National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science

Aquatic Resources along the Little Missouri River near Theodore Roosevelt National Park, North Dakota

Natural Resource Technical Report NPS/NGPN/NRTR—2013/777

ON THIS PAGE
Mark Andersen of the Wyoming Natural Diversity Database measuring water quality along the Little Missouri River at Sully Creek State Park
Photograph by: Lusha Tronstad, Wyoming Natural Diversity Database

ON THE COVER
Little Missouri River before flowing into Theodore Roosevelt National Park
Photograph by: Lusha Tronstad, Wyoming Natural Diversity Database

Aquatic Resources along the Little Missouri River near Theodore Roosevelt National Park, North Dakota

Natural Resource Technical Report NPS/NGPN/NRTR—2013/777

Lusha Tronstad

Wyoming Natural Diversity Database
1000 East University Avenue, Department 3381
University of Wyoming
Laramie, WY 82071

July 2013

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

The National Park Service, Natural Resource Stewardship and Science office in Fort Collins, Colorado, publishes a range of reports that address natural resource topics. These reports are of interest and applicability to a broad audience in the National Park Service and others in natural resource management, including scientists, conservation and environmental constituencies, and the public.

The Natural Resource Technical Report Series is used to disseminate results of scientific studies in the physical, biological, and social sciences for both the advancement of science and the achievement of the National Park Service mission. The series provides contributors with a forum for displaying comprehensive data that are often deleted from journals because of page limitations.

All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible, technically accurate, appropriately written for the intended audience, and designed and published in a professional manner. This report received informal peer review by subject-matter experts who were not directly involved in the collection, analysis, or reporting of the data. Data in this report were collected and analyzed using methods based on established, peer-reviewed protocols and were analyzed and interpreted within the guidelines of the protocols.

Views, statements, findings, conclusions, recommendations, and data in this report do not necessarily reflect views and policies of the National Park Service, U.S. Department of the Interior. Mention of trade names or commercial products does not constitute endorsement or recommendation for use by the U.S. Government.

This report is available from the Northern Great Plains Inventory & Monitoring Network website (http://science.nature.nps.gov/im/units/NGPN), the Natural Resource Publications Management website (http://www.nature.nps.gov/publications/nrpm/), and the WYNDD website (http://www.uwyo.edu/wyndd/reports-and-publications/). To receive this report in a format optimized for screen readers, please email irma@nps.gov.

Please cite this publication as:

Tronstad, L. 2013. Aquatic resources along the Little Missouri River near Theodore Roosevelt National Park, North Dakota. Natural Resource Technical Report NPS/NGPN/NRTR—2013/777. National Park Service, Fort Collins, Colorado.

NPS 387/121725, July 2013

Contents

Figures

Page

Tables

Page

Abstract

Water quality in the Little Missouri River, North Dakota is potentially threatened by high concentrations of fecal coliform bacteria and a water diversion. Fecal coliform concentrations are above the standards for recreation in one reach of the Little Missouri in and near the South Unit of Theodore Roosevelt National Park. Concentrations are near the recreation water quality standard in and near the North Unit of the Park. Additionally, a water diversion was built upstream of the South Unit of Theodore Roosevelt National Park, which may alter the hydrology of the river. To investigate ecosystem quality along the Little Missouri River, I collected basic water quality, fecal coliform concentrations, Escherichia coli concentrations, and aquatic invertebrate samples at four sites. Basic water quality was similar among sites. Fecal coliform concentrations were above the water quality standard for recreation at all sites, and E. coli exceeded the limit at the downstream site. Aquatic invertebrate samples indicated that the sites upstream of the South Unit of Theodore Roosevelt National Park were in good condition according to the Macroinvertebrate Biotic Integrity Multimetric Index of North Dakota. However, individual metrics suggested that the water diversion may be impacting the river. The Little Missouri River at the North Unit of Theodore Roosevelt National Park had lower ecosystem quality according to the Macroinvertebrate Biotic Integrity Multimetric Index of North Dakota and individual metrics, but the lower score may be at least partially due to sand substrate. Fecal coliform concentrations are high in the Little Missouri River, which may be due more to livestock grazing than municipal waste. To reduce fecal coliform concentrations, land management practices will likely need to change throughout the affected watershed and within Park management actions will likely have minimal effect on concentrations. To understand the ecosystem quality of the Little Missouri River at the North Unit of Theodore Roosevelt National Park, a more intensive study should be conducted.

Acknowledgments

I am grateful to Mark Andersen for assisting with field work, and Mark Andersen and Gary Beauvais of the Wyoming Natural Diversity Database for comments on an earlier draft of the report. Thanks to Cody Bish, Kyle Hack, and Oliver Wilmot for their efforts in the laboratory. Chad Sexton and Mike Oehler provided logistical assistance at Theodore Roosevelt National Park. Stephen Wilson, National Park Service, graciously provided shape files and Marcia Wilson, National Park Service, aided the study in countless ways. Thanks to Jeff Hughes of the National Park Service who provided information about the water diversion and Aaron Larsen of the Division of Water Quality in North Dakota for providing information on the Macroinvertebrate Biotic Integrity Multimetric Index of North Dakota. Lisa Murfitt of the Wyoming Department of Agriculture Analytical Services analyzed water samples for bacterial concentrations.

Introduction

The growing human population is placing increased pressure on rivers worldwide (Allan 2004). Forty percent of wadeable streams in the plains and lowland region of the United States is considered to be in poor condition (Paulsen et al. 2008). Habitat, water quality, and biota of rivers are changing in complex ways as a result of anthropogenic activities, such as urbanization, agriculture, forestry, mining, and recreation (Allan 2004). Most studies of land use and ecosystem quality of rivers contrast urbanized, agricultural, and natural areas (e.g., Hall et al. 2009). Streams in cities tend to have higher concentrations of many pollutants, and streams suffer many consequences from draining large areas with impermeable surfaces (e.g., cement and asphalt; Allan 2004). Streams draining agricultural lands tend to have higher concentrations of nutrients (e.g., nitrogen and phosphorus), pesticides, herbicides, and more fine sediments. Overall, the largest stressors for rivers through the United States are high concentrations of nitrogen and phosphorus, excessive fine sediments, riparian disturbance, loss of habitat for fish, and to a much lesser extent, high salinity and acidification (Paulsen et al. 2008).

Rivers are sensitive monitors of what occurs in a watershed because of their position on the landscape. Rivers are always located at a lower elevation compared to the surrounding topography causing water and other material to drain into rivers. Thus, land and water are tightly linked and what occurs on land will affect aquatic resources (e.g., Allan 2004). Many studies have addressed how land use can affect the water chemistry, plants, and animals in river ecosystems. Land use studies generally investigate such effects at several spatial scales (e.g., reach, segment and watershed) and find that certain attributes are more correlated at different scales. For example, water chemistry (e.g., nitrogen concentrations and conductivity) had higher correlations with land use at the watershed scale (Sponseller et al. 2001; Johnson et al. 2007), but invertebrate diversity (Fitzpatrick et al. 2001), life history (Doledec et al. 2006; Vandewalle et al. 2010), and assemblage structure (Feld 2013) had higher correlations at smaller scales (e.g., stream reach).

Agricultural land use and water diversions can reduce the ecosystem quality of rivers. Of the 150 major river basins in North America, 0 to 66% of land in each basin is used for agriculture (Benke and Cushing 2004). Agricultural operations are diverse in what they produce and how the land is managed. Studies have found that farming (e.g., row crop agriculture) has a larger impact on river ecosystem quality compared to pastoral agriculture (Allan 2004). Farming tends to increase nutrient concentrations and fine sediments in rivers. Livestock production can decrease ecosystem quality in rivers if livestock are kept at high densities (Gammon et al. 2002), reduce riparian vegetation (Saunders and Fausch 2007; 2012), or defecate in the water (del Rosario et al. 2002). Similarly, diverting water from streams can impact ecosystem quality. After diverting 40-80% of the flow from three streams for 3 years, invertebrate density, biomass, and certain functional feeding groups (filterers, gatherers and scrapers) decreased in the Yale Myers Research Forest, Connecticut (Walters and Post 2011). The impact of diverting water from streams may depend on the initial quality of the stream. Dewson et al. (2007) noted changes to invertebrates that lived in high quality streams after diverting 85% of the flow for 2 months, but no change in a polluted stream.

The Little Missouri River in western North Dakota is potentially threatened by dewatering, high bacterial concentrations, and energy development. A water diversion was built upstream of

Medora, North Dakota to remove water from the river to irrigate a golf course. The Theodore Roosevelt Medora Foundation has a permit to withdraw water from the Little Missouri River at 18.93 m^3/min and use 493,392 m^3 of water between 1 March and 1 July each year. High concentrations of fecal coliform bacteria have impaired the rivers use for recreation. Fecal coliform are aerobic, gram negative bacteria that include species in the genera *Escherichia* and *Aerobacter* (Sarai 1976). High concentrations of fecal coliform in water can give swimmers cramps and diarrhea. Finally, western North Dakota has been the site of intensive energy development. Energy development typically pumps subsurface water of varying quality to the surface, and hydraulic fracturing can introduce unknown chemicals into ground water that may enter the river. Because of these potential threats, the National Park Service wanted to investigate ecosystem quality of the Little Missouri River near Theodore Roosevelt National Park. The objectives of the study were to 1) measure the degree to which the water diversion impacted the river according to aquatic invertebrates, 2) estimate how the aquatic invertebrate assemblages varied with fecal coliform concentrations, and 3) calculate ecosystem quality along the Little Missouri River using aquatic invertebrates as bioindicators. To answer these questions, I collected and analyzed basic water quality, bacterial concentrations, and aquatic invertebrate samples at four sites along the river.

Study Area

The Little Missouri River is a ~900 km long un-dammed tributary stream of the Missouri River in western North Dakota. The river originates in northeastern Wyoming, runs through the badlands of North Dakota, and flows into Little Missouri Bay of Lake Sakakawea near Killdeer, North Dakota. Because of the natural treeless nature of the landscape and the erodible geology, discharge of the Little Missouri River can fluctuate dramatically. For example, discharge may increase sharply after a thunderstorm. Average annual flow of the Little Missouri River is 13.2 m^3/s at Medora and 15.8 m^3/s at Watford City (mean annual discharge 1904-2011 and 1936-2011, respectively; USGS National Water Information System, www.waterdata.usgs.gov).

Theodore Roosevelt National Park was formed in 1947 and includes 285 km^2 in three units (North Unit, South Unit, and Elkhorn Ranch Unit). About 18.3 km of the Little Missouri River flows through the South Unit of Theodore Roosevelt National Park and ~26 km flows through the North Unit of the Park. All units in the Park are dominated by native grasslands. The areas surrounding the Park are mainly used as range for livestock production and energy development.

The Little Missouri River had a well-developed riparian area and fine benthic substrates. The riparian area surrounding the Little Missouri River was dominated by cottonwood (*Populus deltoides*), willow (*Salix* spp.), and juniper (*Juniperus* sp.). The riparian area was often separated from the river by steep cliffs. The depth of the Little Missouri River varied among sites, but was generally wadeable where I sampled (<1.5 m deep). River flow was 4.4 m^3/s (at Medora) and 7.0 m^3/s (near Watford City) during this study (USGS National Water Information System, www.waterdata.usgs.gov). Benthic substrates were composed of cobble, gravel, silt, sand, clay, and coal. The organic matter content of benthic substrates varied among sites.

The Little Missouri River was a Class II water with a designated use of recreation; therefore, water quality must be maintained for safe human contact (e.g., swimming). Under the Clean Water Act of 1972, each river was assigned a class based on the designated uses of the water (e.g., drinking water, fisheries). The water quality of the river must be within standards set for each designated use. Currently, the Little Missouri River from its confluence with Deep Creek to the confluence with Andrew's Creek (77.6 km) was not supporting its recreation designation, because of high concentrations of fecal coliform (Water Quality Integrated Report 2010). In addition, the recreation designation was threatened by high concentrations of fecal coliform in three additional reaches of the Little Missouri River (255 km; confluences of Little Beaver Creek to Deep Creek, Beaver Creek to Highway 85, and Highway 85 to Cherry Creek) and one tributary stream (68.4 km; Deep Creek).

I collected samples at four sites along the Little Missouri River that were chosen to address the study objectives. Site #1 was located on State land upstream of the water diversion by the Theodore Roosevelt Medora Foundation (Figure 1a, Table 1). I collected samples downstream of the diversion at Sully Creek State Park (site #2). Site #3 was located immediately before the river flowed into the South Unit near the USGS gauging station. Finally, I sampled the Little Missouri River where it flowed out of the North Unit upstream of highway 85 (site #4; Figure 1b).

Table 1. Locations of the four sites sampled along the Little Missouri River near Theodore Roosevelt National Park (Datum NAD83).

Site	Latitude (North)	Longitude (West)	Date sampled
Site #1	46°50’51.6”	103°32’56.2”	1 Sept 2011
Site #2	46°53’32.8”	103°32’21.3”	31 Aug 2011
Site #3	46°54’58.0”	103°31’55.3”	31 Aug 2011
Site #4	47°35’26.5”	103°15’14.5”	30 Aug 2011

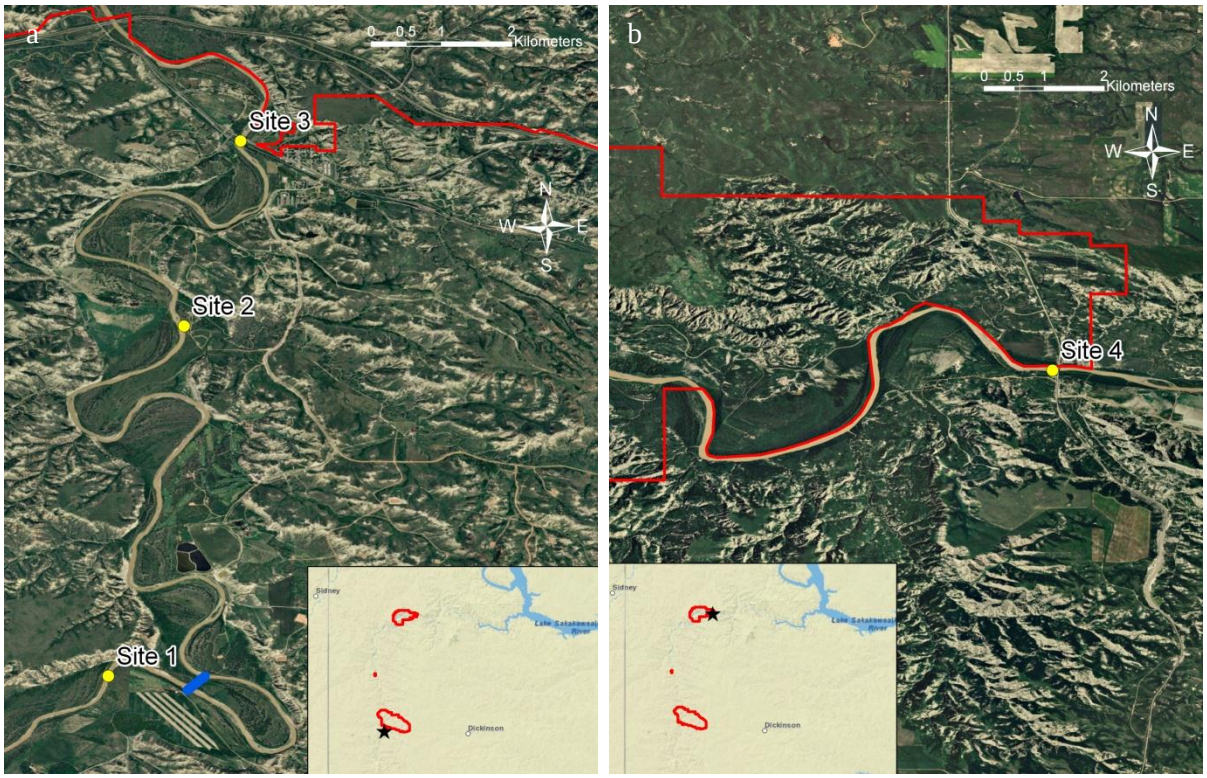

Figure 1. I sampled four sites (yellow circles) along the Little Missouri River near the South Unit (a) and North Unit (b) of Theodore Roosevelt National Park (red boundary). The river changed course after the aerial imagery was taken, thus the blue line (a) shows the new course of the river creating an oxbow. The inset maps, showing western North Dakota, outline the location of Theodore Roosevelt National Park (red boundaries) and the area enlarged (star).

Methods

To estimate conditions at each site, I measured basic water quality, water clarity, and bacterial concentrations. I measured basic water quality using a Yellow Springs Instrument (YSI) Professional Plus that was calibrated daily. Water clarity was estimated by lowering a Secchi disk into the water until the disk disappeared from sight. I collected two water samples from each site to measure the concentration of fecal coliform and *E. coli* using the Colilert method (SM9223B, Eaton and Franson 2005). Water samples were immediately placed on ice and shipped to the Wyoming Department of Agriculture Analytical Services. All samples were received by the laboratory within 30 hours of collection. Finally, I recorded the location of each site using a Garmin eTrex GPS unit (Datum NAD83).

To measure the abundance and diversity of aquatic invertebrates in the Little Missouri River, I collected aquatic invertebrate samples using a Hess sampler. Five samples were collected at each site. Typically, I collected samples in the most dominant habitats according to their abundance at each site. To collect invertebrates, I placed the Hess sampler (500 µm mesh, 860 cm^2 sampling area, Wildlife Supply Company) into the substrate and agitated the sediment and vegetation when present. Samples were elutriated to separate invertebrates from substrate (e.g., gravel). Samples were preserved with ~80% ethanol and transported to the laboratory where invertebrates were sorted from debris. Each sample was checked for invertebrates by two qualified people to insure that all invertebrates were removed. Invertebrates were counted and identified under a dissecting microscope using appropriate keys (Needham et al. 2000; Smith 2001; Merritt et al. 2008; Thorp and Covich 2010).

To estimate ecosystem quality at each site, I calculated several bioassessment metrics using invertebrate data. Based on the data collected, previous studies (e.g., Resh and Jackson 1993; Kerans and Karr 1994), and models developed for North Dakota (Environmental Protection Agency 2009), I selected 22 metrics to compare sites (Table 2). I choose a variety of metrics including measures of richness, abundance, community diversity, pollution tolerance, habit, and functional feeding group. Pollution tolerance values of invertebrate taxa were taken from Bowles et al. (2008) and Barbour et al. (1999). Functional feeding group and habit were from Merritt et al. (2008). To distinguish among sites, I used ANOVA to compare abundance and bioassessment metrics for each sample (DataDesk6.1). Differences among sites were distinguished using Bonferroni multiple comparison tests, where differences were significant when $p < 0.0125$ (0.05/4; where I had four sites). Reported variance is standard error.

North Dakota developed the Macroinvertebrate Biotic Integrity Multimetric Index created under the Environmental Monitoring and Assessment Program (EMAP) West by the Environmental Protection Agency (Environmental Protection Agency 2009). These metrics have been developed for the rangeland plains of North Dakota (Northwestern Great Plains and Northwestern Glaciated Plains Ecoregions) in which Theodore Roosevelt National Park is located. To develop the multimetric index, they collected aquatic invertebrate samples from a range of streams in western North Dakota and determined reference sites using landscape, physical habitat, and water chemistry data. After analysis, the six best metrics that represented a variety of measures (e.g., richness, habit) were chosen based on aquatic invertebrate data from these reference sites. These metrics were scaled from 0 to 100 based on the 5^{th} and 95^{th} percentiles of the data. The 25^{th} percentile was the threshold for most disturbed sites and the 5^{th}

percentile was the threshold for least disturbed sites based on reference sites. These metrics were applied to non-reference reaches to place a site in one of three categories (least disturbed, moderately disturbed, and most disturbed). To do this, the average value of the six metrics (0-100 scale) were calculated and compared to thresholds.

I used the North Dakota Macroinvertebrate Biotic Integrity Multimetric Index for the rangeland plains to understand how sites along the Little Missouri River compared to the greater region. I calculated the six metrics used by the rangeland plains index for each site along the Little Missouri River. Using the index, I scored each metric (0-100) using the same scale as the Environmental Protection Agency (2009). Finally, I calculated an average value of the six metrics at each site and compared these values to the thresholds reported.

Table 2. The equations used to calculate bioassessment metrics. A variety of metrics were calculated that included measures of richness, abundance, community diversity, pollution tolerance, habits, and functional feeding group. EPT stands for the insect orders Ephemeroptera, Plecoptera, and Trichoptera. Hilsenhoff's Biotic Index (HBI) is used to estimate average pollution tolerance of an individual in the invertebrate assemblage. All richness metrics were calculated at the lowest taxonomic level used in the present study (typically genus).

Metric	Equation	Predicted response to impact
% clingers	$\left(\frac{abundance_{clingers}}{total\ abundance}\right) \times 100$	Decrease
% EPT taxa	$= \left(\frac{richness_{EPT}}{taxa\ richness}\right) \times 100$	Decrease
% filterers	$\left(\frac{abundance_{filterers}}{total\ abundance}\right) \times 100$	Decrease
% gatherers	$\left(\frac{abundance_{gatherers}}{total\ abundance}\right) \times 100$	Decrease
% Chironomidae	$= \left(\frac{abundance_{Chironomidae}}{total\ abundance}\right) \times 100$	Increase
% predator taxa	$= \left(\frac{richness_{predators}}{taxa\ richness}\right) \times 100$	Decrease
% predators	$= \left(\frac{abundance_{predators}}{total\ abundance}\right) \times 100$	Decrease
% clingers taxa	$\left(\frac{richness_{clingers}}{taxa\ richness}\right) \times 100$	Decrease
% intolerant taxa (0-5)	$\left(\frac{richness_{tolerance0-5}}{total\ abundance}\right) \times 100$ Number of taxa with tolerance values 0 to 5.0	Decrease
% intolerant (0-5)	$\left(\frac{abundance_{tolerance0-5}}{total\ abundance}\right) \times 100$ Abundance of taxa with tolerance values 0 to 5.0	Decrease

Table 2 (continued). The equations used to calculate bioassessment metrics. A variety of metrics were calculated that included measures of richness, abundance, community diversity, pollution tolerance, habits, and functional feeding group. EPT stands for the insect orders Ephemeroptera, Plecoptera, and Trichoptera. Hilsenhoff's Biotic Index (HBI) is used to estimate average pollution tolerance of an individual in the invertebrate assemblage. All richness metrics were calculated at the lowest taxonomic level used in the present study (typically genus).

Metric	Equation	Predicted response to impact
% tolerant (6.0-7.0)	$\left(\frac{abundance_{tolerance6-7}}{total\ abundance}\right) \times 100$ Abundance of taxa with tolerance values 6.0 to 7.0	Increase
% tolerant (8.0-9.0)	$= \left(\frac{abundance_{tolerance8-9}}{total\ abundance}\right) \times 100$ Abundance of taxa with tolerance values 8.0 to 9.0	Increase
% tolerant individuals (>7)	$= \left(\frac{abundance_{tolerant>7}}{total\ abundance}\right) \times 100$ Abundance of taxa with tolerance values >7	Increase
EPT richness	Richness of mayflies, stoneflies, and caddisflies	Decrease
EPT/midge abundance	$= \frac{abundance_{EPT}}{abundance_{Chironomidae}}$	Decrease
HBI	$= \sum_{i=1}^{n} \frac{abundance_i \times tolerance_i}{total\ abundance}$	Increase
% non-insects	$= \left(\frac{abundance_{non-insects}}{total\ abundance}\right) \times 100$	Increase
% tolerant taxa (>7)	$= \left(\frac{richness_{tolerant>7}}{taxa\ richness}\right) \times 100$ Number of taxa with tolerance values >7	Increase
Taxa diversity	$= -\sum_{i=1}^{s} p_i \times \ln(p_i)$ Where p_i is the proportion of the i^{th} taxa	Decrease
Taxa evenness	$= \frac{taxa\ diversity}{\ln(taxa\ richness)}$	Decrease
Taxa richness	Number of taxa in a sample	Decrease
Total abundance	Total number of individuals (ind/m^2)	Decrease

Results

Basic water quality of the Little Missouri River was similar among sites. All sites were supersaturated with oxygen, but site #4 had the highest dissolved oxygen concentration (Table 3). The pH level was basic (>7) and reducing conditions dominated (oxidation-reduction potential <200 mV) at all sites. Site #4 had the most turbid water (shallowest Secchi disk depth). Fecal coliform concentrations all exceeded 2419.6 colony forming units (CFU)/100 mL. Fecal coliform at sites #2, #3 and #4 were analyzed first and these samples surpassed the maximum concentration measured without diluting. Samples from site #1 were measured later and diluted based on results from the other samples. Therefore, a measured concentration is only reported for site #1. *E. coli* concentrations were highest at site #4 and lowest at site #3. Water quality standards for the parameters I measured were met in the Little Missouri River, except *E. coli* exceeded the standard at site #4 (North Dakota 2001). Additionally, concentrations of fecal coliform were at least one order of magnitude higher than contact recreation standards for North Dakota (Environmental Protection Agency 2003).

Table 3. Water quality at four sites along the Little Missouri River near Theodore Roosevelt National Park. Basic water quality was measured with a Yellow Springs Instrument Professional Plus sonde (ORP = oxidation-reduction potential). Average fecal coliform and *E. coli* concentrations were measured using the Colilert method (n = 2; CFU = colony forming units). All parameters were within North Dakota water quality (WQ) standards except fecal coliform (all sites) and *E. coli* (site #4).

		Little Missouri				WQ
Parameters	**Units**	**Site #1**	**Site #2**	**Site #3**	**Site #4**	**standards**
Water temperature	°C	21.1	22.6	25.5	19.9	≤29.44
Dissolved oxygen	% saturation	101	109	108	105	
Dissolved oxygen	mg/L	8.2	8.3	8.0	8.8	≥5
Specific conductivity	µS/cm	2421	2444	2456	2617	
pH		8.49	8.41	8.44	8.49	6 to 9
ORP	mV	41.6	41.6	33.7	55	
Secchi disk depth	cm	25.5	26.0	24.0	13.5	
Fecal coliform	CFU/100 mL	20,763	>2419.6	>2419.6	>2419.6	≤200*
E. coli	CFU/100 mL	31	11.6	8.7	288	≤126*

* Only during the recreation season from 1 May through 30 September

River substrate and riparian vegetation varied among sites. Benthic substrates were composed of gravel, sand, silt, and some organic matter at site #1. The substrate at site #2 had abundant organic matter, and was mainly composed of sand and silt. Site #3 had gravel and cobble overlying sand. Finally, benthic substrate at site #4 was composed of sand and very little organic matter. I collected coal in substrate at all sites, but it was most abundant at site #4. The riparian

vegetation was primarily cottonwood at site #1 and willow at sites #2 and #4. The riparian area was dominated by cottonwood, willow, and grass at site #3.

Insects (91%) were far more abundant than non-insect taxa (8%; Table 4) in the Little Missouri River. Trichoptera (444 ind/m^2) were the most abundant order of insects followed by Diptera (381 ind/m^2), Ephemeroptera (51 ind/m^2), and Odonata (10 ind/m^2). Total invertebrate density was lowest at site #4 (167 ind/m^2 ± 28) and highest at site #3 (2247 ind/m^2 ± 1170), but differences were not significant (P = .097, F = 2.5, df = 3).

I collected 20 taxa of invertebrates from the Little Missouri River near Theodore Roosevelt National Park (Appendix A). I collected two genera of Trichoptera of which *Cheumatopsyche* (Hydropsychidae; 424 ind/m^2; Figure 2a) were far more abundant than *Nectopsyche* (Hydropsychidae; 17 ind/m^2; Figure 2b). At least four taxa of Diptera lived in the Little Missouri River of which Chironomidae were the most abundant (96%). Five genera and four families of Ephemeroptera lived in the river. *Maccaffertium* (Heptageniidae) were most abundant (14 ind/m^2) followed by *Paracloeodes* (Baetidae; 12 ind/m^2), *Heptagenia* (Heptageniidae; 10 ind/m^2), *Caenis* (Caenidae; 9 ind/m^2), and *Ephoron* (Polymitarcyidae; 6 ind/m^2; Figure 2c). *Gomphus* (Gomphidae; 5 ind/m^2) and *Dromogomphus* (Gomphidae 2 ind/m^2) were the two dragonflies collected in the Little Missouri River. I collected the family Corixidae (Hemiptera; 2 ind/m^2), but I could not identify individuals to genus because all individuals were early instars. The only beetle collected was *Dubiraphia* larvae (Elmidae; 5 ind/m^2). I collected two Crustacea taxa of which Cladocera (81 ind/m^2) were more abundant then *Hyalella* (Amphipoda; 1 ind/m^2). Finally, I collected low abundance of Hydrocarina in the river (9 ind/m^2).

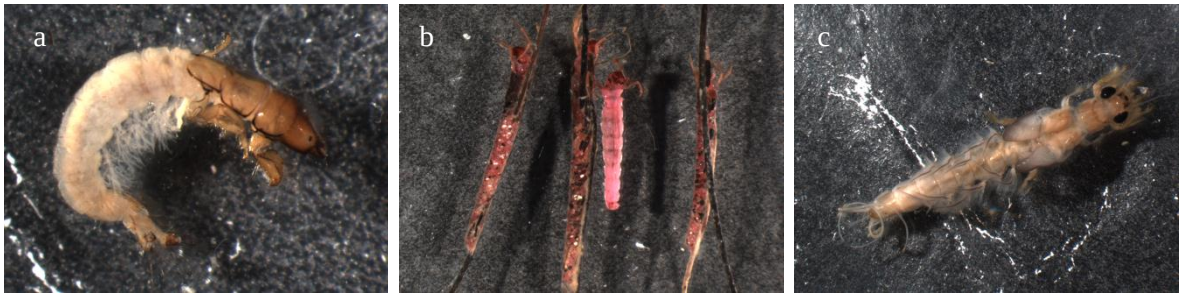

Figure 2. Photos of *Cheumatopsyche* (Hydropsychidae, Trichoptera), *Nectopsyche* (Hydropsychidae, Trichoptera), and *Ephoron* (Polymitarcyidae, Ephemeroptera) from the Little Missouri River.

Table 4. Density (ind/m^2) and standard error of invertebrates at each site. An asterisk denotes that the variable differed significantly among sites. † indicates that the variable was natural log transformed for statistical analysis because of non-normal variance.

Taxa	Site #1	Site #2	Site #3	Site #4
Hemiptera	0±0	0±0	5±5	2±2
Diptera*†	951±284	188±46	240±64	144±37
Coleoptera	7±5	12±9	0±0	0±0
Ephemeroptera*	30±14	14±23	109±30	19±13
Trichoptera	160±75	67±59	1547±920	0±0
Odonata	7±5	30±17	2±2	2±2
Plecoptera	0±0	0±0	19±11	0±0
Crustacea	2±2	0±0	326±258	0±0
Hydrocarina	30±30	5±3	0±0	0±0
Insects*†	1156±237	344±125	1921±989	167±25
Non-Insects	33±30	5±3	326±258	0±0
Total*†	1188±245	249±123	2247±1170	167±28

I calculated 22 bioassessment metrics for each sample at each site (Table 5). Of these, nine metrics detected differences according to site and three metrics detected differences among sites using multiple comparison tests. More taxa ($F = 5.6$, $df = 3$) and EPT taxa ($F = 8.4$, $df = 3$) were collected at site #3 compared to site #4 (Bonferroni multiple comparisons, $P < 0.01$). On the other hand, a higher proportion of predators lived at site #2 compared to site #1 ($F = 8.8$, $df = 3$; Bonferroni multiple comparison, $P = 0.009$). Site #4 had the lowest taxa diversity ($F = 4.6$, $df = 3$), total abundance ($F = 4.9$, $df = 3$), % predator taxa ($F = 4.4$, $df = 3$) and % EPT taxa ($F = 4.0$, $df = 3$), and the highest % Chironomidae ($F = 3.2$, $df = 3$). Finally, taxa evenness was highest at site #2 ($F = 3.5$, $df = 3$).

Several bioassessment metrics had marginal *P*-values (≤0.11) based on the ANOVA comparing sites (Table 5). The % clingers ($F = 3.1$, $df = 3$), % clinger taxa ($F = 3.1$, $df = 3$), and the % individuals with tolerance values >7 ($F = 3.6$, $df = 3$) had the lowest values at site #4. However, site #4 had the highest % gatherers ($F = 2.8$, $df = 3$). Site #3 had the highest % of individuals with tolerance values between 8.0 and 9.0 ($F = 2.4$, $df = 3$) and site #2 had the highest % of sensitive taxa (taxa with tolerance values between 0 and 5.0; $F = 2.6$, $df = 3$). Finally, site #3 was the only site where EPT abundance was higher than Chironomidae abundance (EPT/Chironomidae >1; ($F = 2.8$, $df = 3$).

Using the North Dakota Mulitmetric Index, most sites sampled along the Little Missouri River were in good condition. Sites #1 (50), 2 (50), and #3 (58) scored among the least disturbed sites, but site #4 (34) scored as a moderately disturbed site. According to the index, scores ≥38.2 are considered least disturbed, scores between 22.5 and 38.2 are considered moderately disturbed, and scores <22.5 are considered most disturbed (Environmental Protection Agency 2009).

Table 5. Average values and standard errors for bioassessment metics for each site. I used ANOVA to detect differences among sites. If values were significantly different ($P < 0.05$), I used Bonferroni multiple comparison tests to distinguish among sites. Metrics with non-normal variance were natural log (ln) or square root (√) transformed for statistical analysis.

Metric	**Site #1**	**Site #2**	**Site #3**	**Site #4**	***P*-value**	**Bonferroni**
% Chironomidae	72±13	48±8.8	32±13	79±12	0.05	
% clinger taxa	34±7.3	22±1.6	44±5.6	17±11	0.06	
% EPT taxa	44±8.7	30±5.5	60±2.8	27±11	0.03	
% gatherers	72±13	57±9.0	35±13	81±13	0.07	
% predator taxa	18±4.9	35±6.7	15±4.2	6.7±6.7	0.02	
Ln(% predators)	2.0±0.71	19±6.3	1.8±0.99	2.0±2.0	0.005	1 vs. 2
% intolerant (0-5)	3.2±1.6	16±9.5	16±7.7	14±12	0.19	
Ln(% tolerance; 6-7)	22±12	12±5.0	43±13	2.0±2.0	0.26	
% tolerance (8-9)	0±0	0±0	8.0±4.7	1.4±1.4	0.11	
Ln(% tolerant; >7)	1.8±0.67	8.0±4.1	8.1±4.7	0.87±0.87	0.06	
Ln(% tolerant taxa; >7)	16±4.6	13±5.6	14±5.4	10±10	0.12	
% intolerant taxa (0-5)	23±8.1	18±6.3	43±4.1	17±11	0.09	
EPT richness	2.2±0.37	2.0±0.55	4.2±0.73	0.60±0.24	0.0014	3 vs. 4
√EPT/Chironomidae	0.73±0.55	0.59±0.27	5.4±2.7	0.43±0.39	0.076	
HBI	6.0±0.15	5.6±0.29	6.1±0.24	5.9±0.13	0.25	
% clinger	23±13	10±1.8	51±13	3.4±2.1	0.087	
Ln(% filterers)	21±12	3.3±2.8	50±16	1.4±1.4	0.21	
% non-insects	2.2±1.9	4.5±2.9	7.0±5.0	0±0	0.44	
Taxa diversity	0.66±0.13	1.4±0.21	1.1±0.22	0.43±0.13	0.017	
Taxa evenness	0.41±0.078	0.78±0.73	0.63±0.11	0.47±0.095	0.04	
Taxa richness	5.2±0.80	6.4±0.93	7.0±1.2	2.4±0.24	0.008	3 vs. 4
Ln(total abundance)	1188±245	349±123	2247±1170	167±28	0.01	

Discussion

The peer-reviewed literature contains little information about the Little Missouri River in North Dakota. Miller and Friedman (2009) discussed floodplain formation and destruction during the last century using photographs. Personius and Eddy (1955) described the fish assemblage of the Little Missouri River before Garrison Dam was built in 1953 and flooded 64 km of the river. Kelsch (1994) studied the fish assemblage of the river after the dam was built. Finally, Beyer et al. (1995) investigated the effects of aerial grasshopper spraying on the aquatic invertebrates of the Little Missouri River. Unfortunately, Beyers et al. (1995) did not include any information on what invertebrates they collected in the river besides the order mayflies. However, the Little Missouri River has far more taxa than the Missouri River below Garrison Dam (Angradi et al. 2006). I am not aware of a published study describing the aquatic invertebrates of the Little Missouri River.

Rust (2006) collected invertebrates along the Little Missouri River at Theodore Roosevelt National Park for her thesis. Similar to the current study (5.9), she reported an average tolerance value of 5.5 for an invertebrate in the assemblage. Additionally, Rust (2006) and the current study reported similar taxa richness (6 and 5.25, respectively) and % clingers (30% and 22%, respectively). However, I found a higher % EPT taxa (40%), % Chironomidae (58%), and % gatherers (61%) than Rust (2006; 16.5%, 32%, and 40%, respectively). Also, fecal coliform concentrations reported by Rust (2006) were lower (mean = 787 CFU/100mL) than the current study. The differences in the invertebrate assemblage may be attributed to sampling methods (dip net vs. Hess sampler), date of collection, and areas sampled. Unfortunately, Rust (2006) did not describe where along the Little Missouri River or when she sampled in Theodore Roosevelt National Park, so I cannot compare more site specific information.

Aquatic invertebrates have been used to monitor water bodies since the 12th century in Europe (Cairns and Pratt 1993). Europeans developed indicator organisms to identify the level of organic pollution (i.e., sewage) that a water body was affected by (Saprobien system). For example, water bodies exclusively colonized by worms and blood midges (*Chironomus*) were considered severely polluted whereas waters colonized by mayflies and caddisflies were considered high quality. In the United States, biomonitoring began in the 1870s when Forbes studied the Illinios River. Later, biomonitoring in the United States was strongly influenced by Ruth Patrick who developed methods using abundance and richness of multiple taxonomic groups (Patrick 1949).

Currently, two types of bioassessment methods are widely used in the United States. Multivariate or predictive models use statistical models to predict expected (reference) conditions and compare these values to observed conditions (e.g., Ode et al. 2008). To make the models, biological data (e.g., aquatic invertebrates) are matched to environmental factors not thought to be affected by anthropogenic activities (e.g., channel slope, elevation) at reference sites. To estimate the level of impairment at a site, observed data are compared to model predictions based on environmental factors. The ratio of the taxa observed to the taxa expected are reported. On the other hand, multimetric indexes combine several bioassessment metrics into a single measure to estimate the level of impairment (Kerans and Karr 1994; Ode et al. 2008). Multimetric indexes are created by collecting biological samples at a range of reference sites. Based on the data from reference sites, the best metrics are chosen by eliminating correlated

metrics, using a variety of measures (e.g., richness, habit, abundance), and selecting metrics that best differentiate the data. The selected metrics are scaled (e.g., 0-100), average values of the metrics are calculated, and thresholds are developed to estimate the level of impairment. To measure the habitat quality along a reach in question, biological data are collected, metrics are calculated, and a final average score is compared to thresholds.

Multimetric indexes have been widely used to monitor biological assemblages. Multimetric indexes were first developed for fish (Karr 1981) and later developed for aquatic invertebrates (Kerans and Karr 1994). Currently, most bioassessment programs in the United States use multimetric indexes, including North Dakota. Two multimetric indexes were developed for North Dakota, because bioassessment models developed for larger areas often do not perform well (Ode et al. 2008). A multimetric index was developed for the rangeland plains of North Dakota, which includes the Little Missouri River Basin. The six metrics used in the rangeland plains index were EPT richness, % clingers (abundance), % gatherers (abundance), % predator taxa, % taxa with tolerance values of 0 to 5.0, and % of individuals with tolerance values of 8.0-9.0 (Environmental Protection Agency 2009). According to this index, sites #1, #2, and #3 located upstream of the South Unit of Theodore Roosevelt National Park were all considered least disturbed sites.

In the current study, site #1 was considered a control site because it was upstream from a water diversion for a golf course, and site #2 was downstream of the water diversion. Interestingly, both sites #1 and #2 had the same multimetric score. Therefore, the water diversion appeared to have minimal effects on the river at baseflow according to the Macroinvertebrate Biotic Integrity Multimetric Index of North Dakota. Walters and Post (2011) and Dewson et al. (2007) found that invertebrate density decreased after 40-85% of stream water was diverted. Similarly, I estimated that density decreased from 1188 ind/m^2 at site #1 to 349 ind/m^2 at site #2. Site #2 was located within a state park and the benthic substrates at this site had abundant organic matter. Given the habitat, I expected to collect more invertebrates at site #2. Additionally, Walters and Post (2011) discovered that filterers, gatherers, and scrapers were sensitive to decreases in river flow, but predators were more resistant. In the current study, the density of filterers and gatherers were much lower at site #2 (5 and 184 ind/m^2 respectively) compared to site #1 (160 and 949 ind/m^2 respectively), but scraper densities were similar. Interestingly, predator density in the Little Missouri River were higher at site #2 compared to site #1. I did not observe a change in taxa richness between sites #1 and #2, similar to the results of Walters and Post (2011). Using individual metrics, site #2 may be adversely affected by the water diversion. Allan (2004) recommends interpreting data using individual metrics to help understand the causal mechanisms operating in a river.

The water diversion between sites #1 and #2 has been active since 2002. Although the permit would allow the Theodore Roosevelt Medora Foundation to remove 493,392 m^3/year, they have removed less than half of their right on average (Table 6). Given their maximum water removal rate and average annual discharge of the river, they can remove up to 2.4% of the flow. However, their permit only allows them to remove water during the high flow season between 1 March and 1 July, thus the percentage would typically be lower on a daily basis. In addition, the Foundation is allowed to remove 493,392 m^3/year which is 0.12% of the average annual water volume of the river. Thus, the removal by the Foundation is much less than the studies described

above. Another concern from the diversion may be the fertilizer and herbicides applied to the golf course the water is irrigating.

Table 6. Water diverted from the Little Missouri River upstream of the South Unit of Theodore Roosevelt National Park by the Theodore Roosevelt Medora Foundation. Information from the North Dakota State Water Commission.

Year	Volume m^3	Rate m^3/min
2002	14,678	6.81
2003	374,978	18.93
2004	249,163	18.93
2005	303,436	18.93
2006	338,837	9.46
2007	285,674	12.11
2008	301,956	9.46
2009	144,687	12.11
2010	221,163	8.71
2011	115,330	7.57

Site #4 was considered a moderately disturbed reach according to the Macroinvertebrate Biotic Integrity Multimetric Index of North Dakota. In addition, site #4 had the lowest taxa diversity, total abundance, % predator taxa, % EPT taxa, EPT richness and taxa richness, and the highest % Chironomidae. Site #4 was located upstream from the highway 85 bridge that crossed the Little Missouri River. At first thought, the development over the Little Missouri River might have caused the impairment at site #4. For example, the river bed was disturbed when the bridge was constructed and may have altered the substrate. However, the substrate at all sites contained sand and is probably typical of the geology. Furthermore, site #3 was sampled upstream of a bridge and this site scored the highest according to the Macroinvertebrate Biotic Integrity Multimetric Index of North Dakota. I did attempt to sample farther upstream from the bridge, but steep banks prohibited me from such actions. After surveying the site, I estimated that the reach was dominated by sand and collected representative samples.

Site #4 may appear to have lower ecosystem quality, but the lower metrics may be a byproduct of sand substrate. Not only was site #4 a substantial distance downstream from the other sites,

but site #4 also differed in habitat from the other sites. The river was wider (>3x compared to other sites) and shallower at site #4. Additionally, the substrate was composed of sand and coal with very little organic matter or other structure. Lower densities and biomass of invertebrates have been found on sand compared to other substrate types (e.g., Benke et al. 1984; Bourassa and Morin 1995). Sand may have fewer invertebrates, because sand has low stability, is consistently shifting, and has little interstitial space (Wood and Armitage 1997). Furthermore, sand can have low organic matter (food) content, which may limit the secondary production of invertebrates (Soluk 1985). Yamamuro and Lamberti (2007) found that both organic matter content and predator-prey relationships were critical in understanding the assemblages in sand substrate. Chironomidae are often the most abundant invertebrates in sand (Yamamuro and Lamberti 2007), and a higher proportion of Chironomidae is often considered a sign of river impairment. Similarly, I collected the most Chironomidae at site #4, the site with the sandiest substrate. Finally, the average tolerance value of an invertebrate in the assemblage (~6) was similar among sites, further demonstrating that ecosystem quality may not have been reduced at site #4. Instead, the lower metric may have been at least partially due to sand substrate. However, further investigation of water quality and invertebrate assemblages in the North Unit of Theodore Roosevelt National Park may be warranted, especially regarding the high fecal coliform and *E. coli* concentrations.

Unfortunately, I cannot distinguish how aquatic invertebrate assemblages in the Little Missouri River varied with fecal coliform concentration, because exact concentrations were not measured at most sites. However, previous studies showed that aquatic invertebrates can respond to bacterial concentrations. For example, Chironomidae densities increased after cow manure additions to California streams (del Rosario et al. 2002). In the Little Missouri River, 58% of invertebrates were Chironomidae. Similarly, invertebrate densities and fecal coliform concentrations were positively correlated in a study investigating the effects of feral hogs on streams in Louisiana (Kaller and Kelso 2006). They found fewer mayflies, and more Tanypodinae (subfamily of Chironomidae), snails, and riffle beetles in areas with higher fecal coliform concentrations. Finally, Olive (1976) investigated the aquatic invertebrates of the Cuyahoga River in Ohio along a gradient of fecal coliform concentrations (130 – 11,000 FCU/100 mL). He sampled invertebrates and water chemistry monthly for a year. Besides fecal coliform, no other impairments were detected (e.g., trace elements, nutrients). Olive (1976) discovered that >3 times more taxa were located in reaches with lower concentrations of fecal coliform and that >50% of the taxa in these reaches were composed of intolerant taxa.

Some invertebrate bioassessment metrics were specifically designed to detect certain types of pollution. For example, Hilsenhoff's Biotic Index (HBI) was developed to detect organic pollution (Hilsenhoff 1987; 1988). Organic pollution can come from many sources (e.g., wastewater discharge, livestock grazing), but organic pollution usually decreases dissolved oxygen concentrations and increases bacterial concentrations (Simon and Buikema 1997). Organic pollution may alter the invertebrate assemblage through different pathways. For example, some invertebrates are sensitive to low dissolved oxygen concentrations, such as stoneflies. On the other hand, increased bacterial concentration may alter the food web, because aquatic invertebrates can probably eat these bacteria (Simon & Buikema 1997). After enriching a stream with cow manure, del Rosario et al. (2002) found that all invertebrates were consuming the manure, but the gathering mayfly, *Paraleptophlebia*, was enriched the most.

The source of fecal coliform in the Little Missouri River may be primarily from livestock grazing. Another source of bacteria may be from human sources (e.g., wastewater , septic tanks), because urban areas can have a disproportionately large impact on rivers (Allan 2004). However, the dominant land use in the area is livestock grazing and only small towns are located along the river. Using different management practices, such as watering livestock away from the river and managing for well-developed riparian areas, can greatly decrease bacterial concentrations. Well-developed riparian areas can buffer the effects of surrounding land use (e.g., Sponseller et al. 2001; Feld 2013). To estimate the source of bacterial contamination, the ratio of certain bacterial groups can be used. For example when the ratio of fecal coliform to fecal streptococci is between 0.1 and 0.7, the source of bacteria are likely from livestock (Muenz et al. 2006). But when the ratio is >4, bacteria are likely from human contamination.

Despite the fact that the reach near the South Unit of the Park is not meeting its designated use because of high fecal coliform concentrations, the river appears to have fairly good water quality compared to other rivers in western North Dakota (Environmental Protection Agency 2009). In contrast, the recreation designation along the Little Missouri River near the North Unit is threatened because of high fecal coliform (concentrations are near standards); however, the index scores this site as moderately disturbed compared to other rivers in western North Dakota. The park itself likely cannot change the bacterial concentrations of the water, because the source is probably distributed throughout the watershed. However, wildlife in the park may contribute to the bacterial concentrations if they frequent the river or if substantial runoff occurs after storms. Only by working together with landowners in the watershed will bacterial concentrations likely be reduced.

Literature Cited

Allan, J. D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology Evolution and Systematics 35:257-284.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. EPA 841-B-99-002, U.S. Environmental Protection Agency, Washington, D.C.

Benke, A. C., and C. E. Cushing, editors. 2004. Rivers of North America. Elsevier Academic Press, San Diego.

Benke, A. C., T. C. Van Arsdall, D. M. Gillespie, and F. K. Parrish. 1984. Invertebrate productivity in a subtropical blackwater river: the importance of habitat and life history. Ecological Monographs 54:25-63.

Bourassa, N., and A. Morin. 1995. Relationships between size structure of invertebrate assemblages, and trophy and substrate composition in streams. Journal of the North American Benthological Society 14:393-403.

Bowles, D. E., M. H. Williams, H. R. Dodd, L. W. Morrison, J. A. Hinsey, C. E. Ciak, G. A. Rowell, M. D. DeBacker, and J. L. Haack. 2008. Protocol for monitoring aquatic invertebrates of small streams in the Heartland Inventory and Monitoring Network. National Park Service, Fort Collins.

Cairns, J., and J. R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. Pages 10-27 in D. M. Rosenberg and V. H. Resh, editors. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York, NY.

North Dakota. 2001. Standards of quality for waters of the state in North Dakota Department of Health, Bismarck. Available at: http://www.ndhealth.gov/WQ/SW/Z6_WQ_Standards/B_WQ_Standards.htm

del Rosario, R. B., E. A. Betts, and V. H. Resh. 2002. Cow manure in headwater streams: tracing aquatic insect responses to organic enrichment. Journal of the North American Benthological Society 21:278-289.

Dewson, Z. S., A. B. W. James, and R. G. Death. 2007. Invertebrate community responses to experimentally reduced discharge in small streams of different water quality. Journal of the North American Benthological Society 26:754-766.

Doledec, S., N. Phillips, M. Scarsbrook, R. H. Riley, and C. R. Townsend. 2006. Comparison of structural and functional approaches to determining landuse effects on grassland stream invertebrate communities. Journal of the North American Benthological Society 25:44-60.

Eaton, A. D., and M. A. H. Franson. 2005. Standard Methods for the Examination of Water and Wastewater, 21st Centennial edition. American Public Health Association, Washington DC.

Environmental Protection Agency. 2003. Bacterial water quality standards for recreational waters (freshwater and marine waters) status report in Office of Water, U.S. Environmental Protection Agency, Washington DC.

Environmental Protection Agency. 2009. An ecological assessment of USEPA Region 8 streams and rivers. Environmental Protection Agency, Denver.

Feld, C. K. 2013. Response of three assemblages to riparian and catchment-scale land use: implications for designing catchment monitoring programmes. Freshwater Biology 58.

Fitzpatrick, F. A., B. C. Scudder, B. N. Lenz, and D. J. Sullivan. 2001. Effects of multi-scale environmental characteristics on agricultural stream biota in eastern Wisconsin. Journal of the American Water Resources Association 37:1489-1507.

Gammon, J. R., W. C. Faatz, and T. P. Simon. 2002. Patterns in water quality and fish assemblages in three central Indiana streams with emphasis on animal feed lot operations. Pages 373-417 in Biological Response Signatures: Indicator Patterns Using Aquatic Communities.

Hall, R. O., Jr., J. L. Tank, D. J. Sobota, P. J. Mulholland, J. M. O'Brien, W. K. Dodds, J. R. Webster, H. M. Valett, G. C. Poole, B. J. Peterson, J. L. Meyer, W. H. McDowell, S. L. Johnson, S. K. Hamilton, N. B. Grimm, S. V. Gregory, C. N. Dahm, L. W. Cooper, L. R. Ashkenas, S. M. Thomas, R. W. Sheibley, J. D. Potter, B. R. Niederlehner, L. T. Johnson, A. M. Helton, C. M. Crenshaw, A. J. Burgin, M. J. Bernot, J. J. Beaulieu, and C. P. Arango. 2009. Nitrate removal in stream ecosystems measured by N15 addition experiments: total uptake. Limnology and Oceanography 54:653-665.

Hilsenhoff, W. L. 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist 20:31-39.

Hilsenhoff, W. L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society 7:65-68.

Johnson, R. K., M. T. Furse, D. Hering, and L. Sandin. 2007. Ecological relationships between stream communities and spatial scale: implications for designing catchment-level monitoring programmes. Freshwater Biology 52:939-958.

Kaller, M. D., and W. E. Kelso. 2006. Swine activity alters invertebrate and microbial communities in a coastal plain watershed. American Midland Naturalist 156:163-177.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6:21-27.

Kelsch, S. W. 1994. Lotic fish-community structure following transition from severe drought to high discharge. Journal of Freshwater Ecology 9:331-341.

Kerans, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications 4:768-785.

Merritt, R. W., K. W. Cummins, and M. B. Berg, editors. 2008. An Introduction to the Aquatic Insects of North America, 4th edition. Kendall Hunt Publishing, Dubuque, IA.

Muenz, T. K., S. W. Golladay, G. Vellidis, and L. L. Smith. 2006. Stream buffer effectiveness in an agriculturally influenced area, southwestern Georgia: responses of water quality, macroinvertebrates, and amphibians. Journal of Environmental Quality 35:1924-1938.

Needham, J. G., M. J. Westfall, and M. L. May. 2000. Dragonflies of North America, Revised edition. Scientific Publishers, Gainesville, Florida.

Ode, P. R., C. P. Hawkins, and R. D. Mazor. 2008. Comparability of biological assessments derived from predictive models and multimetric indices of increasing geographic scope. Journal of the North American Benthological Society 27:967-985.

Olive, J. H. 1976. Chemical, physical, and biological assessment of water quality in the Cuyahoga River, Ohio USA 1973-1974. Ohio Journal of Science 76:5-15.

Patrick, R. 1949. A proposed biological measure of stream conditions, based on a survey of the Conestoga Basin, Lancaster County, Pennsylvania. Proceedings of the Academy of Natural Sciences of Philadelphia 101:277-341.

Paulsen, S. G., A. Mayio, D. V. Peck, J. L. Stoddard, E. Tarquinio, S. M. Holdsworth, J. Van Sickle, L. L. Yuan, C. P. Hawkins, A. T. Herlihy, P. R. Kaufmann, M. T. Barbour, D. P. Larsen, and A. R. Olsen. 2008. Condition of stream ecosystems in the US: an overview of the first national assessment. Journal of the North American Benthological Society 27:812-821.

Resh, V. H., and J. K. Jackson. 1993. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. Pages 195-223 in D. M. Rosenberg and V. H. Resh, editors. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman & Hall, New York.

Rust, J. 2006. Establishing baseline data for aquatic resources in National Parks of the Northern Great Plains Network. Thesis. South Dakota State University.

Sarai, D. S. 1976. Total and fecal coliform bacteria in some aquatic and other insects. Environmental Entomology 5:365-367.

Saunders, W. C., and K. D. Fausch. 2007. Improved grazing management increases terrestrial invertebrate inputs that feed trout in Wyoming rangeland streams. Transactions of the American Fisheries Society 136:1216-1230.

Saunders, W. C., and K. D. Fausch. 2012. Grazing management influences the subsidy of terrestrial prey to trout in central Rocky Mountain streams (USA). Freshwater Biology 57:1512-1529.

Simon, K. S., and A. L. Buikema. 1997. Effects of organic pollution on an Appalachian cave: changes in macroinvertebrate populations and food supplies. American Midland Naturalist 138:387-401.

Smith, D. G. 2001. Pennak's Freshwater Invertebrates of the United States, 4th edition. John Wiley and Sons, Inc., New York.

Soluk, D. A. 1985. Macroinvertebrate abundance and production of psammophilous Chironomidae in shifting sand areas of a lowland river. Canadian Journal of Fisheries and Aquatic Sciences 42:1296-1302.

Sponseller, R. A., E. F. Benfield, and H. M. Valett. 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology 46:1409-1424.

Thorp, J. H., and A. P. Covich, editors. 2010. Ecology and Classification of North American Freshwater Invertebrates, 3rd edition. Elsevier, New York.

Vandewalle, M., F. de Bello, M. P. Berg, T. Bolger, S. Doledec, F. Dubs, C. K. Feld, R. Harrington, P. A. Harrison, S. Lavorel, P. M. da Silva, M. Moretti, J. Niemela, P. Santos, T. Sattler, J. P. Sousa, M. T. Sykes, A. J. Vanbergen, and B. A. Woodcock. 2010. Functional traits as indicators of biodiversity response to land use changes across ecosystems and organisms. Biodiversity and Conservation 19:2921-2947.

Walters, A. W., and D. M. Post. 2011. How low can you go? Impacts of a low-flow disturbance on aquatic insect communities. Ecological Applications 21:163-174.

Wood, P. J., and P. D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. Environmental Management 21:203-217.

Yamamuro, A. M., and G. A. Lamberti. 2007. Influence of organic matter on invertebrate colonization of sand substrata in a northern Michigan stream. Journal of the North American Benthological Society 26:244-252.

Appendix A

List of invertebrate taxa collected in the Little Missouri River near Theodore Roosevelt National Park.

- Insects
  - Coleoptera (Beetles)
    - Elmidae (Riffle beetles)
      - *Dubiraphia*
  - Diptera (Trueflies)
    - Chironomidae (Non-biting midge)
      - Non-Tanypodinae
      - Tanypodinae
    - Ceratopogonidae (No-see-ums)
      - *Probezzia*
    - Simuliidae (Blackflies)
      - *Simulium*
  - Ephemeroptera (Mayflies)
    - Caenidae (Small squarefill mayflies)
      - *Caenis*
    - Heptageniidae (Flat-headed mayflies)
      - *Maccaffertium*
      - *Heptagenia*
    - Baetidae (Small minnow mayflies)
      - *Paracloeodes*
    - Polymitarcyidae (Pale burrowing mayflies)
      - *Ephoron*
  - Hemiptera (True bugs)
    - Corixidae (Water boatmen)
  - Plecoptera (Stoneflies)
    - Perlidae (Common stoneflies)
      - *Acroneuria*
    - Nemouridae (Brown stoneflies)
  - Odonata (Dragonflies and Damselflies)
    - Gomphidae (Clubtails)
      - *Dromogomphus*
      - *Gomphus*
  - Trichoptera (Caddisflies)
    - Hydropsychidae (Net-spinning caddisflies)
      - *Cheumatopsyche*
    - Leptoceridae (Long-horned caddisflies)
      - *Nectopsyche*
- Arachnida
  - Hydracarina (Water mites)
- Crustacea
  - Amphipoda (Scuds)
    - *Hyalella*
  - Cladocera (Water fleas)

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

NPS 387/121725, July 2013

National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science
1201 Oakridge Drive, Suite 150
Fort Collins, CO 80525

www.nature.nps.gov