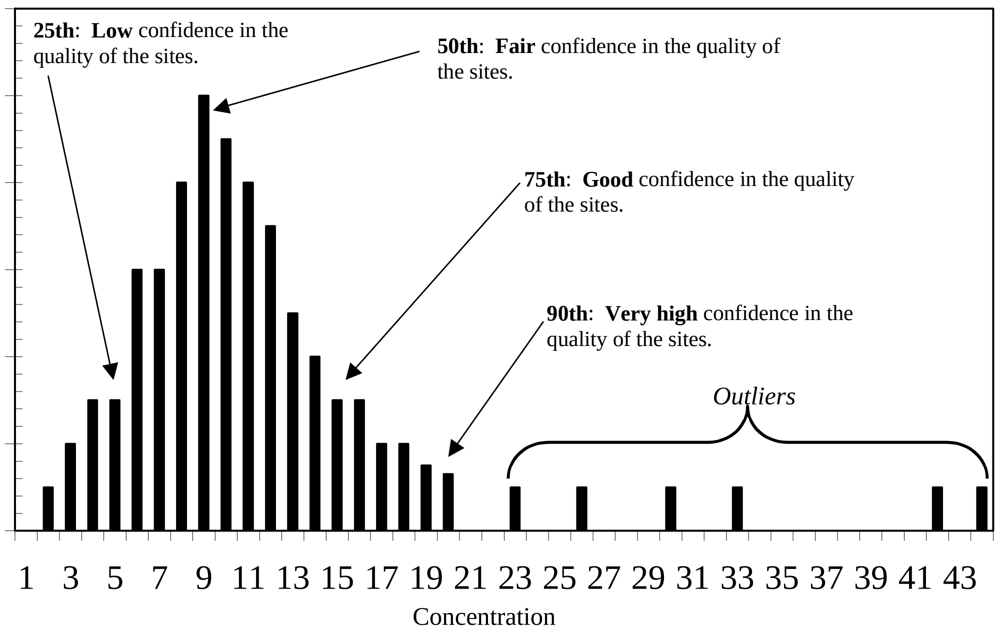


Figure 10.3. Conceptual approach to selecting appropriate nutrient criteria from a group of comparison sites based upon on the overall confidence in the quality of the sites. The histogram shown is for that of a hypothetical nutrient concentration distribution in a group of comparison sites.

10.3 Approach Taken to Develop Algae and Nutrient Criteria for Hi-line Wadeable Streams

There are certainly good rationales for establishing nutrient limits in this region. It was clearly demonstrated in Section 7.0 that algae are N limited in these streams, and it was further shown in sub-Section 8.1.2 that $NO_{2/3}$ was significantly correlated to floating & benthic algae biomass. But what approach to take? The approach ultimately taken and described below was a combination of methods that:

1. Determined what might constitute a nuisance algae level for this region considering the approach described in 10.2, and then comparing those results to literature values for nuisance algae.

2. Used the present study's $NO_{2/3}$-algae biomass relationship to establish a threshold nutrient limit, and

3. Extended the findings of the $NO_{2/3}$-algae biomass relationship to the other nutrients that were measured in the study.

Since the level of confidence in the comparison sites of the present study is good, the 75th percentile of the comparison sites' distribution is a reasonable point at which to set thresholds for nutrients and algae density. In the comparison sites, the 75th percentile for various algae measurements are as follows: streambed cover by filamentous algae, 25%; floating & benthic algae, 61 mg Chl *a*/m^2; and Chl *a* in stream water (phytoplankton), 14 µg/L. How these values compare to other regulatory or recommended limits in the literature are discussed below.

With the exception of Big Sandy Creek, average filamentous algae streambed cover did not exceed 30% in the study sites and is no higher than about 25% in the comparison sites (Fig. 8.5). Welch et al. (1988) suggests that around 20% streambed cover may represent a nuisance threshold, and the New Zealand Ministry for the Environment recommends no more than 30% cover by filamentous algae to protect recreational & aesthetic uses of its gravel-bottom streams (Biggs 2000). Given these considerations, an average streambed cover by filamentous algae of 25% appears to be reasonable for Hi-line wadeable streams during the growing season (May through September), and is the same as the value at the 75th percentile of the comparison sites. A single-sample maximum of 30% during the growing season is also recommended.

For floating & benthic algal biomass, the value at the 75th percentile of the comparison sites was 61 mg Chl *a*/m^2, and ninety-five percent of the comparison site samples were below 106 mg Chl *a*/m^2 (n=65). The highest average summer floating & benthic algae density in a comparison site was 85.0 mg Ch *a*/m^2 (Fig. 8.2). For the complete set of sites, ninety-five percent of the samples were below 143 mg Chl *a*/m^2 (n=243) and the highest average summer floating & benthic algae density at a given site was 127 mg Chl *a*/m^2 (Fig.8.2). Regulatory or recommended limits in the literature are very similar to these values. Average summer benthic Chl *a* is not to exceed 100 mg /m^2 in the Clark Fork River (ARM 17.30.631), and the New Zealand government recommends a maximum of 120 mg Chl *a*/m^2 filamentous algae during the summer (Biggs 2000). An EPA search of the literature found that about 150 mg Chl *a*/m^2 was a generally agreed upon maximum (USEPA 2000a). Given these facts, a growing season average (averaged from May through September) not to exceed about 65 mg Chl a/m^2 appears to be reasonable and should protect beneficial uses in Hi-line streams. 110 mg Chl *a*/m^2 appears to be a reasonable single-sample maximum, since 95% of the comparison sites were below this value. These algae densities should automatically protect other beneficial uses, as algae densities from 100-150 mg Chl *a*/m^2 are lower than levels thought to impact other uses such as aquatic life and fisheries (Biggs 2000).

Phytoplankton biomass was quantified in this study by measuring the Chl *a* concentration (in ug/L) of the stream water. Ninety-five percent of the comparison site samples did not exceed 20.0 ug/L. The mean summer Chl *a* in the comparison site with the highest overall stream water Chl *a* concentration (Wolf Cr at Wolf Point; Fig. 8.6) was 15.3 ug/L. And as mentioned above, the value at the 75th percentile of all the comparison sites is 14 ug/L. Ninety-five percent of the phytoplankton samples from all study sites were below 29.0 ug/L. Excess phytoplankton growth can also be a recreational & aesthetic nuisance, as was quite clear in the cases of Horse Tied Creek and Wolf Creek nr Medicine Lake. In wadeable streams of the Hi-line region, 15 ug/L is recommended as a growing season average (May through September) and 20 ug/L is suggested as a single-sample limit. It should be noted that 15-20 ug/L is much higher than the EPA's year-round recommendation for stream water Chl *a* in the region, which is 3.2 ug/L (USEPA 2001).

Non-algal benthic plants (macrophytes) were also measured in this study, and have been presented in terms of both their streambed area coverage (Fig. 8.4) and biomass (Fig. 8.1). However at this time there is too little information to determine if the observed densities are appropriate, or what biomass in these streams might constitute a "nuisance" level of macrophytes.

Nitrogen was found to limit algal growth in Hi-line streams and $NO_{2/3}$ in particular was significantly correlated to floating & benthic algal density. Using the least-squares regression equation of the line shown in Fig. 8.9 ($y = 23.302x + 13.741$), a maximum limit for floating & benthic algae of 110 mg Chl a/m^2 is equal to a $NO_{2/3}$ concentration of 4.1 µg/L. The value 4.1 µg $NO_{2/3}$ /L is at the 89th percentile of the comparison sites' cumulative frequency distribution, and at the 89th percentile of the entire dataset (Table 10.1). Because the level of confidence in this study's comparison sites is good but not high, it is prudent to set the nutrient criteria a little lower, down to the 75th percentile of the reference sites cumulative frequency distribution. This equates to a value of 2.3 µg $NO_{2/3}$ /L. No other nutrient in the present study was found to significantly correlate to floating & benthic algal biomass. However, is has been shown that nutrient limitation can shift (e.g., from N to P) because of changing environmental or seasonal conditions (Cotner et al. 2004), and so it is prudent to recommend limits on other nutrients. Given that the 75th percentile of the comparison sites is the appropriate percentile limit for $NO_{2/3}$, then it is reasonable to set the concentrations for the other nutrients at the same percentile. This approach results in limits on other nutrients as follows: total N 1,044 µg/L; total P 153 µg/L; TDP 49 µg/L; and SRP 8.0 µg/L (Table 10.1). These values would be most appropriately applied during the aquatic-plant growing season (May through September).

How do these nutrient values compare to EPA's recommended values? EPA has published recommended concentrations for nutrients in this region of the country (EPA 2001) and suggests that total N concentrations should be no higher than 960 µg/L to be in a "reference" condition. EPA does not have reference sites *per se*, therefore they calculate their values strictly as the 25th percentile of their whole-population data set. There is reasonable agreement between this study's recommendation (1,044 µg/L) and the EPA's suggestions (960 µg/L) for TN, however other recommended nutrient values are very different. EPA's recommends 60 µg/L for $NO_{2/3}$ and 41 µg/L for TP (year round) for this region; no recommendations are made for TDP or SRP. Others who have reviewed EPA's 2001 criteria relative to reference sites or undisturbed watersheds find that the EPA-recommended criteria are frequently too low (Ice and Binkley 2003), but in this case some EPA criteria may be too low (e.g., TP) while others, $NO_{2/3}$ in particular, would be too high especially during the summer aquatic-plant growing season.

Table 10.1. Cumulative frequency distribution for nutrients in the comparison sites, and for the entire study dataset.

Cumulative Percentile	Nutrient Concentration (ug/L)				
	$NO_{2/3}$	TDP	SRP	Total P	Total N
Comparison sites					
5th percentile:	0.6	13	1.7	24	247
10th percentile:	0.6	17	1.8	30	433
25th percentile:	0.6	22	2.7	49	596
50th percentile:	0.8	31	4.7	99	770
75th percentile:	2.3	49	8.1	153	1044
90th percentile	6.7	142	15	215	1210
95th percentile	34	225	21	326	1455
All sites					
5th percentile:	0.6	13	1.3	25	298
10th percentile:	0.6	15	1.6	29	463
25th percentile:	0.6	21	2.2	47	708
50th percentile:	0.8	30	6.6	83	979
75th percentile:	2.2	47	20	141	1345
90th percentile	5.2	131	62	285	1799
95th percentile	13	231	151	529	2202

SECTION 11.0
GENERAL DISCUSSION

11.1 Beneficial Uses

11.1.1 Which Beneficial Uses are Impacted by Excess Aquatic Plants?

Because water across the state naturally varies in quality, the state of Montana has grouped waterbodies into water-use classes (ARM 17.30.606 through 17.30.614). Montana's water use classification is largely based on salinity and temperature characteristics of major stream basins. Large-scale basins (e.g., the Yellowstone River drainage) each have an underlying classification that applies to all waterbodies in the basin, except for those waterbodies specifically named and classified otherwise. The water-use classifications specific to the study stream sites in the present study are shown in Table 11.1. For each of these classes there are specific uses of the water (beneficial uses) that the waterbodies within the class are expected to be maintained suitable for; these are detailed in ARM 17.30.620 through 17.30.629. Beneficial water uses include bathing, swimming and recreation, agriculture, non-salmonid fishes and associated aquatic life, and waterfowl. With the exception of the A-closed water class, state law treats all beneficial uses as equally important and waterbodies must be maintained suitable to support all of them. Associated with each water-use class are a number of water quality standards designed to protect the beneficial uses. In addition to the class-specific standards, Montana also has a number of general prohibitions (standards) that apply to all state waters, regardless of their class (see ARM 17.30.637).

It was demonstrated in Sections 7.0 and 8.0 that algae levels (e.g., percent filamentous stream bed cover) were affected by nutrients and that in some cases (Big Sandy Creek) unusually high levels developed. Further, it was shown that unusually high concentrations of stream-water Chl *a* (phytoplankton) were linked to a specific type of impact, that of cattle congregating in the stream bottom. Given that state waters have beneficial uses that must be protected, what specific beneficial use or uses does excess algae impact? The question can best be answered by reviewing some of MT's narrative water quality standards.

It is clear from Table 11.1 that the stream sites in this study belong to a number of different water-use classes. One beneficial use that they all have in common, however, is that they must be maintained suitable for swimming and recreation. Intimately linked to the recreation use is the concept of aesthetics. The linkage between recreation and aesthetics has already been established in Montana in the Clark Fork River. Benthic algal biomass ranging from 100-150 mg Chl a/m^2 has been found to be the threshold for an aesthetic nuisance (see Section 10.0), and dense growths of benthic algae in excess of this range have plagued the Clark Fork River for many years, accompanied by citizen complaints about the visual and aesthetic nuisance (Watson et al.1990). Studies linked the Clark Fork algae problem mainly to municipal nutrient sources. Since Montana

Table 11.1. Water-use classifications for each of the study stream sites.

Stream Site Name	Montana Water-Use Classification
RED GROUP STREAMS	
Wolf Creek at Wolf Point	B-2
Frenchman Creek	B-3
Willow Creek (S)	B-3
Battle Creek	B-3
Willow Creek (N)	B-3
Larb Creek	B-3
Little Muddy Creek	C-3
Wolf Creek nr Medicine Lake	C-3
Beaver Creek	B-3
GREEN GROUP STREAMS	
West Fork Poplar River	B-2
Redwater River	C-3
Rock Creek	B-3
Butte Creek	B-2
Big Sandy Creek	B-3
Middle Fork Poplar River	B-2
Smoke Creek	C-3
Porcupine Creek	B-3
Sheep Creek	C-3
Clear Creek	B-1
OTHERS	
Shotgun Creek	C-3
Horse Tied Creek	C-3

has a water quality standard that states (in essence) that surface waters must be free from substances that will create conditions which produce undesirable aquatic life (ARM 17.30.637(1)(e)), and given that the Clark Fork's excess algae are considered a visual and aesthetic nuisance, than the excess algae could reasonably be considered undesirable aquatic life. This line of reasoning apparently made sense to the Board of Environmental Review, because in 2002 the Board adopted standards limiting the amount of algae that is allowed on the river bottom to no more than 150 mg Ch *a*/m^2. So in Montana, excess nuisance algae that cause a visual and aesthetic impact are considered nuisance aquatic life, and therefore an impact to recreational uses of state waters.

Since a linkage has already been established between the recreational use of state waters and the aesthetic impact of nuisance algae on state waters, the same approach was applied to streams in this study. The comparison sites in the present study supported all of their beneficial uses, including the recreation use, and therefore the benthic algae densities found in those streams were considered appropriate for other wadeable streams in the

Hi-line region (see Section 10.3). Recreation is an integral part of the existing uses of these streams. For example, some of the streams in this study support warm-water game fishes (e.g. northern pike, walleye, sauger) and are routinely used by fisherman. Many of these streams are located in areas where both waterfowl and upland game hunting occurs, and hunters use the streams, stream corridors and surrounding riparian areas as hunting grounds.

But the impacts of excess algae do not end at recreational uses. Nuisance filamentous algae also clog irrigation ditches and intakes and can require costly measures to control. Therefore excess algae can also impact agricultural beneficial uses. And excess algae can result in wide diel swings in dissolved oxygen, which may impact fisheries and aquatic life.

Algae is not the only aquatic plant that may impact beneficial uses, as excess growth of macrophytes can also be viewed as undesirable aquatic life (Wong et al. 1979; Chambers et al. 1991). Although more difficult to gauge, it does not appear that macrophytes are terribly excessive in these streams. Chambers et al. (1991) reported macrophyte densities in Canadian prairie rivers in excess of 1,000 g dry weight /m^2 downstream of nutrient-rich municipal sewage outfalls; the highest benthic aquatic plant density in the present study that was composed mainly of macrophytes (95%) was 386 g AFDW/m^2 (data not shown). Some of the minimally impacted sites in the present study had as much (or more) macrophyte streambed cover as the most-impacted or intermediate-condition sites. Interestingly, the stream that supported the highest density of macrophytes (Middle Fork Poplar River) was noted by study participants for having some segments with very attractive macrophyte beds. However, more data is required before recommendations on macrophyte biomass or streambed cover limits can be made.

In summary, excess algae impacts the recreation beneficial use, and may also impact fisheries, aquatic life and agricultural beneficial uses.

11.1.2 Which Beneficial Uses Are Impacted by the Lack of Benthic Aquatic Plants?

One of the main tenets of this study is that excess benthic aquatic plant growth (particularly algae) can be a nuisance, and by state law needs to be addressed. And it resulted that excess algae was a problem is some streams, for example Big Sandy Creek. However the present study also revealed that some of the most incised, unstable channels in the study had some of the lowest benthic aquatic plant biomass, particularly for macrophytes (see Section 8.0). Three of the nine stream sites in the Red group (incised streams) had no macrophytes whatsoever, whereas all of the Green group sites (un-incised) had some macrophytes (Fig. 8.4). All of the macrophyte species found in the Red and Green groups can propagate vegetatively, and floods play one of the most important roles in their dispersal (Barrat-Segretain 1996). As a group these plants are well adapted to particular hydrologic regimes (few scouring flows, constant submersion), and therefore their complete absence from certain streams would suggest a very high degree of scouring and instability.

Given that stream incisement may result in too little benthic aquatic plants and also lead to other problems such as elevated bank erosion, the question then becomes, which beneficial uses are impacted in incised streams? It is clear that the recreational use is being impacted in streams devoid of benthic plant life as much as it is in streams with too much algae. Collapsing and unstable stream banks and a complete lack of benthic aquatic plants is every bit as much an impact to recreational uses as is excess algae. In England, macrophytes are valued as components of regional streams and the decline of some species (e.g., *Ranunculus spp*.) is a cause of concern and action by governmental agencies (Flynn et al. 2002). But more, the fisheries, aquatic life and waterfowl uses are also impacted in these streams. Aquatic vegetation is a major source of cover for fish and also supports much of the other aquatic life that fish depend upon. And waterfowl can definitely be impacted by the loss of macrophytes. *Potamogeton pectinatus* is considered the most important pondweed for ducks, *Myriophyllum exalbescens* is considered fair duck food, and the nutlets and foliage of *Zannichellia palustris* are eaten by waterfowl, especially in brackish pools (Fassett 1972). All of these plant species have been found in, and are part of, the native flora of the region's streams (Klarich 1982; Section 8.0).

From these examples it is apparent that in streams of certain hydrologic patterns too little benthic aquatic plant biomass is indicative of channel stability problems, and it can further be concluded that the recreational, fisheries, aquatic life, and waterfowl uses of incised streams devoid of benthic aquatic plants (particularly macrophytes) are being impacted. In this region the presence of a healthy and widely distributed macrophyte community should be taken to indicate that wadeable streams have a reasonable level of morphological stability, and that at least some beneficial uses are being supported.

11.2 Incised Stream Channels and Beneficial Uses

As discussed in Section 4.0, morphological instability in the Hi-line region's wadeable streams is likely driven by a combination of factors that include current and historic land uses (e.g., grazing), upland soil characteristics, and climate factors such as drought, all working together. Unusually heavy grazing was recorded in the American Great Plains starting around 1880 (Weaver and Albertson 1956; Leopold 1994), and along the Hi-line the period of heavy open-range grazing largely ended by 1907 (Spritzer 1999). Limited photo documentation (discussed in Section 1.0) showed intense stream erosion and incisement occurring in the region, at least in some locales, by no later than 1923. Thus many of the impacts seen today are likely to be legacy impacts remaining from over a hundred years ago, possibly related to the intense open range grazing period and then to the early homesteading period, which occurred immediately after (see Section 1.0). Hydrological modifications that were developed a little later in the 20th century also play a role. For example, the St. Mary Diversion (built1917) and the Fresno Dam (built 1939) have caused a three-fold increase in average Milk River flows (Bradley and Smith 1985), and it appears that incisement of the Milk River resulting from these hydrological modifications is affecting some of the tributaries. Hydrological modifications and loss of vegetation and rooting mass along stream banks from present-day cattle trampling and grazing were the two most common types of human-caused impacts observed in streams in this study. These impacts are probably the most common for the region in general.

In order to help examine stream characteristics the incised stream channels were separated out as a group (the Red group) from the un-incised stream channels (the Green group; Section 6.0). In spite of the fact that all sites in the Red group were incised, some of them were supporting all of their beneficial uses. A good example of an incised (Red group) channel that supported its beneficial uses is Wolf Creek at Wolf Point. Wolf Creek at Wolf Point is an F channel with a low entrenchment ratio (1.32), and yet it was found to support a very healthy fishery (Bramblett et al. 2003), it did not have excess algae problems, it had enough geomorphic stability to develop a robust macrophyte population, and it had a reasonably diverse and complete riparian zone. The stream was supporting its fishery, aquatic life, recreation, and waterfowl uses. Wolf Creek at Wolf Point is in sharp contrast to another Red group stream (Willow Creek {S}), which had no macrophytes, high relative proportions of phytoplankton, eroding and collapsing stream banks, only a fair fishery (Table 3.1), and a riparian zone comprised almost exclusively of annual grasses and sandbar willow. Willow Creek (S) was not supporting its waterfowl, aquatic life, and recreational uses, and was only partially supporting its fishery use.

If streams are located in basins with a large proportion of impermeable soils (see Fig. 4.2), it is more likely that stream incisement will be observed and that the stream channels will be unstable and have high erosion rates. Bank stabilization by vegetation is the best treatment of gullies in valley fill (Leopold 1994), and this would surely help to improve the beneficial use support of such streams. During any incised-stream improvement effort in this region, it should be determined whether or not any of the observed instability is related to current human activities. If yes, then best management practices (BMPs) such as riparian vegetation improvements and cattle grazing controls along the banks could be put in place to help improve stability. It would be equally important to examine the condition of the uplands to see if improvements could be made to management activities that would help increase water infiltration, since low water infiltration of soil leads to increased runoff (Rauzi and Hanson 1966) and flashy stream flows. If stream channel instability and incisement were due strictly to historic impacts, and there did not appear to be any further BMPs that could be applied, there may be little more that could be done at the present time. If a stream's current condition were caused exclusively by natural factors, then the MT DEQ would take no action to improve it.

11.3 Nitrogen Limitation and its Implications

One of the more interesting findings of the study was the strong indication that N was clearly the most limiting nutrient to benthic algae growth in these streams, in spite of the streams' great variation in riparian habitat and morphology. A review of the literature on the subject located few studies on nutrient limitation in northern prairie streams, however the studies that were found reported similar results. Bahls et al. (1992) showed that stream-water nutrient ratios indicated N limitation in a subset of the same streams examined in the present study. Neel (1985) evaluated water-column measurements of N and P in a northern prairie stream in North Dakota over a ten-year period, and concludes that during the growing season there was a lesser demand for P than for N, thus indicating N limitation. An intensive study of prairie lakes in Saskatchewan, Canada, directly north of the present study area, revealed that N was a strong factor controlling algal blooms. In fact, large reductions in municipal P-loading to the lakes since 1977 have not substantially reduced the lakes' algal abundance (Hall et al. 1999). Finally, in an examination of available stream nutrient data from across the United States, it was found that streams on glacial till soils had the highest average dissolved phosphate concentrations (Binkley 2001). All of the streams in this study (with the exception of Larb Creek) were located on glacial till, and all of them had measurable and sometimes high soluble P concentrations (Table 7.2).

As discussed in Section 7.0, N limitation occurs when aquatic plants are insufficient in N but have sufficient P. Soluble P was measured in streams throughout the sampling season, long after the spring freshet had subsided. By midsummer many streams had low TSS concentrations and therefore measurable soluble P concentrations were not simply the result of equilibrium processes between the solution phase and suspended P on particles (Froelich 1988). Significant quantities of P may have been fluxing from sediments. Sediment P flux is greatly enhanced when the water above the sediments is anaerobic (Mortimer 1941 & 1942), but can also occur at high levels when the water above the sediments is aerobic (Suplee and Cotner 2002).

The ready availability of P in these streams might also be related to the type of soil and geology (glacial till) on which the streams are located, as stream water of other streams located in glacial till were found to have especially high soluble P concentrations compared to other soil types (Binkley 2001). The streams in the present study were located in the Northwestern Glaciated Plains ecoregion, which represents the most southern extent of the last (Wisconsin) continental glaciation (Omernik 2000). The soil through which these streams flow is fairly young, as it has developed only over the past 13,000 years (Letkeman et al. 1996). In unweathered parent material, P is usually in the form of the mineral apatite and is generally immobile in soils. However, over the long periods of time during which soil develops P can be substantially translocated via biocycling and downward leaching from precipitation (Smeck 1973). So perhaps the relative youth of the region's soils plays a role in affecting P availability, in that not enough time has passed and not enough leaching has occurred to have substantially reduced available soil P.

Given that N limits algal growth, what does this imply from an algae control standpoint? Clearly, the focus of any algae control effort in these streams needs to consider sources of N. Saline seeps in the region are a localized N source that can contain highly elevated levels of nitrate (Brown et al. 1982; SD DENR 2004) and need to be addressed. Halvorson et al. (2001) studied spring wheat –fallow systems in North Dakota and show that nitrate levels increase in the soil during and following drier years, and then during wetter precipitation periods nitrate may leach below the root zone. They suggest that in order to reduce potential ground-water contamination from nitrate, it may be necessary to reduce N fertilization rates of the drier years, when N use by plants is low. A USGS study on the Fort Peck Indian Reservation found that dryland crop-fallow practices elevated nitrate concentrations in soil porewater and groundwater regionally (Nimick and Thamke 1998). The report suggests a number of land management practices that could reduce nitrate loading to groundwater which include: decreasing fertilizer use to reduce the amount of stubble residue; planting deep-rooted crops to reduce soil percolation; increasing the land area enrolled in the Conservation Reserve Program; and increasing the land area that has perennial vegetative cover. (Some of these activities would also help decrease overland water flow and the resulting flashiness of area streams by maintaining a denser plant cover better capable of taking up precipitation.) Protecting and restoring wetlands and riparian areas that trap nutrients would also be beneficial. All of these activities could be implemented in basins having excess algal problems in their streams. It would also be helpful to better limit cattle access to certain stream sites where they tend to congregate and cause localized nutrient loading problems, such as was noted at Wolf Creek nr Medicine Lake and Horse Tied Creek.

SECTION 12.0 OBSERVATIONS, CONCLUSIONS, AND RECOMMENDATIONS

12.1 Observations and Conclusions

Overall, the data from the study suggest that stream incisement is one of the major stressors to aquatic plants and other aquatic life in these streams. This is probably the result of scouring flows that can periodically occur in incised channels. Nutrients were important because they were found to play a role in affecting excessive algae growth. These findings, and others, are elaborated upon in the points below:

1. TSS and turbidity were both significantly correlated (negatively) to total aquatic plant biomass, percent streambed cover by macrophytes, and percent streambed cover by filamentous algae. Entrenchment ratio was significantly correlated (positively) to both total aquatic plant biomass and floating & benthic algal biomass. A higher entrenchment ratio indicates that a stream is less incised (downcut) and has better access to its flood plain; therefore, the entrenchment ratio-aquatic plant correlations indicate that more incised streams had lower benthic plant biomass. TSS and turbidity were probably acting as surrogate measures for high erosion, substrate instability, and scouring flows, all of which tend to be higher in incised streams. Taken together, these correlations suggest that stream incisement and the associated scouring flows are one of the major stressors to aquatic plants and other aquatic life in these streams.

2. Permeability of regional soils was significantly correlated (positively) to the entrenchment ratio of the study streams (i.e., incised channels were found in the basins with the least permeable soils). Rosgen C channels were generally located in basins having a high proportion of permeable soils, whereas G channels were usually located in basins having a high proportion of impermeable soils. The data suggested that in basins where $\geq$ 90% of the land area has impermeable to moderately-slow permeability soils the stream channels are likely to be incised. Stream channel incisement in this region is probably driven by the combined influences of plant cover of the uplands and the riparian, inherent soil permeability, and variation in climate factors such as intensity of rainfall and drought. Once the vegetated cover is lost or diminished, streams in more impermeable basins are probably more vulnerable to incisement and headcutting. An historical investigation into regional stream morphology changes suggested that some of the incisement observed in the region occurred in the late 19th and early 20th centuries, possibly as a result of the open-range grazing period and the period of early homesteading.

3. The stream sites demonstrated diverse physical characteristics (riparian type, channel morphology, etc.), and to help better explain some of the data it became necessary to develop ecological subgroups. Using cluster analysis two major

groups, “Red” and “Green”, were developed using stream entrenchment and riparian characteristics. The Red group was composed of Rosgen G and F streams having low entrenchment ratios, low canopy density (0-14.4%) and low to intermediate woody vegetation density (0-58%). The Green group was comprised of streams that were moderately to slightly entrenched (Rosgen B, C, and E classes), had generally low canopy density (0-4.4%) and had low to moderate woody-vegetation density (0-35%). Two streams did not fall into either of these groups because their riparian areas where so different from either group. One of the streams had high canopy density (60%) and high density of woody vegetation (83%; Horse Tied Creek) while the other had high canopy density (63%) and no woody vegetation at all (Shotgun Creek). Streams of the Red group had lower total aquatic plant biomass, lower density of floating & benthic algae, fewer sites with macrophytes, and higher relative proportions of phytoplankton compared to streams of the Green group.

4. Data from four different lines of evidence (algal cellular C:N:P ratios, water column nutrient concentration ratios, algal alkaline phosphatase activity, and large numbers of heterocysts on cyanobacteria) clearly indicated that N was the limiting nutrient to floating & benthic algae in the study streams.

5. The $NO_{2/3}$ concentration in stream water was significantly (p-value $\leq$ 0.05) correlated (positively) to floating & benthic algal biomass and this finding supported those in No. 4 above. No other nutrient measured in the study (TN, TP, TDP or SRP) was significantly correlated to floating & benthic algal biomass. Because N is the most limiting nutrient to algae growth in these streams, any effort at controlling algae growth will need to consider practices that reduce N loading.

6. EPA’s recommended procedure for selecting nutrient criteria – whereby the 25th percentile of a whole population’s distribution is compared to the 75th percentile of a distribution of “reference” sites – did not work well for this region. There was too much overlap between the distribution of the comparisons sites and the distribution of the entire set of study sites for the approach to work (this was true of both N and P). An alternative approach was developed which identified N, P and algal biomass criteria based on the level of confidence in the *quality* of the comparison sites (see sub-Section 10.2 for details). Based on this “quality of the comparison sites” approach, the comparison sites in this study were rated as being of “good” quality, and so values corresponding to the 75th percentile were identified as appropriate criteria.

7. In the comparison sites the average floating & benthic algal Chl *a* density was 43.6 mg/m^2, the median was 19.7 mg/m^2, the value at the 75th percentile was 61 mg/m^2, and ninety-five percent of the samples were below 106 mg/m^2.

8. With the exception of Big Sandy Creek, average filamentous algae streambed cover did not exceed 30% in all the study sites and the value at the 75th percentile

of the comparison sites was 25%. Big Sandy Creek (68% mean streambed cover) was apparently polluted with excess nitrate.

9. Stream water Chl *a* ranged from 0.4 to 515 µg/L and was used as a measure of phytoplankton biomass. The value at the 75th percentile of the comparison sites was 14 µg/L, and ninety-five percent of the comparison site samples were below 20 µg/L. Two sites that had very specific types of nutrient impacts (cattle congregating within a small area in the channel) had unusually high stream-water Chl *a* concentrations. Streams that were devoid of macroscopic benthic aquatic plants (macrophytes, filamentous algae) had much higher relative proportions of phytoplankton then did streams with substantial standing biomass of benthic aquatic plants.

10. Excessive floating algae, benthic algae and phytoplankton algae impacted the recreational beneficial uses of some study streams. The complete lack of benthic plants such as filamentous algae and macrophytes impacted the recreational, fisheries, aquatic life, and waterfowl beneficial uses of other study streams.

11. Light availability (canopy cover), substrate, electrical conductivity, and flow status (intermittent or perennial) were unable to explain any variation in floating & benthic algal biomass.

12. When evaluating nutrient availability in prairie steams using diatoms metrics, fine sediment benthic samples (the core or “mud plug” sampling method) from depositional area such as pools are best. This is because fine sediment benthic samples are collected in areas that accumulate diatom taxa from all other habitats.

13. Aquatic macroinvertebrate samples from riffle areas are useful for assessment of sedimentation issues, but riffles are not common in many prairie streams because of the lack of gravel and because many become intermittent during base flow. Aquatic macroinvertebrate samples from depositional areas such as pools had high taxonomic diversity, but much work remains to be able to use these data to make conclusions about stream condition. Corixids (water boatman) and Odonates (dragonfly larvae) hold promise as potential indicators of water quality and habitat condition in prairie streams.

14. Excess selenium does not appear to be a problem in these streams. Selenium was measured in all of the sites in both years of the study, none of which violated state water quality standards. Out of 78 samples collected, only one was above the detection limit.

12.2 Recommendations

All of the values below are applicable to wadeable streams (Strahler order 1 through 4) of the Northwestern Glaciated Plains ecoregion (Omernik 2000), otherwise known as the Hi-line region. They are intended for the aquatic-plant growing season, May through September.

1. Streambed cover by filamentous algae: for a single sample, the value should not exceed 30%; for an average over the growing season, the value should not exceed 25%. These values should prevent nuisance levels of growth.

2. Floating & benthic algae: for a single sample, the value should not exceed 110 mg Chl a/m^2; for an average over the growing season, the value should not exceed 65 mg Chl a/m^2. Together these values should maintain algae below a nuisance level.

3. Stream water Chl *a*: for a single sample, the value should not exceed 20 ug/L; for an average over the growing season, the value should not exceed 15 µg/L. These concentrations should keep phytoplankton below a nuisance level.

4. The nitrate + nitrite concentration should not exceed 2.3 µg/L, which should prevent nuisance floating & benthic algae. This is the most important nutrient among those listed here, as it showed a significant correlation to floating & benthic algae density.

5. The total N concentration of stream water should not exceed 1,044 µg/L in order to prevent nuisance algae.

6. The total P concentration of stream water should not exceed 153 µg/L in order to prevent nuisance algae. This value is of less significance than the N recommendations, because stream algae appear to be N limited.

7. The TDP concentration should be kept below 49 µg/L and the SRP concentration below 8 µg/L in order to prevent nuisance algae. These values are of less significance than the N recommendations, because stream algae appear to be N limited.

In this region the presence of a healthy and widely distributed macrophyte community should be taken as indicative that the streams have a reasonable level of morphologic stability. As a group macrophytes are well adapted to particular hydrologic regimes (few scouring flows, constant submersion) and their complete absence from a stream should be taken as potentially indicating that the site has streambed stability problems and scouring flows.

The Red and Green groupings can be used to help reduce intra-regional variability of stream site characteristics and to better apply appropriate comparison sites. Caution should be used when using the groups over time, as stream types may evolve and some members of the Red groups may eventually be better placed among those of the Green group.

In the Hi-line region, watershed restoration efforts should consider the soil permeability in the basins of the streams in question. To slow or diminish regional stream incisement, streams located in basins with high proportions of low permeability soils will probably require more careful management than those in higher-permeability basins. Knowing in advance the types of soils in a basin may help direct the choice of best management practices selected for the basin.

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APPENDIX A

Evaluation of EMAP vs. Rosgen Stream Morphology Measurements in Determining a Rosgen Stream Type

Rosgen (1996) suggests that bankfull mean depth and substrate D_{50} be determined from measurements taken from bankfull to bankfull, whereas EMAP makes the same measurements but only within the wetted channel (USEPA 2000b). Both methods result in a total count of 100 particles for the D_{50} estimate.

To evaluate the impact of the EMAP/Rosgen methodological differences on the identification of a Rosgen stream type, ten streams were selected in 2002 and measured using both Rosgen Level II and EMAP methods. It resulted that EMAP mean bankfull depths were higher, and therefore the W/D ratios calculated were lower. The magnitude of the difference between EMAP and Rosgen W/D ratios varied; Rosgen W/D ratios were equal to or as much as three times higher then EMAP W/D ratios. However out of the ten stream sites reviewed in only one case would the EMAP W/D ratios have resulted in a Rosgen stream class different than that which would have been identified using only the Rosgen methods.

There were also considerable differences between Rosgen and EMAP in there respective measurements of sediment D_{50}. In general, EMAP's protocols resulted in smaller sediment D_{50} values compared to Rosgen. EMAP and Rosgen pebble counts were conducted in five of the study streams in 2002. In two streams the resulting channel-material size class (1-6 scale; Rosgen 1996) were identical, in the other three the EMAP pebble counts resulted in a smaller channel material size class then the Rosgen counts (e.g., class 6 [silt/clay] rather then class 5 [sand]). Figure A shows the difference in channel material D_{50} estimates for two of the streams using Rosgen vs. EMAP methods.

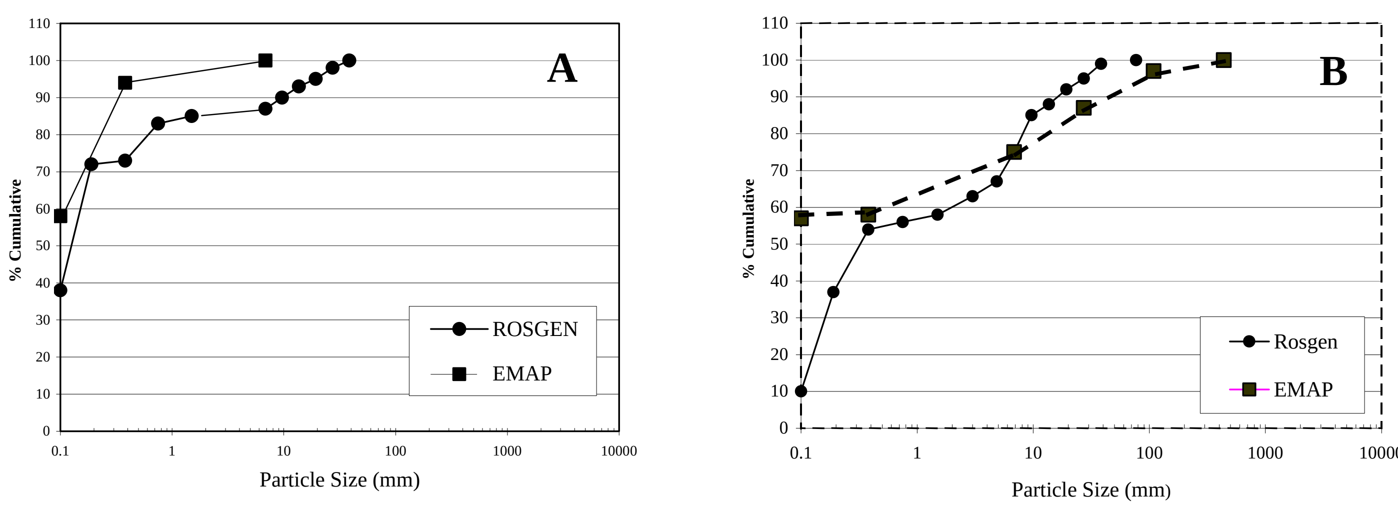

Figure A. Comparison of Rosgen and EMAP pebble counts in two streams. A. Wolf Creek nr Wolf Point (F5 stream type). B. Middle Fork Poplar River (C5 stream type).

APPENDIX B

Effect of Acid Rinsing of Samples on the Algal C:N:P Ratios

Based on approaches described in the literature (Froelich 1980; Nieuwenhuize at al. 1994), it was decided at the initiation of the project to acid rinse all algae samples with12% HCl (followed by a tap water rinse) if they were to be analyzed for C and N content. The intent of the procedure was to remove carbonates on the benthic algal material that, if included in the analysis, would artificially inflated the C fraction of the sample and altered the calculated ratios. In 2002, we looked more closely at this procedure because of concern that we may have been impacting the N content of the cells during the rinsing process. Therefore, paired samples were prepared as either acid rinsed or not acid-rinsed, and each pair was submitted for C & N analysis.

It was observed during most rinsings of the algal material that fizzing and off gassing occurred, indicating that carbonates were being removed. However, this process does not appear to have had a systematic impact on the N content of the algal material. Twenty-four paired samples were analyzed, and there is no significant (p-value ≤ 0.05) difference between the N:P ratio of the acidified vs. the un-acidified samples (Mann-Whitney test; p-value = 0.98). That is, N was not systematically lower in all cases where acid was used to treat the samples. This is demonstrated in Fig. B.1.

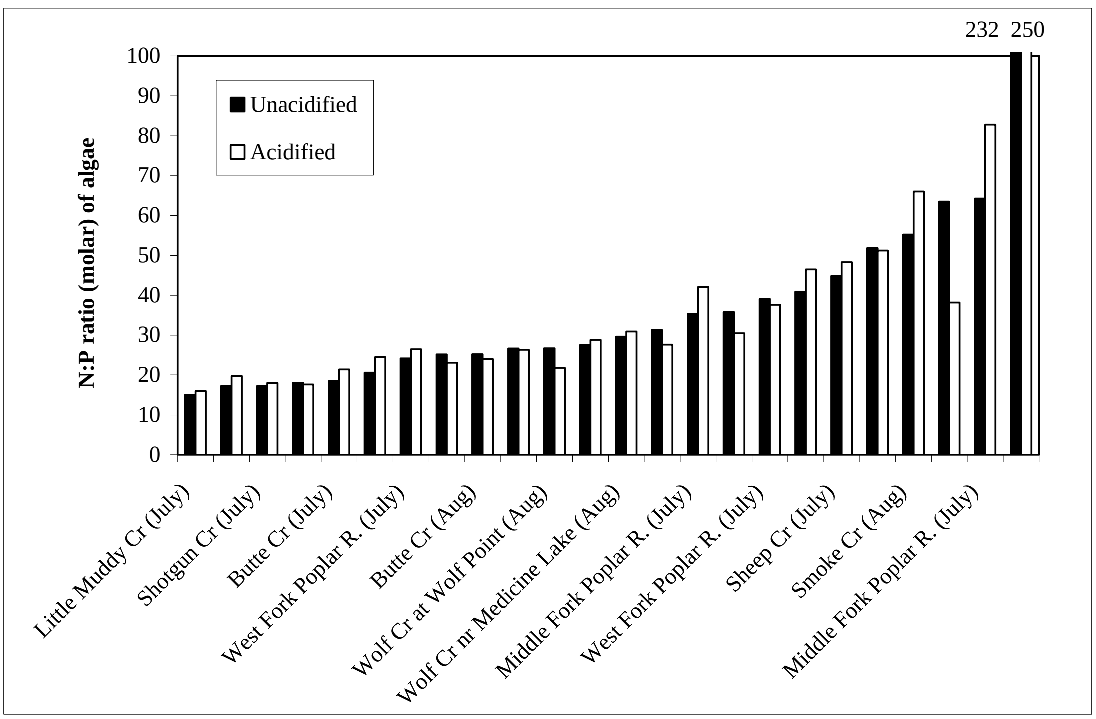

Figure B.1. Comparison of N:P ratios of samples having been either acid rinsed (12 % HCl) or not acid rinsed prior to analysis. Sites arranged in order from lowest to highest to help with interpretation of results.

On the other hand, there is a highly significant difference between C:N ratios of the acidified vs. the un-acidified samples (Mann-Whitney test; p-value = 0.002). The acidified samples have consistently lower ratios (Fig. B.2) and indicate that inorganic C was (as desired) being systematically removed from the algae samples. From these data it appears that acid rinsing is an effective means of preparing the samples to remove inorganic C, without unduly impacting the N content of the algal cells.

None of the samples analyzed for P-content were ever rinsed with HCl. Algal P-content was undertaken as a separate analytical process; see Solorzano and Sharp (1980). The recovery efficiency of this method was evaluated by analyzing a dried tomato leaf standard, which had a NIST Certificate of Analysis of 0.216% P. Recoveries ranged from 83-91%. Tomato leaves were selected because they most resembled the algal material analyzed in this project.

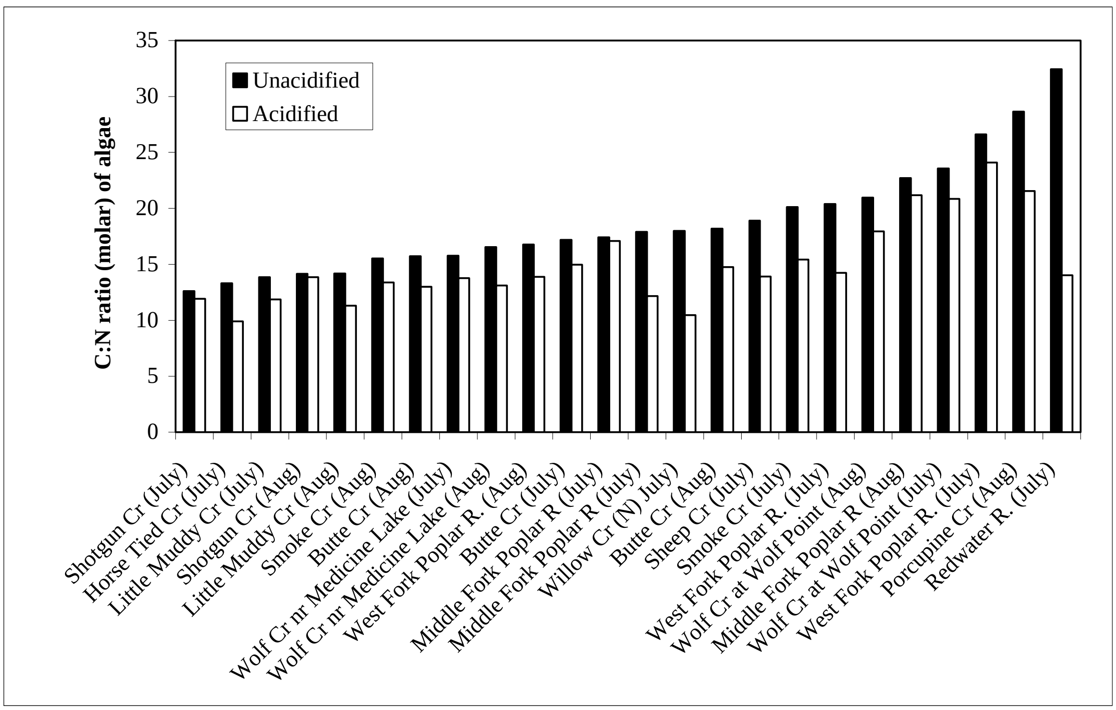

Figure B.2. Comparison of C:N ratios of samples having been either acid rinsed (12 % HCl) or not acid rinsed prior to analysis.

APPENDIX C

Temporal Plots of Nutrient Concentrations

When monthly data was plotted for a single year (Fig C.1A&B), or combined for both years of the study (Fig. C.2A&B), no clear month-to-month patterns emerged in either the reference & minimally impacted sites or in the study sites as a whole. A lack in pattern was also noted for total P during the same time periods (data not shown). This is further demonstrated in Fig. C.3A & B, which shows the monthly total N and total P concentrations for all of the reference & minimally impacted sites, two sites of intermediate condition, and two most impacted sites (condition rankings based on the results from Section 3.2). The only obvious temporal pattern in Fig. C.3 is a similarity in the changes of total N concentrations in the West Fork Poplar River and Butte Creek, both of which are in the same drainage. Rock Creek, which is located in the adjacent watershed to these streams, shows a similar seasonal pattern.

Much like the intra-year variability in nutrient concentrations, year-to-year variation was also high and did not demonstrate any clear tendencies. Three sites (Willow (N), Rock, and Porcupine creeks) were sampled for nutrients during the spring and summer months of both years of the study (2001 and 2002). Data are shown in Fig. C.4A &B. Instream concentrations during the same months of different years were highly variable, and any trend of increasing or decreasing nutrient concentrations through the season of a given year at a given site was generally not the same trend seen in the other year. Willow Creek (N) is a good example of this (Fig. C.4).

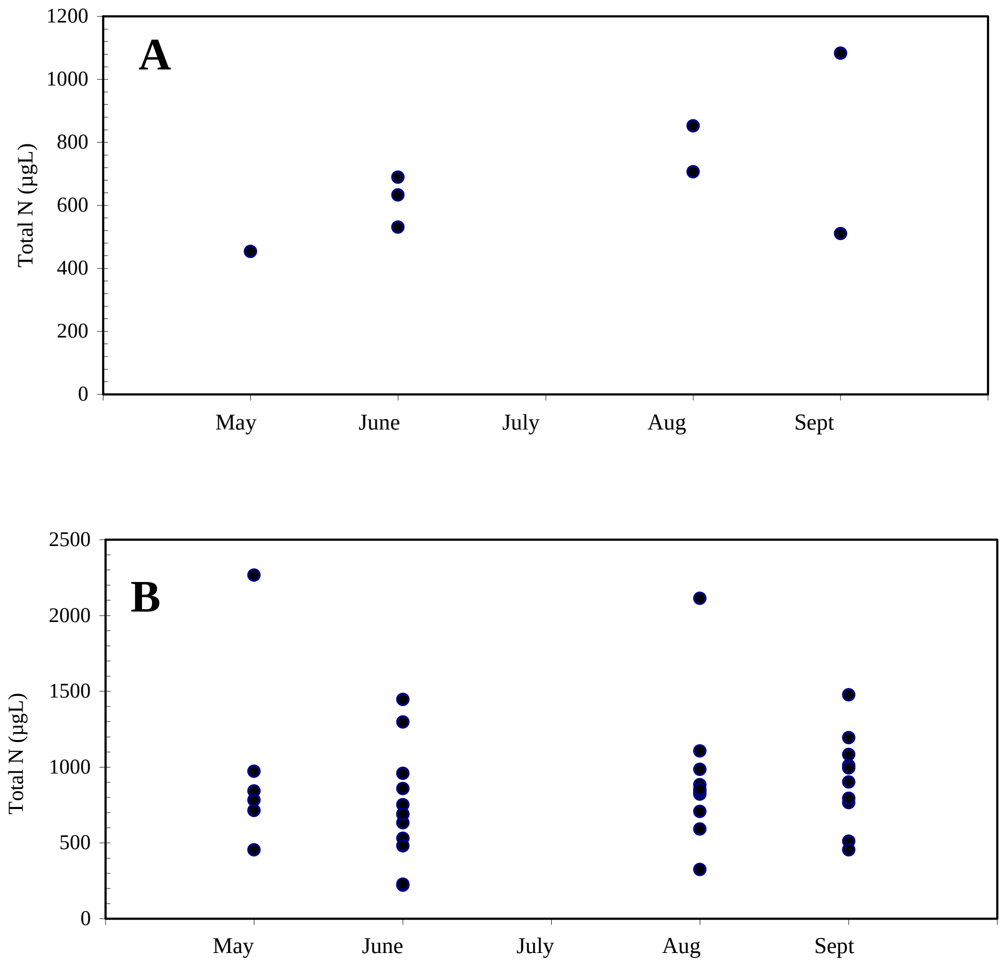

Figure C.1. A. Temporal variation in total N concentrations from May to Sept. in the ref/min. impacte sites, 2001. B. Temportal variation in total N concentration from May through Sept. in all sites, 2001.

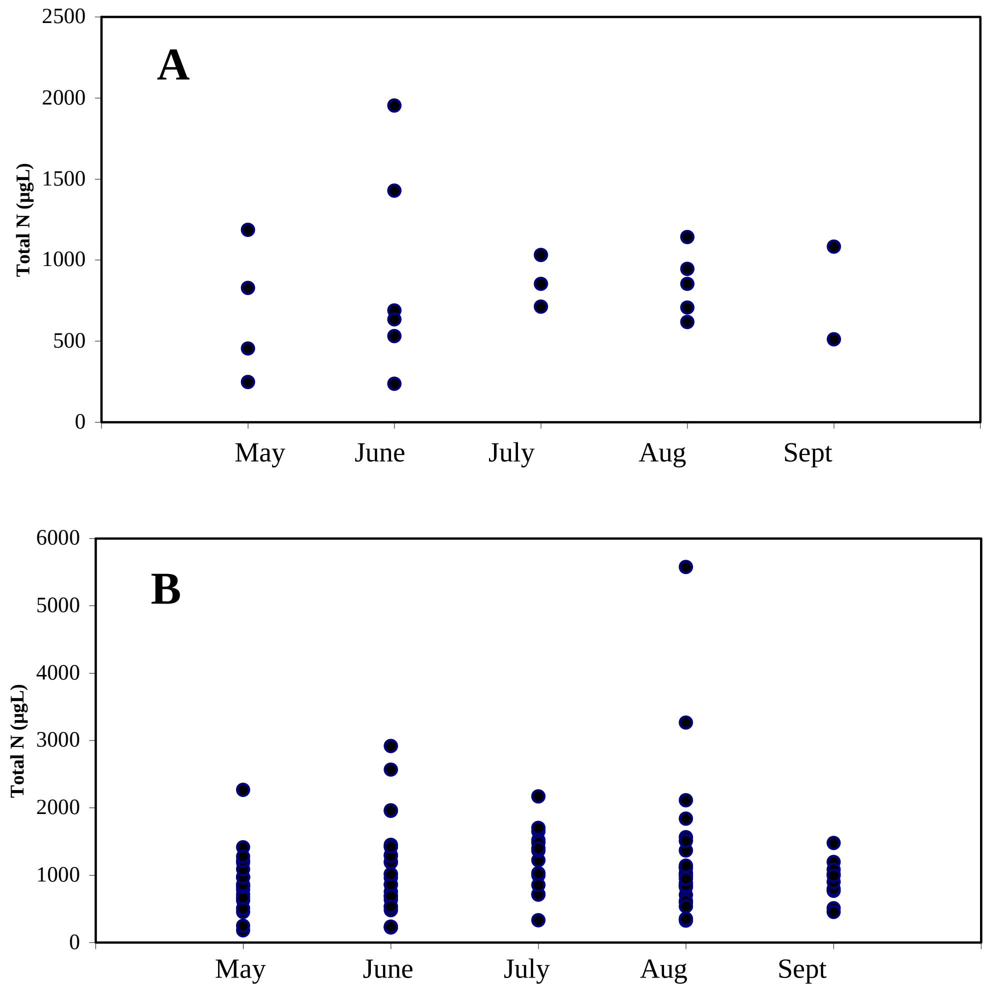

Figure C.2. A. Temporal variation in total N concentrations from May to Sept. in the ref/min. impacted sites, 2001 & 2002. B. Temporal variation in total N concentration from May through Sept. in all sites, 2001 and 2002.

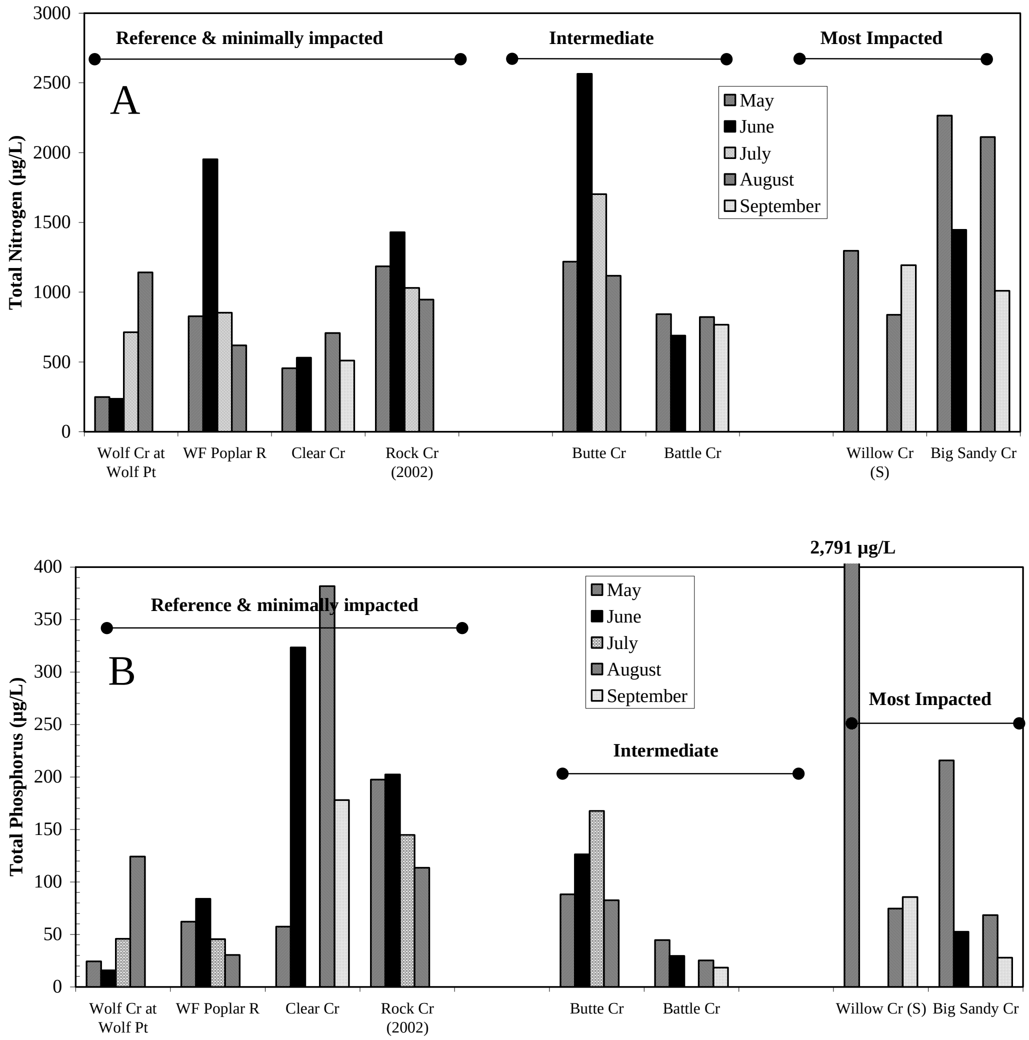

Figure C.3. A. Monthly variation in total N concentrations in the ref/min. impacted sites, two sites from the intermediate-condition group and two sites from the most impacted group. B. Total P concentration for the same sites and time periods as A.

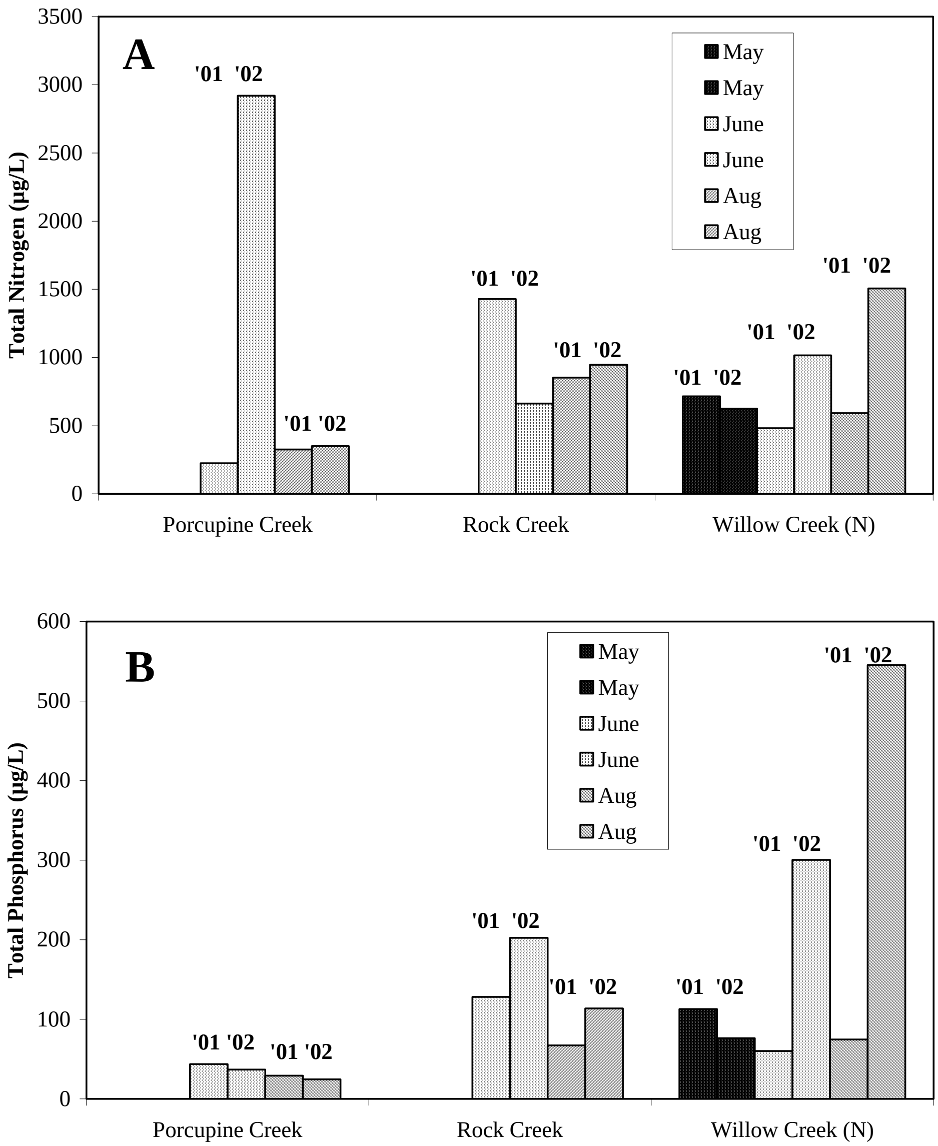

Figure C.4. A. Year-to-year variation in monthly total N concentrations. Year samples were collected are shown above the bars. B. Year-to-year variation in monthly total P concentrations in the same streams.