BIOLOGICAL TOOLS FOR WATER SECURITY IN THE NORTHERN GREAT PLAINS

A Thesis Submitted to the College of Graduate and Postdoctoral Studies

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ABSTRACT

Rivers and streams are a critical component of Northern Great Plains ecosystems and an extremely valuable source of surface water. Despite their importance, these waterbodies have traditionally received much less scientific attention than forested regions. This thesis provides insight into what best available natural communities may still exist, and how alterations in the physical characteristics of rivers and streams can push communities outside of their range of natural variation.

Specifically, I show that since the construction of a hydroelectric dam on a large Northern Great Plains river densities of benthic macroinvertebrates have increased significantly through time in the river downstream relative to reference sites and pre-dam conditions. My findings indicate that although there is a notable loss in sensitive taxa such as mayflies and stoneflies, other midge taxa colonize this unique, cold water habitat and create a new community of benthic macroinvertebrates. Further, reaches downstream of the dam are significantly cooler than reference through the summer into August and do not reach the temperature optima of reference reaches. I use these results to develop a reference condition model that assesses current condition and can be used to monitor recovery through mitigation of these perturbations. My results have implications primarily for understanding and quantifying the ecosystem impacts of hydroelectric energy production, but also range expansion of cold-water tolerant taxa, the life-history of select groups of invertebrates, and ultimately the forage resources available to the fish assemblages of this river system.

Further, as Northern Great Plains rivers are typified by considerable flow variability, particularly in the presence of water control structures, fine sediment (<63μm) is readily suspended, especially during periods of high discharge. Assessment of the impacts to biota by anthropogenic stressors must therefore occur within the context of dynamic turbidity and background flow conditions. I developed a model in which discharge is a principal determinant of in-stream suspended sediment. This relationship was explored with a case study showing that macroinvertebrate community structure is strongly correlated with suspended sediment gradients and ultimately predicted by discharge. Factors affecting sediment loads and ecosystem responses in managed systems should be considered so that in-stream water quantity and quality needs are met. This new understanding should allow for the development of improved ecosystem based flow management objectives.

Finally, I develop a multivariate and predictive model based on the reference condition approach for the Northern Great Plains region of Saskatchewan from benthic macroinvertebrate communities and environmental abiotic data collected at 280 reference sites and 10 test sites. Reference sites were classified into groups characterized by similar macroinvertebrate communities. This model predicted 68.7% of the sites correctly using cross-validation. Of the 10 test sites, two were stressed (one waste water and one urban site) while three were classified as impaired (one waste water and two reservoirs).

These models are effective tools that provide a practical means of evaluating biotic condition of streams in the Northern Great Plains.

ACKNOWLEDGEMENTS

Thanks Douglas Chivers, for taking me on as a student and giving me the liberty to pursue, and guidance through, the scientific agenda I brought to the lab. During the years I built this thesis I had the privilege of learning with many talented young thinkers, and our inquiries coalesced into the Troutreach program of the Wildlife Federation. I am grateful in particular to the counsel and discussion of Aaron Bell, Stephen Srayko, Dale Parker, Michelle Bowman, Michael Cavallaro, Matt Wooton, Michael Pollock and John-Mark Davies.

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FORMAT OF THE THESIS

This thesis has been organized as a manuscript-style thesis. As a result, there is some repetition of information among the chapters. I am the senior author on each published work, and as such was the person primarily responsible for designing and carrying out the work. This includes the data collection, analysis and writing.

Chapter 2 was published in Fundamental and Applied Limnology

Phillips, I.D., M.S. Pollock, and D.P Chivers. 2016. Benthic communities through the construction of a major reservoir and 40 years of change. Fundamental and Applied Limnology. 188(4): 279-288.

Chapter 3 was published in Journal of Great Lakes Research

Phillips, I.D., M.S. Pollock, M.F. Bowman, D.P. Chivers. 2015. Thermal alteration and macroinvertebrate response below a large Northern Great Plains reservoir. Journal of Great Lakes Research. 41:155-163.

Chapter 4 was published in Freshwater Science

Phillips, I.D., J-M Davies, M.F. Bowman, and D.P. Chivers. 2016. Macroinvertebrate communities in a Northern Great Plains River are strongly shaped by naturally occurring suspended sediments: implications for ecosystem health assessment. Freshwater Science. 35: 1354-1364.

Chapter 5 is currently an unpublished manuscript.

CHAPTER 1: INTRODUCTION – ECOSYSTEM HEALTH MONITORING AND ASSESSMENT IN NORTHERN GREAT PLAINS USING BENTHIC MACROINVERTEBRATES

Setting objectives for sustainable management of aquatic resources requires a system for identifying high quality aquatic communities, and understanding the underlying physical characteristics that support them. Without an understanding of the latter, expectations of what aquatic systems and communities should be present in the absence of human activity is challenging at best and may even be impossible. Fundamentally, environmental assessment of human impact requires a sound understanding of what conditions should exist independent of any anthropogenic perturbation. These baselines or controls are categorized commonly as reference condition in biological monitoring, and the method of evaluating condition against reference condition has been labelled as the Reference Condition Approach (RCA; Wright 1995).

Using the RCA to evaluate condition in aquatic systems originally developed approximately 25 years ago with the River Invertebrate Prediction and Classification System (RIVPACS) in the United Kingdom (Wright 1995). Since its inception, the RCA has been adopted in jurisdictions across the globe such as the Australian adaptation of RIVPACS labelled the Australian River Assessment System (AUSRIVAS; Nichols et al. 2010), and even here in Canada with the Canadian Aquatic Biomonitoring Network (CABIN; Reynoldson et al. 2001). The similarity between all these approaches is the underlying approach of using reference site environmental descriptors to discriminate between community groupings. This then allows assignment of group membership to sites that are thought to have been impacted by human activity into appropriate reference groups based on the site's underlying environmental descriptors (Bailey et al. 2004).

A fundamental challenge to the strength of RCA, or any control baseline in environmental management for that matter, is the question to what quantified standard does a study draw its quantified reference data from? Reference condition has many interpretations across a range of contexts, and is likely best described for the intended purposes of the biological assessment in a specific study (Stoddard et al. 2006). Stoddard et al. (2006) attempted to establish essentially a "terms of reference" when discussing what reference condition a study is referring to when it attempts an RCA. Specifically, these authors define reference condition as historical condition, least disturbed condition, minimally disturbed condition, and best attainable condition.

As the name infers, in the Northern Great Plains, the historical condition would refer to the aquatic communities that existed prior to European colonization in the late 19th century. Least disturbed condition, in contrast, accepts a degree of human activity across the landscape; however, selection of this condition as a benchmark seeks the best available biological, chemical and physical characteristics considering the highly developed state of a landscape. Minimally disturbed condition requires that the sites being held as controls only have minor levels of human activity. The distinction between least disturbed condition and minimally disturbed condition is that the latter shows very minimal amounts of anthropogenic perturbation whereas least disturbed condition can still occur in an area with extensive disturbance.

Best attainable condition implicitly accepts a highly developed landscape with little possibility of removing human activity. Therefore, best attainable is the condition that may differ from what an un-impacted environment may possess, but given mitigation, rehabilitation, and best management practices, for example, it is the condition human intervention can improve an aquatic system to resemble.

Aquatic macroinvertebrate communities change (species composition, abundance of individuals, and species diversity) depending on the type and amount of pollution present in the system (Hilsenhoff 1988). Measurable, predictable change enables development of ecological health indices. Other jurisdictions (e.g., United States, Britain, European Union, Australia) have developed macroinvertebrate aquatic health measures that together with water chemistry, are used to establish and monitor surface water quality objectives. However, macroinvertebrate measures for the Northern Great Plains (NGP) have been more difficult to develop because the natural extremes (e.g., winter, drought, landscape disturbance, high nutrient and productivity) of this region result in a community characterized by taxa already more tolerant to abiotic extremes than those found elsewhere in North America (Fig. 1.1). Therefore macroinvertebrates in the NGP may be insensitive to pollution relative to other regions. As such, feasibility of an aquatic health measure using NGP macroinvertebrates requires investigation before application of existing tools from other jurisdictions.

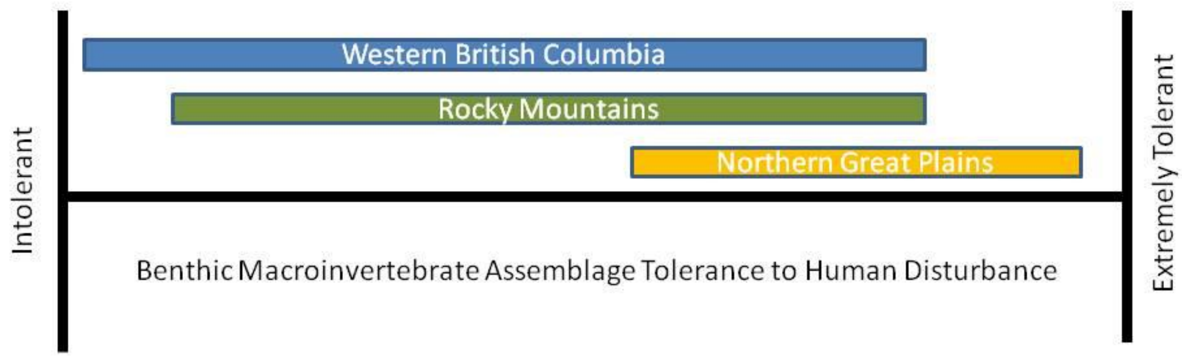

Figure 1.1. Conceptual comparison of benthic macroinvertebrate tolerances between the Northern Great Plains (NGP) and more stable environments of British Columbia.

Most benthic macroinvertebrate-based biomonitoring programs employ a collection of metrics (here used to describe both metrics and indexes) characterizing the structure and function of the assemblage, and condition of the waterbody. In particular, metrics such as species richness and evenness express the biodiversity of the assemblage, whereas % filterers characterize the community's functional capacity (i.e. biofiltering). The Hilsenhoff Biotic Index (HBI) assigns known organic pollution tolerance scores to all taxa detected at a site, and then assesses the tolerance of the assemblage based on the total score (Hilsenhoff 1988). However, metrics must be validated for particular regions and stressors to account for variability across ecoregions.

Goals of the present research

The current research was carried out with the intent of developing better understanding of basic ecological variability in Northern Great Plains rivers and streams, and to develop tools for setting site-specific objectives of ecosystem health.

The specific goals addressed in this study are as follows:

1) Evaluate benthic macroinvertebrate community changes in a large Northern Great Plains river due to hydroelectric reservoir construction. I use historical community data pre-dating the construction of a large, Northern Great Plains reservoir in reference reaches of the river system and the river downstream of the dam, as well as repeated sampling 20 and 40 years after its construction to determine what the communities resembled prior to construction and how they may have changed through time.
2) Design an ecosystem health assessment tool based on hydroelectric community impacts to set site-specific objectives and evaluate biological condition. Large hydroelectric dams directly alter the abiotic condition of rivers by releasing water with large differences in temperature relative to natural conditions. These alterations in the thermal regime along with alterations in flow often result in altered ecosystems downstream. I will assess changes in temperature and benthic communities downstream of the hydroelectric reservoir above, and develop a Test Site Analysis-based model for evaluating impact and to monitor recovery.
3) Construct a model characterizing underlying constraints on benthic macroinvertebrate community structure in regulated medium-sized rivers and streams. For rivers that are typified by considerable flow variability, fine sediment is readily suspended during periods of high discharge. Assessment of the impacts to biota by anthropogenic stressors must therefore occur within the context of dynamic turbidity and background flow conditions.
4) Develop a multivariate and predictive model for ecosystem health based on the reference condition approach for the Northern Great Plains region. This model will be based on

community classification at least-impacted rivers and streams, and using discrimination analysis it will allow me to establish what underlying abiotic characteristics define site-specific expectations for communities at disturbed waterbodies.

Although basic underlying physical influences on the natural variability of benthic macroinvertebrate communities are better understood elsewhere in the world, this is poorly understood in the Northern Great Plains. Further, how anthropogenic perturbation may force deviation outside this natural variation is even less well studied and the following chapters have been organized to address these challenges. Each chapter is a stand-alone contribution that has been published or submitted to the primary literature.

CHAPTER 2: BENTHIC COMMUNITIES THROUGH THE CONSTRUCTION OF A MAJOR RESERVOIR AND 40 YEARS OF CHANGE

Produced in:

Phillips, I.D., M.S. Pollock, and D.P Chivers. 2016. Benthic communities through the construction of a major reservoir and 40 years of change. *Fundamental and Applied Limnology*. 188(4): 279-288.

Abstract: Large hydroelectric dams have proliferated across the world through the 20th century. The resulting reservoirs can impart large changes to the abiotic and biotic environments upstream and downstream through changes in thermal regime of the rivers. Most understanding of the impacts these dams have on riverine communities is derived from studies conducted long after dams are constructed, with little information on what the communities resembled prior to construction and how they may have changed through time. In the current study we provide historical community data pre-dating the construction of a large, Northern Great Plains reservoir in reference reaches of the river system and the river downstream of the dam, as well as repeated sampling 20 and 40 years after its construction. In exploring changes in the community through time we find that densities of benthic macroinvertebrates have increased significantly through time relative to reference sites and pre-dam habitats. Further, these communities have changed through time to be different downstream of the reservoir in both soft and hard sediment. Our findings indicate that although there is a notable loss in sensitive taxa such as mayflies and stoneflies, other midge taxa colonize this unique, cold water habitat and create a new community of benthic macroinvertebrates.

Introduction

Hydroelectric dams have been developed across the globe often with the intention of producing a renewable and low-carbon emitting source of power, yet the reservoirs created by such impoundments dramatically change upstream and downstream environments (Olden & Naiman 2010). Aquatic biodiversity in particular is affected by the hydrologic alteration (Bunn & Arthington 2002) and modified thermal regimes (Ward 1985). Freshwater fish and insects require species-specific accumulation of thermal units above a species-specific daily temperature threshold (Olden & Naiman 2010), and the depression of instream temperatures can result in inadequate annual temperatures for the growth of pre-existing taxa leading to community change (Lehmkuhl 1972, Phillips et al. 2015). Fish and insects too have acute and chronic thresholds for surviving and thriving, and altering natural temperature regimes can have more immediate short-term effects than longer term generation growth (Vannote & Sweeney 1980, Coutant 1987). In addition to mere temperature influences on the metabolism, growth and diapause of aquatic organisms, depressed temperatures also impact aspects of water quality such as nutrient concentrations, organic matter, dissolved oxygen and solute fluxes (Webb 1996, Caissie 2006) having indirect structuring effects on aquatic communities.

Despite the growing evidence and understanding of the altered ecosystems downstream of dams (e.g., Takao et al. 2008, Phillips et al. 2015), few studies have followed how communities changed prior to dam construction through subsequent decades. As Vinson (2001) identifies, the prevailing knowledge of the ecosystem-altering effects of dams is limited to studies covering 1-3 years, despite a lifespan exceeding 100 years. Thus, long-term planning is difficult (Petts 1980). Therefore, studying community change through time provides valuable insight into instream flow ecosystem health expectations as ecosystems are dynamic and new environments such as dam tailwaters require time to be colonized by organisms tolerant to altered conditions.

The South Saskatchewan River (SSR) is one of the largest rivers in North America (Martz et al. 2007), and has recently been identified as one of the most stressed major systems in Canada (Swainson 2009). Instream flow volumes have been substantially reduced and natural flow patterns altered through damming, water abstraction and climate warming (Schindler & Donahue 2006). Built in 1967, the Gardiner Dam and associated Coteau Creek Power Station is often seen as a point-source of these anthropogenic perturbations (Swainson 2009) where the effect of flow fluctuation and low water temperature (resulting from hydropeaking and hypolimnetic release respectively) is potentially problematic for benthic fauna downstream (Phillips et al. 2015).

Most information on the impact of the Gardiner Dam on benthic invertebrates is limited to particular taxa. The pioneering work of Lehmkuhl (1972) focused early attention on the impacts of this dam on mayfly populations, identifying thermal-stress related impacts on the life histories of 15 species. Lehmkuhl (1972) suggested the specific reason these species were absent below the dam was improper temperature sequence for breaking diapause, and/or inadequate degree-days needed for growth and adult emergence. Fredeen (1981) reported results of a blackfly control program through the mid-20th century and anecdotally noted the conspicuous absence of 3 species (*Simulium* [*Psilopelmia*] *bivittatum* Malloch, *S.* [*Psilopelmia*] *griseum* Coquillett, and *S.* [*Gnus*] *arcticum* Malloch) in the area downstream of Gardiner Dam after construction of the hydroelectric project. Fredeen (1981) attributed these absences to a lack of egg and larval drift with impoundment. Of even greater concern for conservation in this system may be the loss of taxa too poorly studied to appreciate the impacts of thermal pollution. For example, the South Saskatchewan River is one of only two rivers known to host the rare mayfly *Acanthomola pubescens* (Whiting & Lehmkuhl 1987), represented by a holotype specimen from Lemsford Ferry (51º 1′ 27″ N, 109º 7′ 59″ W). However, impacts of the Gardiner Dam on the entire benthic community are incompletely known, and no study has compared current condition to historical communities prior to dam construction.

In our study, historical records were used to test the hypothesis that there have been changes in the density and community composition of benthic macroinvertebrates through time downstream of Gardiner Dam relative to reference sites. The specific hypothesis is that Trichoptera and Ephemeroptera will decrease in density, while more tolerant Diptera and Oligochaeta will increase in density. Because previous work has suggested that the current cold-water temperatures downstream of Gardiner Dam prevent release from diapause in Trichoptera and Ephemeroptera, I hypothesize that they will have demonstrated a decrease over time. Further, the high tolerances of Chironomidae and Oligochaeta, as well as their prevalence below other reservoirs in Saskatchewan today (e.g., E.B. Campbell Dam [Mason 1983]) suggest that they may increase due to reservoir construction. Examining effects at two different time points following the construction of the dam provides us with an indication of the speed of community change that may occur.

Methods

Study area

The Saskatchewan River system begins in the Rocky Mountains in Alberta to the west, and discharges into the Lake Winnipeg in the boreal forest to the east. Within the province of Saskatchewan, the river system is divided into the North and South Saskatchewan Rivers, for the majority of its length before converging downstream of the city of Prince Albert (Fig. 2.1). The mean discharge of the SSR at Saskatoon in 2008 was 213 $m^3 \bullet s^{-1}$, and of the NSR at Prince Albert in 2008 was 222 $m^3 \bullet s^{-1}$ (http://www.wsc.ec.gc.ca/applications/H2O/index-eng.cfm). The two branches mostly flow through the Prairie Ecozone but they transition to the Boreal Plains Ecozone near their confluence. Prairie taxa typify the fauna with some community overlap with the minor boreal rivers that flow into the mainstem river in its Boreal Plains Ecozone (Miyazaki & Lehmkuhl 2011). See Miyazaki & Lehmkuhl (2011) for a detailed description of the Saskatchewan River system and its aquatic insect fauna.

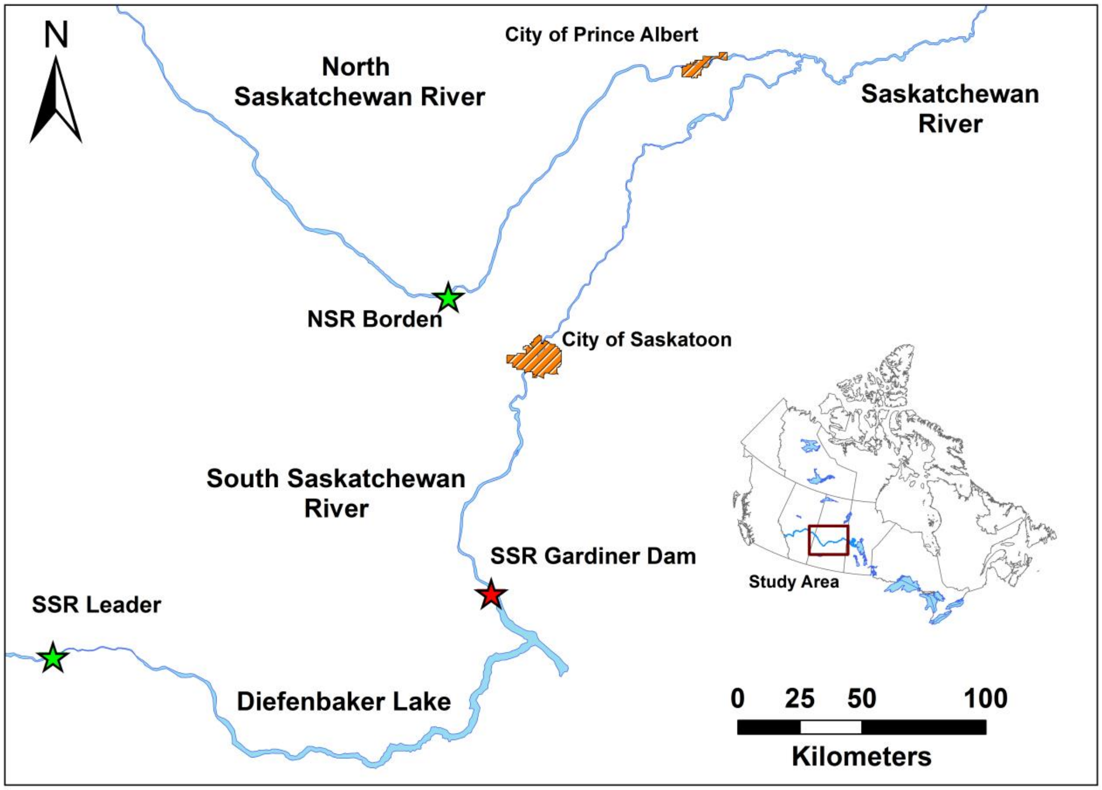

Figure 2.1. Map of the Saskatchewan River system with reference sites (green stars) and dam site (red star).

Sample collection

In August 2008 benthic macroinvertebrates were collected at each of three stations on the river using sampling techniques specific to the habitat and organisms sought; however, sampling effort was consistent across the study (Fig. 2.1). In particular, five benthic grab samples using a Peterson Dredge (base = ~0.022 m^2) were collected from soft sediment at each station. Each sample consisted of three grabs integrated into a single sample. Further, five Hess samples (base

= ~0.086 m^2) were collected from five separate riffles in each station. All benthic samples were concentrated in a 500 µm mesh net before preservation. Mean densities of individual orders in each station were estimated separately for soft sediment and riffle habitats by averaging the five Peterson dredge and five Hess samples, respectively.

Sample preservation and processing

Organisms were stored in 80% ethanol and returned to the laboratory where they were sorted under 7 X magnification. Benthic macroinvertebrates were identified to lowest possible designation using keys for North America (Merritt & Cummins 1996) and Western Canada (Dosdall & Lehmkuhl 1979, Brooks & Kelton 1967, Clifford 1991, Larson et al. 2000, Webb 2002, Webb et al. 2004). Additional taxa-specific keys and literature were used for accurate identification of the unique midge fauna (Diptera: Chironomidae) occurring in the SSR system (Hirvenoja 1973, Oliver 1976, Oliver & Roussel 1983). Voucher series were deposited in both the Water Security Agency of Saskatchewan Invertebrate Voucher Collection (Saskatoon, Saskatchewan), and the Royal Saskatchewan Museum (Regina, Saskatchewan). Further, all taxa occurrence records were submitted to the Saskatchewan Conservation Data Centre with the Ministry of Environment (http://www.biodiversity.sk.ca/).

Historical comparisons

A literature search was conducted of limnological and fisheries research on the Saskatchewan River system to obtain benthic macroinvertebrate data pre-dating the construction of the Gardiner Dam (1957-58 [Reed 1959]), and 20 years post-dam construction (1987-88 [Merkowski

1987 for the NSR, and Miles & Sawchyn 1988 for the SSR]). Further, the Saskatchewan Aquatic Macroinvertebrate Database (AquaTax 2009, Saskatoon, Saskatchewan) was accessed for added evidence of invertebrate order presence or absence downstream of the Gardiner Dam. These were the only sources of historical data identified in the literature search. These studies report benthic macroinvertebrate densities; however, because each study reported differing resolutions for its taxonomic identifications it was necessary to standardize our community analyses to a common level (order). The earlier studies collected benthic macroinvertebrates using Surber samplers (base = ~0.093 m^2) for riffle habitat and Ekman grabs (base = ~0.025 m^2) for soft sediment (earlier studies also used 500 μm mesh nets) thus requiring surface area standardization amongst all four methods (Surber, Ekman, Peterson and Hess samplers). Previous studies comparing Eckman and Peterson grabs (Lewis et al. 1982) and Surber and Hess samplers (Taylor et al. 2001) have found they do not produce significantly different estimates of benthic macroinvertebrate densities and we assume they can be compared in the current study.

Temperature records do not exist at the Gardiner Dam (SSR Gardiner) pre-dating dam construction; however, they do exist for reaches overlapping the SSR at Leader (SSR Leader) and the NSR at Borden (NSR Borden; Reed 1959). We make the assumption there is no significant influence on this annual regime between the SSR Gardiner location and the reaches at the SSR Leader area immediately upstream since they are both at roughly the same latitude, there would have been no point sources of thermal pollution in this area at the time, and it is a relatively small distance for such a large river. As such, we used these data to shed-light on the pre-dam temperature conditions.

Statistical analyses

Data for the current analysis was standardized to order because historical data was only identified to this level. However, identifying 2008 samples to the lowest possible taxonomic designation was performed in order to suggest hypotheses explaining why changes may have occurred through time.

We used a non-parametric two-way ANOVA test through the Minitab statistical software (version 17; Minitab Inc., Pennsylvania, USA) to compare changes in benthic macroinvertebrate density through time (10 y Pre-Dam, 20 y Post-Dam, and 40 y Post-Dam) and across sites (SSR1 Leader [control], SSR2 Gardiner Dam [test], and NSR1 Borden [control]). The collections in historical records were made in similar periods during the ice-free season, in silt-sand substrates for grab samples, in cobble stretches for Surber/Hess samplers, and all samples were standardized to area sampled (e.g., • m^{-2}).

Next, we compared the macroinvertebrate community that existed pre-dam construction to that 20 and 40 years after construction at the same sites as the historical comparisons of benthic density above. However, we were unable to make community inferences of how the structure of soft sediment has changed since before the dam because the sampling in 1957 (Reed 1959) produced zero benthic invertebrates from soft sediment at the SSR Gardiner location.

We used Non-Metric Multidimensional Scaling (NMDS, Clarke & Warwick 2001) to examine benthic community structure. Stress, optimal distance linking metrics (e.g., Mahalanobis, Bray-Curtis, Gowers, and Kulczynski distance metrics), optimal numbers of dimensions prior to analysis, and selected the distance linking metric were calculated based on the highest non-metric fit r^2. The Bray-Curtis distance metric was the optimal distance linking metric for both soft (r^2=0.97) and hard (r^2=0.97) data matrices. A multidimensional solution was considered optimal and with a greater representation of the community structure if it had a value < 0.10, yet acceptable if it had a final stress < 0.18 (Clarke & Warwick 2001). Finally, the optimal number of dimensions to apply were chosen by including subsequent dimensions until a solution was arrived at that explained > 85% of the variance. The *vegan* and *MetaMDS* packages for R, version 2.15.1 (R Development Core Team 2013) was used for NMDS.

Analysis of Similarities (ANOSIM) was applied to evaluate whether groupings revealed in the NMDS were significant using PRIMER Version 6.1.13 (PRIMER-E software, Plymouth, United Kingdom; Clarke & Warwick, 2001). ANOSIM *R* values were considered significant if $p < 0.05$.

Results

A lower mean maximum temperature was observed in the current study than in the series provided by Reed (1959) for 1957 and 1958. Specifically, Reed (1959) recorded a mean ± 1SD of 22.2 ± 1.3 ºC in 1957 and 20.9 ± 1.2 ºC in 1958 for NSR Borden as compared with 19.0 ± 1.6 in 2008. Comparably, Reed (1959) recorded a mean ± 1SD of 22.3 ± 1.4 ºC in 1957 and 20.8 ± 1.1 ºC in 1958 for SSR Leader as compared with 20.9 ± 1.5 in 2008. This difference is likely a product of inter-annual variation and sampling frequency because Reed's (1959) 1958 measurements were closer to those in 2008 (see Table 1 in Reed 1959).

In total, this study collected 7,751 individuals, representing 15 taxa identified at order or coarser taxonomic designation (Table 2.1; see Supplementary Data for detailed 2008 taxa lists (https://www.schweizerbart.de/papers/fal/detail/188/87124/Benthic_communities_through_the_construction_of_a_major_reservoir_and_40_years_of_change?af=search). Two way ANOVA using ranked data indicated a significant interaction between time and site (F = 14.74, df = 4, p = <0.001). A series of ranked one-way ANOVAs were conducted to isolate the treatment effects (note a ranked one-way ANOVA was used in lieu of a Kruskal-Wallis test to maintain consistency in data analysis). Ranked one-way ANOVAs indicate no change in macroinvertebrate density through time at the NSR Borden site (F = 3.70, df = 2, p = 0.053) or the SSR Leader site (F = 2.66, df = 2, p = 0.11). There was, however, a significant increase in density at SSR Gardiner post dam (F = 22.66, df = 2, p = <0.001), with the highest density being reported 40 years after dam construction (Tukey Post Hoc test $p < 0.05$; Fig. 2.2).

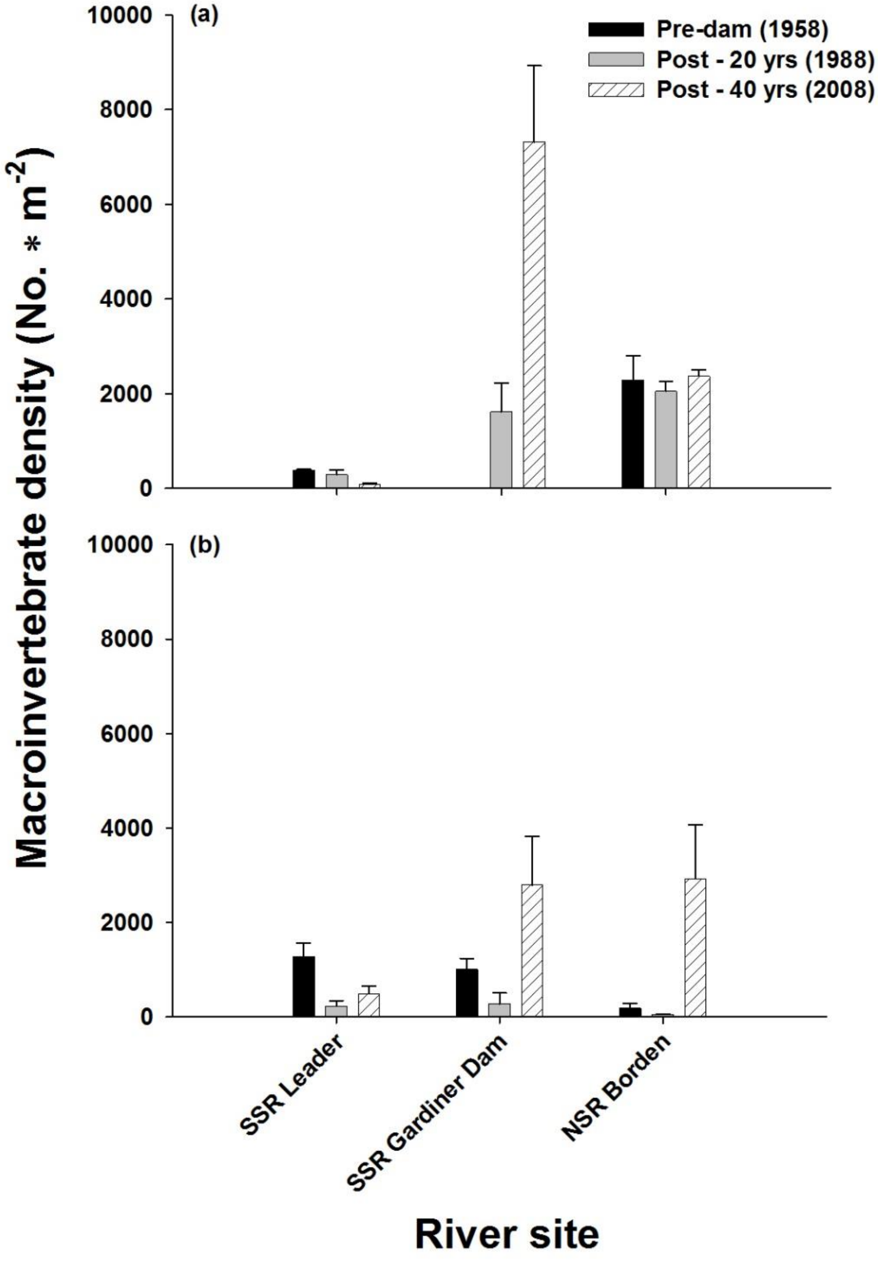

Figure 2.2. Comparison of mean densities of benthic macroinvertebrate densities (± 1 SD) between pre, post – 20 years, and present (post – 40 years) in (a) soft sediment and (b) coarse sediment.

Table 2.1. Benthic macroinvertebrate taxa counted before the construction of Gardiner Dam (pre), 20 years after its construction (post 20), and 40 years after its construction (post 40) at reference sites (SSR Leader and NSR Borden) and dam site (SSR Gardiner Dam).

	North Saskatchewan River						South Saskatchewan River											
	NSR Borden						SSR Leader						SSR Gardiner Dam					
	Soft Sediment			Hard Sediment			Soft Sediment			Hard Sediment			Soft Sediment			Hard Sediment		
Major taxa	pre	post 20	post 40	pre	post 20	post 40	pre	post 20	post 40	pre	post 20	post 40	pre	post 20	post 40	pre	post 20	post 40
Annelida																		
Hirudinea								1						9				
Nematoda		1				1			1						237			36
Oligochaeta	89	192				81	31	44	2		20	1		132	325		264	283
Mollusca																		
Gastropoda											1			3	1			
Pelecypoda			2			34		1			9	1		26				
Crustacea																		
Amphipoda											1			40			1	1
Decapoda						2												
Insecta																		
Coleoptera					1	1		1			1	2		1			3	
Diptera	51	67	36	9		101	83	62	10	3	142	131		950	1851	2	45	872
Ephemeroptera		4	2	41	3	623		35	4	110	126	22		5		42		
Hemiptera						17			2			1		1				
Megaloptera						1												
Odonata		4				1		1			7							
Plecoptera		1			1	4					4	6		1				
Trichoptera		2	10	1		140		1	1	14	162	1				50		2
Total	140	271	50	51	5	1006	114	146	20	127	473	165	0	1168	2414	94	313	1194

Summarizing the community structure using non-metric multidimensional scaling (NMDS), we see little separation in the benthic community at the SSR Gardiner site today or in 1988 from reference sites (SSR Leader and NSR Borden) in soft sediment (Fig. 2.3a), and ANOSIM confirms the null hypothesis that there is no significant difference between years nor sites (R = 0.123, p >0.05). However, in hard sediment, ANOSIM shows a significant difference in the benthic community at SSR Gardiner Dam relative to reference sites (R = 0.46, p =0.01; Fig. 2.3b). The test site at SSR Gardiner Dam is within the groups of reference sites pre-dam, but in 1988 and 2008 the community is grouped separately from the reference sites and the historical community.

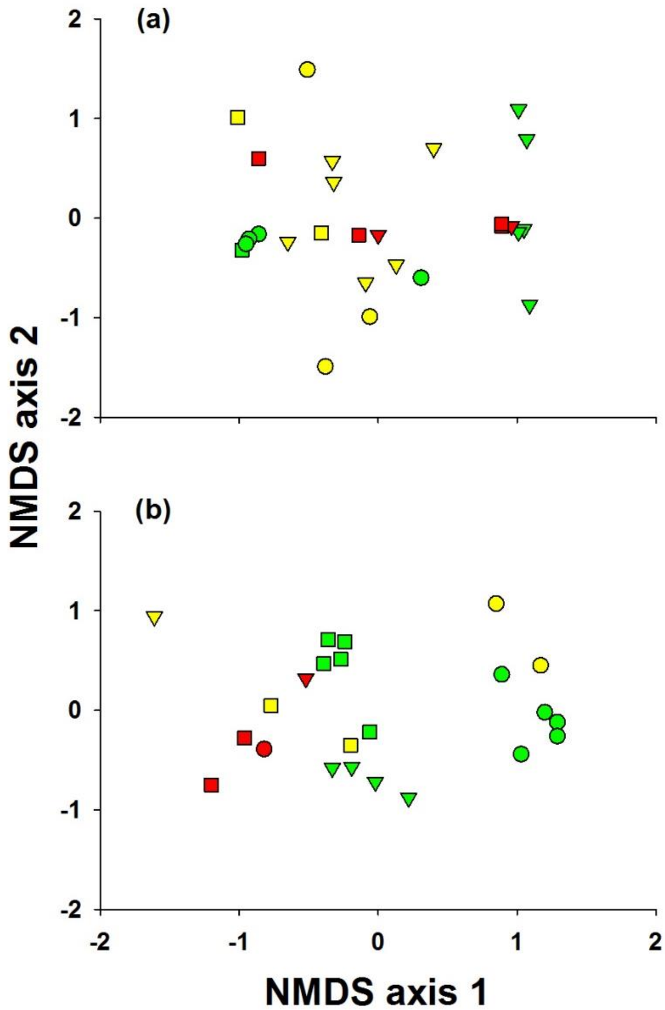

Fig. 2.3. Nonmetric multidimensional scaling ordinations constructed using Bray Curtis distance coefficients for benthic macroinvertebrate assemblages collected from reference sites (NSR Borden [inverted triangles] and SSR Leader [squares]) and the test site at Gardiner Dam (SSR Gardiner Dam [circles]) across years 1957 (pre-dam), 1988 (20 years post-dam), and 2008 (40 years post-dam) in red, yellow and green respectively for soft sediments sampled with Peterson samplers (a), and hard sediment sampled with Hess samplers (b; final stress for 2-dimensional solution = 0.17 [a] and 0.10 [b] respectively).

There was no changes in Trichoptera and Ephemeroptera downstream of the dam in soft sediment (Fig. 2.4a and b respectively), but increased density in Oligochaeta and Diptera (Fig. 2.4c and d respectively) in soft sediment relative to historical conditions. Further, Trichoptera and Ephemeroptera (Fig. 2.5a and b respectively) decreased in hard sediment, while Oligochaeta and Diptera (Figure 2.5c and d respectively) increased to dominate communities in SSR Gardiner Dam.

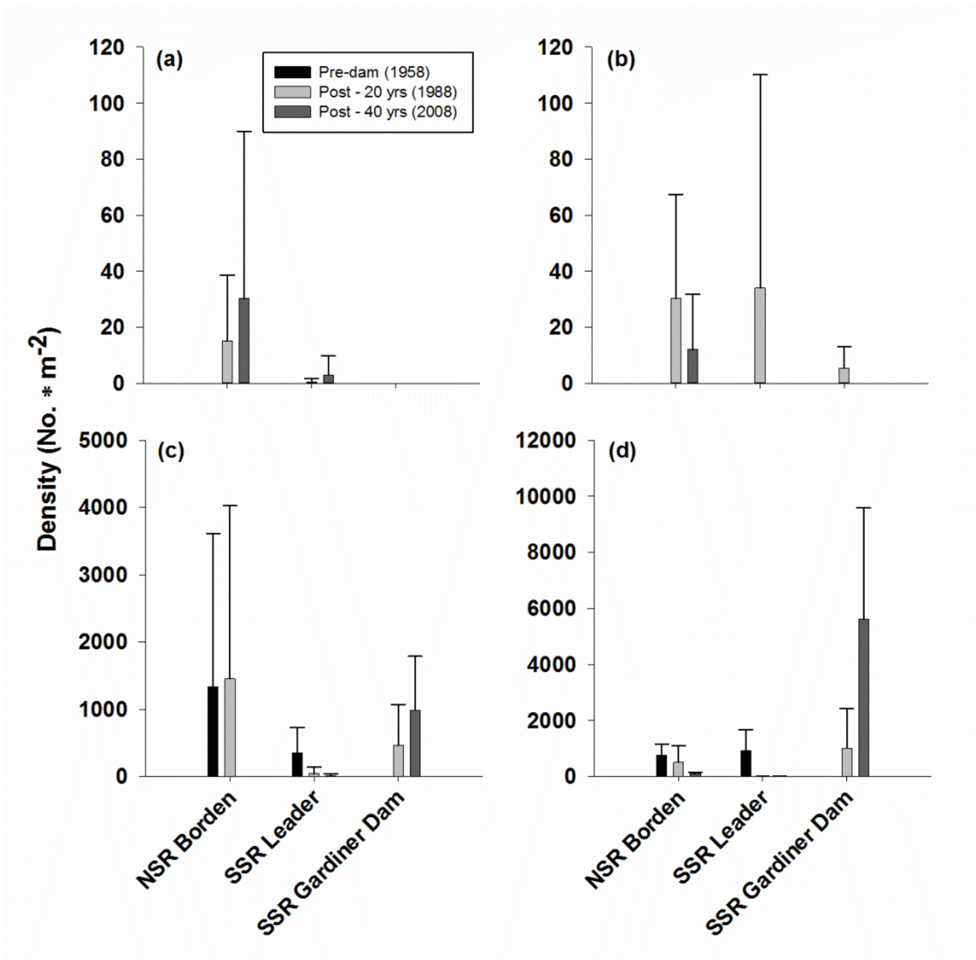

Figure 2.4. Mean benthic macroinvertebrate densities of (± 1 SD) between pre, post – 20 years, and present (post – 40 years) for (a) Trichoptera, (b) Ephemeroptera, (c) Oligochaeta, and (d) Diptera in soft sediment.

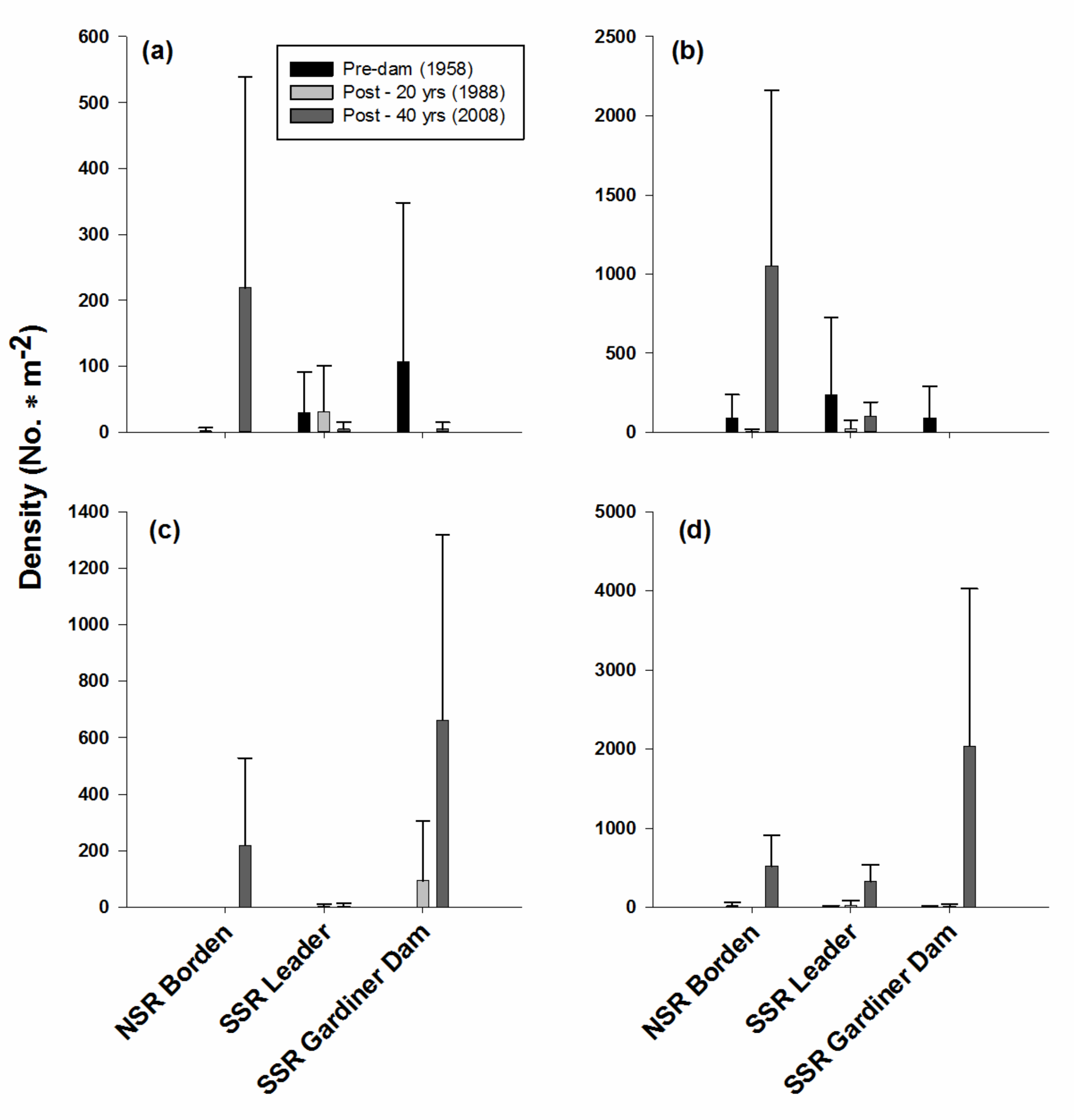

Fig. 2.5. Mean benthic macroinvertebrate densities of (± 1 SD) between pre, post – 20 years, and present (post – 40 years) for (a) Trichoptera, (b) Ephemeroptera, (c) Oligochaeta, and (d) Diptera in hard sediment.

Discussion

These results support the hypothesis that there have been changes in the density and community composition of benthic macroinvertebrates through time downstream of Gardiner Dam. Specifically, Trichoptera and Ephemeroptera decreased and Diptera and Oligochaeta densities increased, with the exception of soft sediment Trichoptera and Ephemeroptera groups which showed no change in density. Overall, benthic macroinvertebrate densities have increased since the construction of the dam in the reach downstream of the dam and there are changes in the benthic community despite a coarse level of taxonomic resolution. In hard sediment, the community has changed from that of 1957, and is much different from that of the communities at reference sites regardless of period. Densities of Oligochaeta and Diptera may have increased at SSR Gardiner Dam through time through release from competition with Ephemeroptera and Trichoptera, or because new niche space was opened that did not exist prior to dam construction (e.g., less scouring from sediment, increased primary production with clear water discharge etc.).

It is impossible to discern what species comprised the benthic communities of the 1950's when the original surveys were completed because the samples have been discarded. However, Lehmkuhl (1972) identified 15 species of Ephemeroptera which were precluded from establishing downstream of the Gardiner Dam ~15 years after the survey and ~3 years post-dam operation initiation. The species listed by Lehmkuhl (1972) could have been the taxa occurring in the Ephemeroptera reported by Reed (1959) and they could provide an indication of what water managers should expect if the impacts of the dam (hydropeaking and cold-water release for example) are mitigated. Lehmkuhl (1972) speculated that the loss of Ephemeroptera downstream of the dam is the result of decreased instream temperature through hypolimnetic release maintaining depressed temperatures. Specifically, the species lost required an increase in temperature to trigger release from diapause, so they could not complete a full lifecycle (Lehmkuhl 1972). However, this study is limited in its ability to make inferences about what specifically may have led to a decline in Trichoptera in hard sediment through time, due to the lack of species-level identification.

Separation of the Gardiner Dam site communities from the others indicates greater differences if these were studied at lower taxonomic resolution; unfortunately, none of the historical samples exist to identify further. By only being able to analyze the historical community at the order level there is a chance of committing a Type II error whereby the results would show no difference between these sites when one actually exists at the species level. Certainly this is possible in the soft sediment at this site, since comparisons of the lowest practical taxonomic designation in 2008 between reference and test sites clearly separated the test sites (Phillips et al. 2015)—likely due to the persistence of stenothermic midge larvae at the Gardner Dam discussed below. Regardless, there is a marked increase in the density of benthic macroinvertebrates downstream of the Gardiner Dam relative to both reference sites today, and historical densities. Ultimately the changes observed in this study result in altered communities and the forage resources available to fish are highly altered.

There are few published reports of the long-term impacts observed in the current study on aquatic biodiversity, which could potentially continue to change as tolerant and diverse taxa such as Diptera colonize these altered environments (typified by the occurrence of cold-tolerant midges such as the genus *Paracladius* [Walker et al. 1991] today [Phillips et al. 2015]). Our results showed changes in the benthic community 40 years after the construction of the dam. However, we found no evidence of a major change in the benthic macroinvertebrate densities 20 years post-construction. Experiments are needed to understand the speed of community changes that occur following dam construction, and how mitigation of temperature and flow effects can change the communities to resemble what would have been historically. The current study provides at least a coarse snapshot of what restoration objectives should target as a community composition end-point at the ordinal level if site-specific objectives are intended to emulate historical biodiversity conditions.

Acknowledgements

We are grateful for the field assistance provided by Kevin Kirkham, Deanne Schulz and Alison Anton. Benthic macroinvertebrate identifications were provided by Dale Parker (AquaTax Consulting, Saskatoon, Saskatchewan). Allen Young, Jennifer Merkowski and Rob Wallace all provided valuable assistance in locating historical fisheries and limnology reports on the Saskatchewan River system. This paper was improved by the comments of two anonymous reviewers. Funding for the field work associated with this work was provided by the Water Security Agency of Saskatchewan, Water Sustainability Fund, while support for manuscript preparation was provided by an NSERC PGS-D scholarship to IDP.

CHAPTER 3: THERMAL ALTERATION AND MACROINVERTEBRATE RESPONSE BELOW A LARGE NORTHERN GREAT PLAINS RESERVOIR.

Produced in:

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Abstract:

Large hydroelectric dams directly alter the abiotic condition of rivers by releasing water with large differences in temperature relative to natural conditions. These alterations in the thermal regime along with alterations in flow often result in altered ecosystems downstream. For sustainable management of this aquatic resource there is a need to balance services a reservoir provides with ecosystem protection. Here we conduct a high-resolution study on the thermal regime and associated aquatic macroinvertebrate community downstream of a hydroelectric dam on a large-order Northern Great Plains River and analyze these findings with a non-central hypothesis test model; Test Site Analysis. Specifically, we monitor the temperature regime of this tailwater environment, and compare the annual change in temperature downstream to reference sites unaffected by the dam. We find that reaches downstream of the dam are significantly cooler than reference through the summer into August and do not reach the temperature optima of reference reaches. This cold-water release, or some other change in flow characteristics, changes abundance, diversity, % Orthocladiinae, and community composition. We used these metrics to compare test sites to reference sites and to quantify the impact of the

Lake Diefenbaker reservoir on community metrics characteristic of temperature stress. Our results have implications primarily for understanding and quantifying the ecosystem impacts of hydroelectric energy production, but also range expansion of cold-water tolerant taxa, the life-history of select groups of invertebrates, and ultimately the forage resources available to the fish assemblages of this river system.

Introduction

Large reservoirs and associated rivers in the Northern Great Plains are fundamentally important in supplying domestic, agricultural, industrial and recreational water resources, particularly in arid environments where rainfall alone is insufficient to meet water demands. These impoundments often adversely affect river ecosystems (Poff and Hart, 2002) through changes in geomorphic processes (Kondolf, 1997), flow regimes (Trotzky and Gregory, 1974; Mims and Olden, 2013) suspended solids, and thermal regimes (Lehmkuhl, 1972; Stanford and Ward, 1981). The changes in water temperature that reservoirs impart on instream conditions downstream is widely recognized as an environmental concern (Poole and Berman, 2001; Caissie, 2006), although environmental flow assessments typically overlook thermal pollution (Olden and Naiman, 2010). If water managers seek to understand the ecosystem consequences of thermal stress and target improvements to dam operations they must have greater resolution in quantifying the thermal regime through time and with distance from the structure (Olden and Naiman, 2010), and how alterations in a thermal regime can impact instream ecology.

Here we focus our study directly on the particular dynamics of the temperature regime, and the effect it has on the benthic community. Because this river system has documented losses of specific taxa due to thermal pollution (Lehmkuhl, 1972) it provides a context from which we can examine specific associations between thermal alterations and benthic communities. Specifically, we seek to evaluate the hypotheses that benthic macroinvertebrate abundance, diversity, and community structure are affected by the Gardiner Dam, and associate these changes to particular impacts of the thermal regime on the life histories of the taxa present in the South Saskatchewan River system. We use several lines of investigation, beginning with a focused study within the South Saskatchewan River in 2007, and then expanding to a regional perspective in 2008 and 2009 whereby we include additional sites in the North and mainstem Saskatchewan rivers.

Finally, we apply Test Site Analysis to statistically evaluate the magnitude of dam impact on temperature-specific metrics. As a benchmark for setting acceptable goals in benthic community composition downstream of the Gardiner Dam, we used reference reaches in the river system upstream of Lake Diefenbaker, neighboring North Saskatchewan River, and the mainstem of the Saskatchewan River. This approach follows the monitoring and assessment concept proposed by Olden and Naiman (2010) whereby the natural thermal regime is used as a template for expectations in environmental flow management. We hypothesize that Test Site Analysis (Bowman and Somers, 2006) will clearly demonstrate differences in reaches downstream of the Gardiner Dam relative to reference sites, and this tool could be used to monitor mitigation efforts in this river system.

Methods

Study sites

The Saskatchewan River system begins in the Rocky Mountains in Alberta to the west, and discharges into the Lake Winnipeg system in the boreal forest to the east. Within the province of Saskatchewan, the river system is divided into the two major branches of the Saskatchewan River system, the North and South Saskatchewan Rivers, for the majority of its length. The mean discharge of the South Saskatchewan River at Saskatoon in 2008 was 213 $m^3 \cdot s^{-1}$, and 222 $m^3 \cdot s^{-1}$ in the North Saskatchewan at Prince Albert in 2008 (http://www.wsc.ec.gc.ca/applications/H2O/index-eng.cfm). These branches converge downstream of the city of Prince Albert to become the mainstem of the Saskatchewan River. For the majority of the length of the two branches they flow through the Prairie Ecozone, with a transition to the Boreal Plains Ecozone near their confluence. Fauna in the Saskatchewan River system are typified by prairie taxa with some overlap with taxa found in boreal rivers that flow into the mainstem river in the Boreal Plains Ecozone. Miyazaki and Lehmkuhl (2011) provide a detailed description of the Saskatchewan River system and its aquatic insect fauna.

In 2007 we selected sites to focus on the Gardiner Dam itself, and the local influence on temperature regimes in the South Saskatchewan River (Table 3.1; Fig. 3.1). To record temperature we submerged and anchored v2 electronic temperature data loggers (Onset Computer Corporation, Bourne, Massachusetts) set at hourly recording intervals for a full annual cycle. Geographically, we established an upstream control site near Leader, Saskatchewan, and then identified treatment reaches at 10 km intervals downstream of Gardiner Dam. The furthest downstream site was located at Clarkboro Ferry, for a total of 14 reaches.

Table 3.1. Steps, objectives, methods used, resolution of analysis, statistical analysis used, and period covered for the methods used in the current study of dam effects on the benthic community. The steps covered are progressive from characterizing the temperature regime effects of the dam in objectives one and two, then applying benthic sampling methods to study the impact the dam has on benthic macroinvertebrate communities in objectives three and four.

Steps	Objective	Method	Resolution	Analysis	Period
1	Characterization of temperature regime downstream of dam	Temperature Loggers	hourly 1 year	Two sample Kolmogorov-Smirnov	2007
2	Characterization of temperature regime relative to reference condition	Temperature Loggers	hourly 1 year	Two sample Kolmogorov-Smirnov	2008
3	Identify metrics of community structure forensic of dam impact	Peterson Grabs - soft sediment, Hess Samplers - hard sediment, Crayfish trapping & survey	lowest practical identification (species where possible)	Community Analysis (ANOVA, ANOSIM, SIMPER, NMDS)	2008
4	Evaluate magnitude of dam impact on cold-metrics relative to reference condition	Multihabitat Travelling Kick and Sweep	lowest practical identification (species where possible)	Test Site Analysis	2008, 2009, 2013

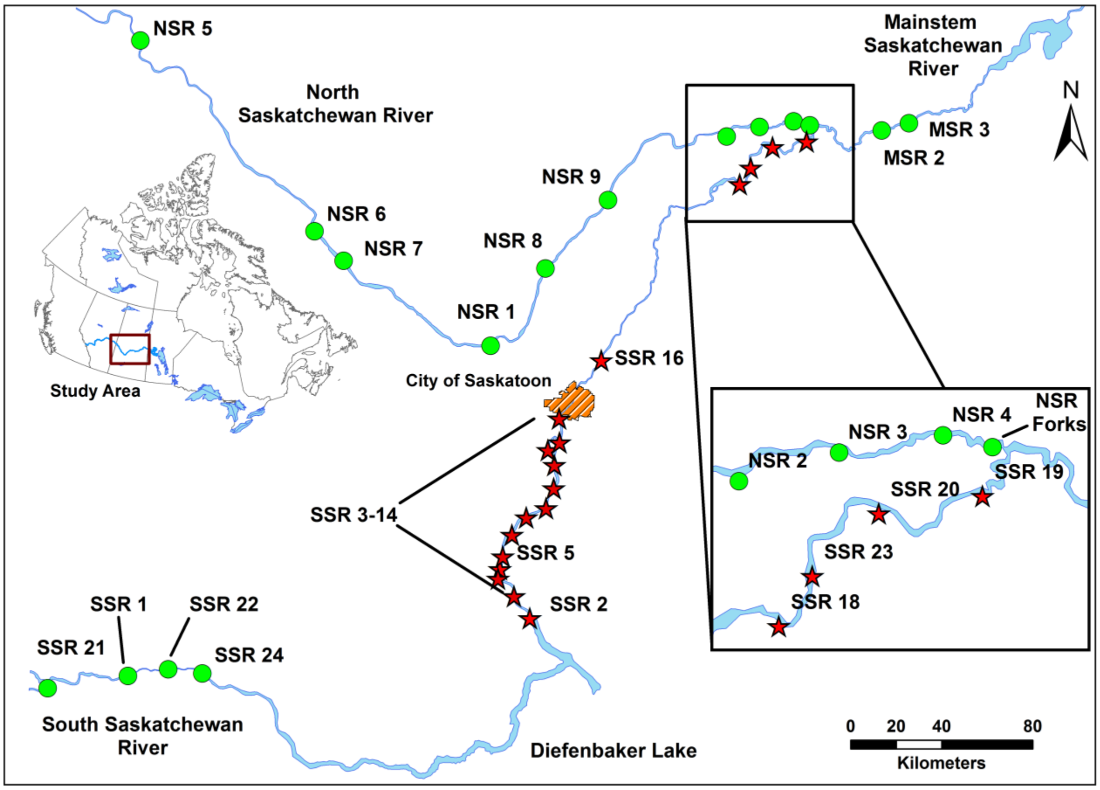

Figure 3.1. Benthic macroinvertebrate community assessment and temperature study sites on the Saskatchewan River System. Stars denote test (red) vs. reference (green) as circles.

In 2008 we deployed temperature data loggers at nine reaches on the South Saskatchewan River, five reaches on the North Saskatchewan River, and two reaches on the Mainstem of the Saskatchewan River proper for 16 reaches at a regional level (Table 3.1; Fig. 3.1). Sites SSR1, SSR2, and SSR16 in Figure 1 were monitored in both 2007 and 2008 and sites SSR3-SSR14 were monitored in 2007. Detailed descriptions of the habitat at site reaches can be found in Pollock et al. (2008), but habitats are typically dominated by sandy sediment (~80%) with patches of cobble and boulder (~20%). Because the river system can range from a kilometer wide and braided to only ~200 m wide, depth varies greatly, from very shallow to nearly 10 m deep in some areas, with averages of ~1 m depth in the study reaches.

Benthic community assessment-dam effects on benthic community

In August 2008 we collected benthic macroinvertebrates at each reach using sampling techniques best suited to the habitat; however, sampling effort was consistent across the study. In each reach we collected five grab samples using a Peterson Dredge (base = ~0.022 m^2; Table 3.1). Each sample was a product of three grabs combined together. We also collected Hess samples (base = ~0.086 m^2; Table 3.1) from five different riffles in each reach (n=5). Mean densities of individual taxa in each reach were estimated separately for soft sediment and riffle habitats by averaging the five Peterson dredge and five Hess samples respectively. We sampled reference sites SSR 1, SSR 16-20, NSR 1-5, MSR 1-2, and test site SSR 2 for a total of 14 sites (Fig. 3.1).

Benthic community assessment-reference condition and Test Site Analyses

We sampled the South, North, and Mainstem Saskatchewan River using the Big River Protocols of the United States Environmental Protection Agency (USEPA; Flotemersh et al., 2006). We applied the reference condition approach to assess the impact of the Gardiner Dam, comparing this test site to reference sites that were uninfluenced by the Gardiner Dam.

The test sites began at Gardiner Dam (SSR 2; Fig. 3.1) on the South Saskatchewan River and continued downstream until the end of the South Saskatchewan River at its junction with the

North Saskatchewan River (SSR 5, SSR 16, SSR 18, SSR 19, and SSR 20; Fig. 3.1). Reference Sites were located on the South Saskatchewan River upstream of Lake Diefenbaker, in the North Saskatchewan River, and on the Mainstem Saskatchewan River (n = 3, 12, and 5 respectively; Fig. 3.1). These sites were sampled between 5 August and 15 September in the year they were sampled (Table 3.1; Table 3.2).

Table 3.2. List of sites used in the Test Site Analysis component of the Lake Diefenbaker, Gardiner Dam impact study.

Site	Test or Reference
SWA_2008_MSR 2	Reference
SWA_2008_NSR 1	Reference
SWA_2008_NSR 2	Reference
SWA_2008_NSR 3	Reference
SWA_2008_NSR 4	Reference
SWA_2008_SSR 1	Reference
SWA_2008_SSR 16	Test
SWA_2008_SSR 18	Test
SWA_2008_SSR 19	Test
SWA_2008_SSR 2	Test
SWA_2008_SSR 20	Test
SWA_2008_SSR 5	Test
SWA_2009_NSR 10	Reference
SWA_2009_NSR 5	Reference
SWA_2009_NSR Forks	Reference
SWA_2009_SSR 24	Reference
WSA_2013_MSR 3	Reference
WSA_2013_NSR 8	Reference
WSA_2013_NSR 3	Reference
WSA_2013_NSR 6	Reference
WSA_2013_NSR 7	Reference
WSA_2013_SSR 23	Reference
WSA_2013_SSR 22	Reference

We used a conventional D-frame net (base of 30 cm, 500 μm mesh) to obtain a single qualitative assemblage sample from each site, comprised of 12 transect sweeps based on the Large River Bioassessment Protocol (LR-BP) developed by Flotemersch et al. (2006) and recommended for large non-wadeable rivers by Johnson et al. (2006). This method consisted of sampling six transects at 100 m intervals on both banks of the reach for a total of 12 transects. Each transect consisted of a 10-m sample zone (5.0 m on each side of transect) extending from the edge of water to the mid-point of the river or until depth exceeds 1.0 m. Six sweeps, each 0.5 m in length, comprise each 5.0 m side transect, and each sweep covers 0.15 m^2 of substrate (i.e., net width of 0.3 m and a 0.5 m length of pass); therefore, six sweeps sampled an approximate area of 0.9 m^2. The six sweeps were allocated proportionately to available habitat within the 10-m sample zone (e.g., snags, macrophytes, cobble). Samples from the entire reach were composited into a single sample. When there was a large amount of sediment, we decanted off the organic material using a swirling technique and 20 L bucket.

Several crayfish capture methods were adapted for the flow conditions. First, we searched the littoral zone of the rivers. Moving upstream, substrate was turned over and of crayfish observed in 100 m transects were recorded (modified from Davies, 1989). Second, D-frame nets (500 µm mesh) were used in a travelling kick and sweep method to collect as many crayfish in the wadeable portion of the river as possible in three, one-hour searches per site. When conducting the benthic macroinvertebrate assessment described above, observations of crayfish were also recorded; however, the small areas covered by these samplers produced low crayfish numbers. Crayfish were also sampled using modified Gee-Minnow traps (Hein et al., 2006) by setting 15 traps for 24 hours.

Sample preservation and processing

All benthic samples were stored in 80% ethanol until returned to the laboratory where organisms were sorted from the organic material under 7 X magnification. Benthic macroinvertebrates were identified to lowest possible designation using keys for North America (Merritt and Cummins, 1996) and Western Canada (Dosdall and Lehmkuhl, 1979; Brooks and Kelton, 1967; Clifford, 1991; Merritt and Cummins, 1996; Larson, 2000; Webb, 2002). Additional taxa-specific keys and literature were used for the identification of unique midge fauna (Diptera: Chironomidae) occurring in the Saskatchewan River system (Hirvenjoa, 1973; Oliver 1976, Oliver and Roussel 1983). Voucher specimens were deposited in both the Water Security Agency of Saskatchewan Invertebrate Voucher Collection (Saskatoon, Saskatchewan), and the Royal Saskatchewan Museum (Regina, Saskatchewan). Taxa occurrence records were submitted to the Saskatchewan Conservation Data Centre with the Ministry of Environment (http://www.biodiversity.sk.ca/).

Data analyses-dam effects on thermal regime

All temperature measurements are summarized by compiling the mean weekly temperature at each site, for the ice-free season from late April to late August to encompass the biologically active period of the year. The temperature series of annual warming through summer and cooling into fall was then compared between sites using two sample Kolmogorov-Smirnov tests in the *ks.test* function from the *truncgof* package for R, version 2.13.1 (KS test; R Development Core Team, 2013). In 2007 we used the site at Leader upstream of Gardiner Dam as reference probability distributions for the KS test in pairwise comparisons to sites from Gardiner Dam downstream to Saskatoon. Similarly, the SSR at Leader was used as a reference probability distribution for the 2008 regional study of temperature regimes, comparing it to the far-field sites on the South Saskatchewan River, and North Saskatchewan River site at Borden, and the Gardiner Dam site.

Data analyses-dam effects on benthic community

We summarized the benthic macroinvertebrate community into metrics of Shannon diversity, evenness, functional feeding group (Merritt and Cummins, 1996), and % Ephemeroptera, Plecoptera, Trichoptera (%EPT). We used an analysis of variance (ANOVA) to compare these metrics between reference sites and each test site in both soft and hard sediment (Table 3.1). Data were $\log_{(n+1)}$ transformed to meet the assumptions of normality and equal variance.

We used an analysis of similarities (ANOSIM), similarity percentages (SIMPER), and non-metric multidimensional scaling (NMDS) ordination (Clarke and Warwick, 2001; Table 3.1) to examine benthic community structure between sites across the South and North Saskatchewan Rivers. ANOSIM and SIMPER analyses were done using PRIMER Version 6.1.13 (PRIMER-E software, Plymouth, United Kingdom; Clarke and Warwick, 2001), and the *vegan* and *MetaMDS* packages for R, version 2.15.1 (R Development Core Team, 2013) for NMDS. Abundance data were $\mathrm{Log}_{(n+1)}$ transformed, then used to calculate the taxa-by-taxa dissimilarity matrix of all benthic taxa in two data sets: soft sediment, and coarse sediment (produced from Peterson and Hess dredges respectively) in all approaches. We calculated stress, optimal distance linking metrics (e.g., Mahalanobis, Bray-Curtis, Gowers, and Kulczynski distance metrics), and optimal numbers of dimensions prior to analysis, and selected the distance linking metric based on the highest non-metric fit r^2. The Bray-Curtis distance metric was the optimal distance linking metric for both Peterson (r^2=0.98) and Hess (r^2=0.98) data matrices. We considered our NMDS solution optimal and with a good representation of the community structure if stress is less than 0.10, yet acceptable if it had a final stress less than 0.18 (Clarke and Warwick, 2001). Finally, we chose the optimal number of dimensions to apply by including subsequent dimensions until we arrived at a solution explaining greater than 85% of the variance.

We used ANOSIM to compare average rank similarities of taxa in samples between the river reaches. ANOSIM calculates a test statistic, the *R*-statistic, which varies between 0 and 1; high values indicate differences between river reaches. A test of the significance of the *R*-statistic is obtained by comparing the observed value to a distribution of values expected under the null hypothesis of no difference between treatments (Clarke and Warwick, 2001). This nonparametric permutation test was used in preference to comparable parametric tests such as multivariate analysis of variance, because the latter are based on assumptions (e.g., that abundances follow a multivariate normal distribution) unlikely to be satisfied for most multispecies data sets (Clarke and Warwick, 2001). In contrast, SIMPER evaluates the role individual species play in contributing to the separation between two groups of samples. We applied SIMPER here to identify which taxa in our communities strongly influenced the separation between groups.

Finally, we constructed a graphical summary of the relationships described above using NMDS, and coded this single ‘test’ Gardiner Dam site differently from the reference sites throughout the Saskatchewan River system for the historical data (at SSR 1, SSR 2, and NSR 1) in one analysis, and the regional comparison for a second analysis.

Data analyses- Test Site Analysis

For the travelling kick and sweep samples used in the Test Site Analysis we used four metrics; abundance (as a proxy for Density), Shannon's Diversity, % Orthocladiinae, and aquatic invertebrate community composition (axis 1 scores from a correspondence analysis ordination of the sites-by-taxa data matrix). Shannon's Diversity Index was calculated using PRIMER and the Correspondence Analysis was calculated using BiPlot for Excel (Smith, 2011). Based on results of the soft and hard sediment community analysis, we *a priori* expect abundance to increase, diversity to decrease, % Orthocladiinae to increase, and the community to change if Gardiner Dam is impacting a site. We then calculated the Mahalanobis distance from the test site to the multi-index centroid of the reference sites at each test site. The final value is then evaluated for deviations from the average reference site condition using Test Site Analysis (Bowman and Somers, 2006). Reference conditions were defined as within the 90th percentile of reference sites.

The overall biotic condition values were determined by calculating the magnitude of difference between a test site and the average reference site condition and evaluating the probability that this difference falls outside the range of variation at reference sites. The results parallel the Water Quality Index (SWA, 2010) grouped into three categories of health with associated statistical thresholds: Significantly different than reference (≥ 90 % chance of being in reference), different (> 10, < 90 %, chance of being in reference), or within reference condition (≤ 10 % chance of being in reference). For each site, condition was assessed using each individual metric as well as using all four metrics at once.

Results

Temperature

All comparisons of mean daily temperature regimes downstream of Gardiner Dam until SSR 16 relative to the upstream control (SSR 1 – Leader) revealed significant differences through the period studied (Sites SSR2 – SSR13 all with test statistic >1.19, df=1, and $p<0.001$). Temperature downstream of the Gardiner Dam was slower to rise in the spring had lower mid-summer maxima, and cooled more slowly in the fall than the upstream reference site (Fig. 3.2).

In the broader regional comparison of 2008 (SSR 1, 2, 15-20, NSR 1-5, and MSR 1 and 2), the area downstream of Gardiner Dam expressed the only significantly different temperature regime relative to the upstream control site on the South Saskatchewan River at Leader (SSR 1; KS Test Statistic 3.77, df=1, $p<0.001$; Fig. 3.2), being ~ 9 °C cooler than reference from the beginning of May through to the end of July.

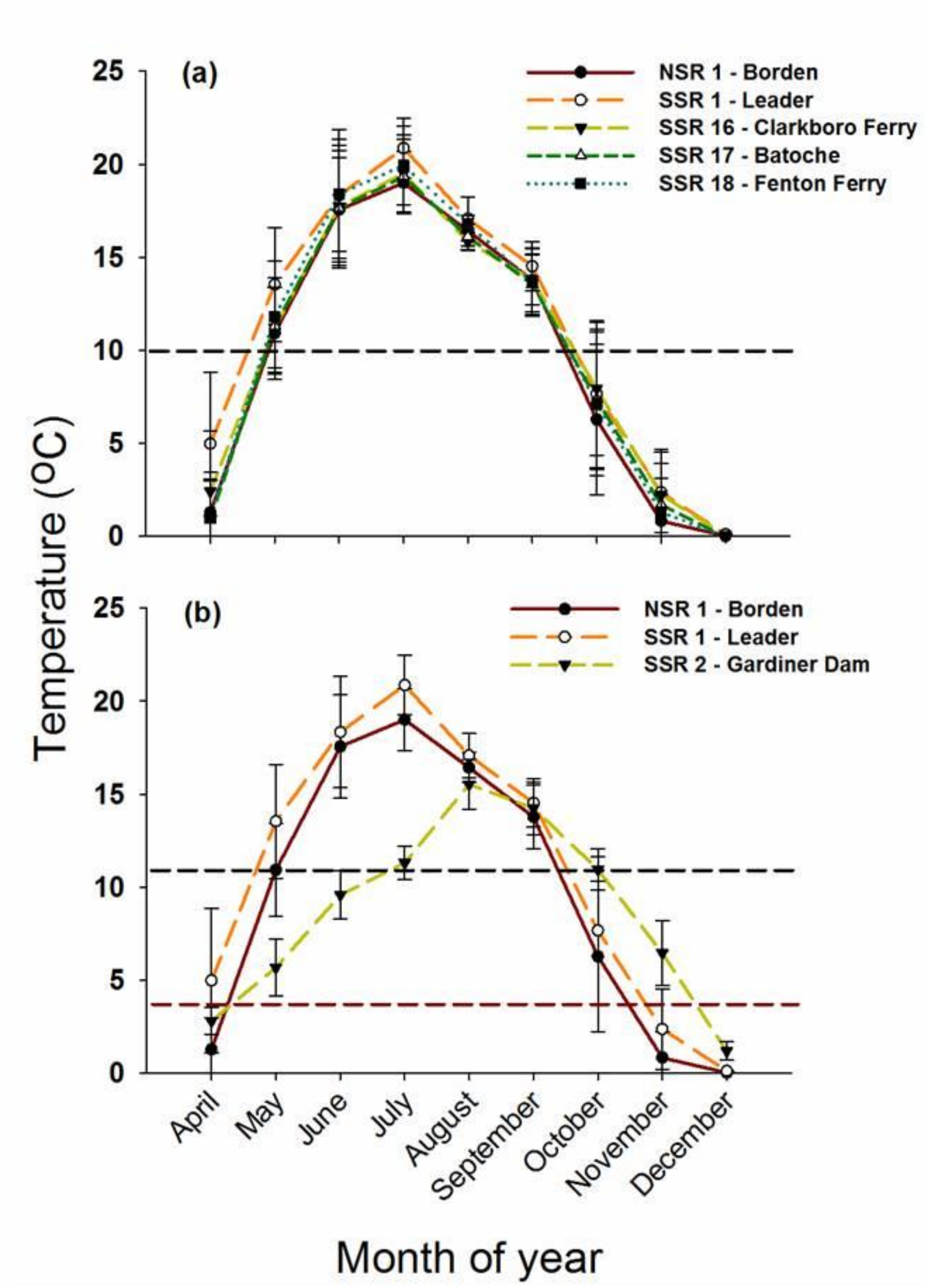

Figure 3.2. Mean monthly temperature (± 1 St. Dev.) overlain with (a) the temperature threshold for growth of the mayfly *Hexagenia limbata*, and (b) the temperature at which eggs of the native crayfish *Orconectes virilis* hatch (upper dashed line) versus the temperature at which eggs of the invasive crayfish *O. rusticus* hatch (lower dashed line). Thermal thresholds are superimposed on the plots as horizontal dashed lines.

Benthic macroinvertebrates- community assessment

In total 34,425 individuals representing 175 different invertebrate taxa were collected. Mean density ranged from 0 to 8,348 ind.•m^{-2} and number of taxa ranged from 0 to 58 taxa. In soft sediment, the number of taxa and Shannon's Diversity were both significantly lower downstream of the Gardiner Dam relative to reference sites (Table 3.3). However, the density of macroinvertebrates, and percentage of predator, scraper and shredder functional feeding groups were all significantly higher than reference sites (Table 3.3).

Hard sediment habitats downstream of Gardiner Dam had a greater number of metrics showing significant differences to reference than soft sediment. Specifically, the number of taxa and Shannon's Diversity decreased, but so too did evenness of the assemblage, and % EPT taxa. Further, the density of macroinvertebrates and % shredder taxa were significantly higher downstream of the dam, but the % filterers, predators, and scrapers were all significantly lower (Table 3.3).

Table 3.3. Summary of analysis of variance (ANOVA) results testing for differences between site downstream of Gardiner Dam (Test Site SSR 2) relative to reference sites (See Table 1 and Figure 1) for variables related to macroinvertebrate assemblage structure. H´ = Shannon diversity; EPT = Ephemeroptera, Plecoptera, Trichoptera, FFG = functional feeding group. Arrows beside F values indicate direction of change relative to reference sites.

	ANOVA			
	Soft sediment		Hard sediment	
Variable	*F*	*p*	*F*	*p*
Number of taxa	56.140 ↓	< 0.001	24.342 ↓	< 0.001
Density	10.632* ↑	0.014	9.311* ↑	0.025
H´	11.777* ↓	0.008	14.160 ↓	< 0.001
Evenness	ns	ns	20.782 ↓	< 0.001
% EPT	ns	ns	13.561* ↓	0.004
% FFG				
Collector-gatherer	ns	ns	ns	ns
Filterer	ns	ns	12.806* ↓	0.005
Predator	11.571* ↑	0.009	6.306 ↓	0.006
Scraper	13.137* ↑	0.004	23.047 ↓	< 0.001
Shredder	11.942* ↑	0.008	13.359* ↑	0.004

* indicates comparisons where variable did not meet assumptions required for parametric evaluation, and are Kruskal Wallis test statistics.
ns' indicates no significant difference

The ordination plots separate samples from reference sites and the site immediately downstream of the dam for both sampling methods (Fig. 3.3). The ANOSIM *R*-statistic values support this finding: soft sediment (*R*-statistic = 0.995, $p = 0.001$) and hard sediment (*R*-statistic = 0.208, $p = 0.013$). Based on the *R*-statistic, there is a large difference reference and test habitats. Overall, 12 species accounted for a total of 91% of the dissimilarity in the soft sediment, and 30 species accounted for a total of 90% of the dissimilarity in the hard sediment between the test site and the reference sites. Within the soft sediment, all 12 of these taxa contributed > 3% of between-group dissimilarity; however, hard sediment was characterized by 9 taxa each contributed > 3% to between-group dissimilarity, comprising 58% of the overall dissimilarity. In soft sediment the midge genera *Tanytarsus* sp. and the freshwater worms Oligochaeta accounted for 11% and 15% respectively in the difference between the dam reach and reference sites. In hard sediment though, it was the freshwater midge Orthocladiinae group *Cricotopus/Orthocladius* that contributed 17% dissimilarity while Oligochaeta explained 11% of the dissimilarity. In both habitats the chironomids and oligochaetes were more abundant downstream of the dam relative to the reference sites.

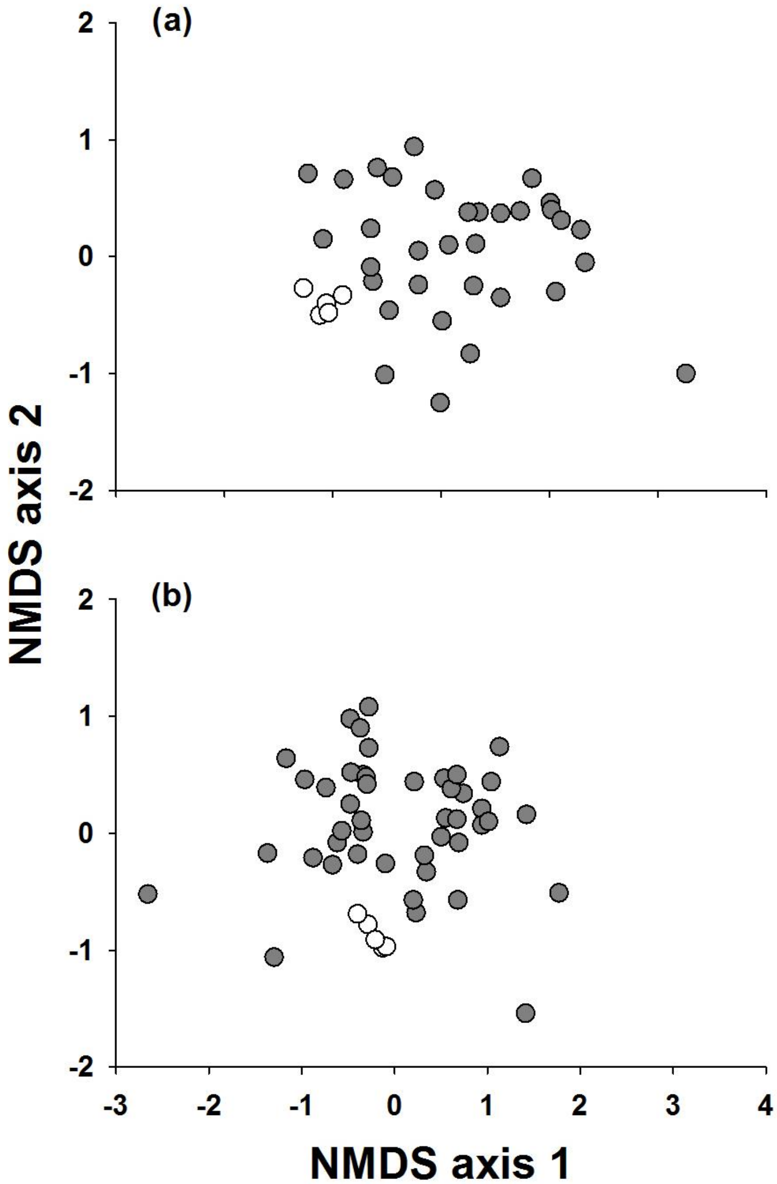

Figure 3.3. Nonmetric multidimensional scaling ordinations constructed using Bray-Curtis distance coefficients for benthic macroinvertebrate assemblages collected from reference sites (grey circles; See Table 1 for list of reference sites) and the test reach (open circles; SSR 2) downstream of Gardiner Dam for soft sediments sampled with Peterson samplers (a), and hard sediment sampled with Hess samplers (b; final stress for 2-dimensional solution = 0.16 [a] and 0.17 [b] respectively).

Crayfish collections yielded nearly 500 crayfish, all of which were *Orconectes virilis* (Hagen). The study site downstream of Gardiner Dam was the only site in the regional survey wherein we did not catch any crayfish, and crayfish were only rarely caught up to 50 km downstream at SSR 5. These two temperature-influenced sites ranked as the two lowest in crayfish population in the Saskatchewan River system.

Life histories

Of the commonly occurring mayflies in this study (>0.1% of total invertebrate abundance), we found adequate life history information for the species *Hexagenia limbata* (Serville) (Britt, 1962; Hudson and Swanson, 1972; McCafferty and Pereira, 1984; Heise et al., 1987) *Tricorythodes minutus* Traver (Newell and Minshall, 1978), *Ephemera simulans* Walker (Britt, 1962), and *Ephoron album* (Say) (Britt, 1962). The tailwater downstream of the Gardiner Dam produced 1,719 degree days exceeding the 10 ^{o}C threshold in 2007 and 1,694 degree days in 2008 above which growth is possible for the mayfly *H. limbata* and did not achieve even the minimum ever recorded for this species (1,728; Heise et al., 1987). In contrast, the reference sites did reach the number of degree days necessary for the reproduction of this species (e.g., SSR 1 = 2,792; SSR 16 = 2,499; and NSR 1 = 2,469 [all for 2008]). We only collected this mayfly in sites with temperature regimes not significantly different than reference sites. Within the South Saskatchewan River *H. limbata* only appears in collections at Clarkboro Ferry (SSR 16), ~120 km downstream of the dam. In addition to the potentially reduced growth of *H. limbata* downstream of the dam, its hatch may also be delayed by approximately a month relative to reference sites as temperatures downstream of Gardiner Dam do not reach the 10 ^{o}C necessary for this species' eggs to hatch until July (Fig. 3.2*a*). The crayfish *Orconectes virilis* too requires a similar temperature of 11 ^{o}C to hatch (Weagle and Ozburn, 1972) and the reach downstream of Gardiner Dam did not reach this temperature until mid-July, over two months after reference sites (Fig. 3.2*b*).

Benthic macroinvertebrates- Test Site Analyses component

In total, 117,177 individuals of 180 invertebrate taxa were collected at sites included in the current analyses. In Test Site Analysis, the benthic macroinvertebrate metrics downstream of the Gardiner Dam in the South Saskatchewan River were significantly different (D = 18.78, nc$P$ < 0.001). Specifically, abundance and % Orthocladiinae downstream of the Gardiner Dam are significantly higher than at reference sites in the Saskatchewan River system (Table 3.4). Benthic macroinvertebrate abundance was assessed to be 4,576 individuals, whereas mean abundance in the reference sites was ~836 (± 847 SD). We estimate these abundances to approximate densities of 4,118 organisms • m^{-2}, and 750 organisms • m^{-2} in the Gardiner Dam test site and mean reference sites respectively.

Table 3.4. Results of the Test Site Analysis component, evaluating the macroinvertebrate metrics of the test site against the reference sites for cold-sensitive metrics; community composition (CA Axis 1), abundance (i.e. density), Shannon's Diversity, and % Orthocladiinae midges. Values below each metric in the table are the non-central P value for individual metrics at individual sites whereas D = the Mahalanobis distance from test site to the reference sites' mean, F = F statistic, and ncP = significance (non-central probability) are based on all four metrics at each site.

Site	**CA Axis 1**	**Abundance**	**Shannon's Diversity**	**% Orthocladiinae**	**D**	**F**	***ncP***
SSR 2	<0.001	<0.001	>0.99	<0.001	18.78	1218.10	<0.001
SSR 5	0.34	>0.99	>0.99	<0.001	6.32	137.87	0.01
SSR 16	>0.99	>0.99	>0.99	>0.99	1.06	3.91	>0.99
SSR 18	0.13	>0.99	>0.99	<0.001	14.35	710.66	<0.001
SSR 19	0.98	>0.99	>0.99	>0.99	2.04	14.34	>0.99
SSR 20	>0.99	>0.99	>0.99	0.16	3.15	34.26	0.71

Further, the benthic macroinvertebrate community occurring at the Gardiner Dam Test Site was significantly different from the reference sites (CA Axis 1; *ncP* = <0.001; Table 3.3). Using scores on the first axis of Correspondence Analysis as a proxy for the benthic community composition, we found that the Gardiner Dam community was markedly different from the other sites (Fig. 3.4). However, Shannon's Diversity was within reference condition (*ncP* <0.99; Table 3.4) at the Gardiner Dam site despite a significant difference in macroinvertebrate abundance and assemblage composition.

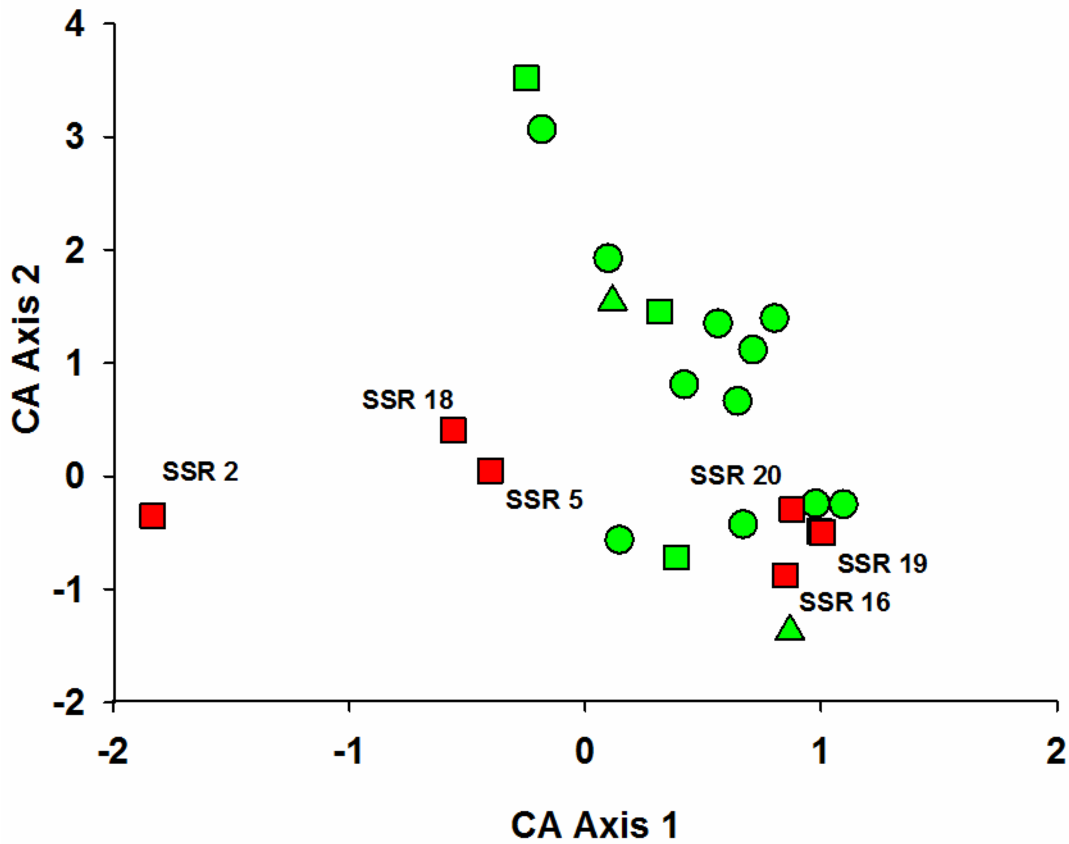

Figure 3.4. Correspondence Analysis ordination of the sites-by-taxa dataset of TK&S samples from the Saskatchewan River system. Red coloured sites indicate test sites, Green colour indicates reference sites. Squares, circles, and triangles represent sites on the South Saskatchewan, North Saskatchewan, and mainstem Saskatchewan Rivers respectively.

The next test site downstream of Gardiner Dam (SSR 5; ~40 km) was also significantly different relative to reference condition (D = 6.3, *ncP =0.01*; Table 4) based on all four metrics. Specifically, the % Orthocladiinae was significantly more abundant relative to reference sites (*ncP*<0.001), and the community (CA Axis 1) was potentially stressed (*ncP*=0.34), but abundance and Shannon's Diversity were within reference (Table 3.4).

The site downstream of the city of Saskatoon is within reference condition (D=1.1, *ncP* >0.99; SSR 16), but the subsequent site downstream (SSR 18) is significantly different (D = 14.3, *ncP* < 0.001). Similar to SSR 5, at SSR 18 the % Orthocladiinae is significantly increased relative to reference sites (*ncP*<0.001), and the community (CA Axis 1) is potentially stressed (*ncP*=0.13), but abundance and Shannon's Diversity are within reference (Table 3.4). Site SSR 19, the next site downstream the South Saskatchewan River, is within reference (D = 2.0, *ncP* >0.99); however, SSR 20, the final site downstream of Gardiner Dam before the forks of the South and North Saskatchewan rivers is potentially stressed, having an increased % Orthocladiinae relative to reference sites (ncP =0.16; Table 3.4).

Discussion

Large riverine systems are among the world‘s most severely impacted ecosystems. Dam related environmental impacts are common in large river ecosystems worldwide, and have been reported as a serious threat to aquatic biodiversity (Nilsson et al., 2005), resulting in decreased benthic macroinvertebrate diversity, increased density and altered assemblage composition (Takao et al., 2008). Quantifying the effects of disturbance in these systems has been hampered by their large size and the absence of comparable sites that can be used in traditional reference condition approaches to assessment. Here we developed an assessment tool tailored to the specific results of our study on the effect of reservoir and dam operation which identified benthos community change, density (abundance), Shannon's Diversity, and the % Orthocladiinae as metrics affected by the presence of Gardiner Dam. The subsequent Test Site Analysis using multiple-habitat collection (US EPA Large Rivers Protocol) on South Saskatchewan River sites downstream of the dam revealed a high impact in the river reaches between the dam and the city of Saskatoon, but then less of a clear trend downstream of the city toward the forks of the North and South Saskatchewan rivers. Although there are no significant tributaries to the Saskatchewan River entering between the Gardiner Dam and the forks of the North and South Saskatchewan Rivers (~300 km) there are multiple human stressors such as urban development, farming, and waste effluent which could affect the river in these reaches, making determination of river impact cause more difficult.

The present benthic community assessment study of the Gardiner Dam in soft and hard sediments indicates significant differences associated with Lake Diefenbaker on the South Saskatchewan River by increasing the density, lowering the diversity, and altering the composition of the benthic macroinvertebrate assemblage, consistent with the findings of Takao et al. (2008). These effects result in an overall significant impact on the benthic macroinvertebrate community of the South Saskatchewan River downstream of the Lake Diefenbaker impoundment with significantly increased abundance, % Orthocladiinae, and altered community structure (CA Axis 1). Using these metrics of dam effects, we can monitor for improvement in river condition under experimental mitigation measures of dam management (such as epilimnetic-release [Olden and Naiman, 2010]).

Our study of the Gardiner Dam indicates effects on the South Saskatchewan River, producing a cold water environment significantly different from natural conditions. We hypothesize that the natural benthic community in the South Saskatchewan River does not experience sufficient degree days to exist because of the observed decrease in instream temperature downstream of Lake Diefenbaker. Consequently, this new cold-water environment produces stress that can be exploited by cold-tolerant invertebrates. We sampled the dominant benthic sediment types (soft sandy and hard cobble) in tailwater environment and found larger densities of chironomids relative to reference sites. In addition, the ecosystem function performed by the benthic community is altered to one dominated by predators, scrapers and shredders in the soft sediment, while filterer, predator and scraper functions are significantly decreased in the hard sediment; and shredders are increased. The % Ephemeroptera/Plecoptera/Trichoptera (EPT) taxa, a common indicator of disturbance and environmental stress in aquatic ecosystems (Merritt and Cummins, 1996), were significantly decreased in hard sediment downstream of the Gardiner Dam. This reduction in EPT taxa has been found in other studies of the impact of dams on communities (Bredenhand and Samways, 2009) suggesting stress on the benthic community because of changes in temperature and flow. However, these changes in EPT taxa could, in part, be a result of a reduction in drift due to the interruption of the reservoir and dam in the river continuum (Marchant and Hehir, 2002).

Although the conditions created in the tailwater are negative for some species, we have also discovered that taxa replacement may be occurring as a new habitat becomes available. In particular, we find the chironomid genus *Paracladius* which is commonly associated with cold-water environments of northern ecosystems and is used in paleolimnological studies as a useful indicator of cold, oligotrophic conditions (Walker et al., 1991). This genus has never before been documented in Saskatchewan, and its occurrence in this tailwater is a valuable forensic character of cold water influence on the benthic community. However, the specific environmental tolerance, thermal plasticity and even life history characteristics for *Paracladius* sp. are poorly known, and further study is needed to project how management of the dam may affect this taxa's prevalence.

Lehmkuhl's original work in 1971 (Lehmkuhl, 1972) identified reductions in mayfly fauna downstream of Gardiner Dam, and suggested that the temperature regime was precluding life cycle completion for species such as *Ephemera simulans* and *Ephoron album*. We echo that suggestion that the temperature regime and possibly flow and turbidity create an environment inhospitable to the mayfly *Hexagenia limbata*. Historically, Webb (2002) and Webb et al. (2004) provide no records of *H. limbata* occurring in even a single collection in the South Saskatchewan River downstream of Gardiner Dam, while documenting this species' occurrence nearly six thousand times at one of our upstream control sites (SSR 1 – Leader) and near our far-field site where the river has returned to a temperature regime similar to the reference sites around Saskatoon (SSR 16 – Clarkboro [current study]).

Lehmkhl (1972) further identified that the only specimens present downstream of the Gardiner Dam were 6 Chironomidae larvae across ~ 27 m of travelling kick and sweep. Although comparisons in methods is difficult between the sampling Lehmkuhl (1972) did in 1971 and the current study, it is reasonable to contrast the current results of high densities and diversity within reference conditions and suggest that the community that colonized the reaches downstream of the Gardiner Dam has not been stable and may have been adapting to the new cold-water environment. Further to this, black fly studies by Fredeen (1981) noted the loss of three species of *Simulium* (Diptera: Simuliidae) downstream of the dam, and the emergence of an otherwise unnoticed pest blackfly *Simulium vittatum* (Zetterstedt) that grew to nuisance population levels. This species is currently absent below the dam, and has reduced in population enough that it is not managed in this area currently.

The South Saskatchewan River is at the northern limit of *E. album's* range in North America and survival would thus already be tenuous as environmental conditions are not optimum relative to waterbodies in the middle of their range (Sweeny and Vannote, 1978). However, the characteristic rapid growth and high production rate of both *E. album* and *H. limbata* in Northern Great Plains ecosystems (Giberson and Galloway, 1985; Heise et al., 1987) likely enable these species to exist in Saskatchewan where annual temperature patterns are characterized by high amounts of thermal energy available for a short period of time through warm mid-summer. The cold waters downstream of Gardiner Dam studied may approach the minimum required number of degree days by 50 and 60 km downstream (Fremling, 1973; McCafferty and Pereira, 1984), but we do not find these species, suggesting that conditions are unfavorable for consistent recruitment between years to maintain a population of these burrowing mayflies. Ultimately, the benthic thermal environment is altered downstream of the dam as it has a lower summer maximum than it otherwise would have achieved as indicated by conditions upstream (SSR 1 - Leader), at the un-impounded reach (NSR 1 - North Saskatchewan River at Borden) or far-field (SSR 16 - South Saskatchewan River at Clarkboro Ferry).

The use of degree day requirements for growth and support of a species enables comparison between separate populations, and provides a metric for thermal disturbance. For example, 10 ° C is widely accepted as the threshold temperature required by *Hexagenia* for growth in its life cycle, but the number of degree days required by this taxon are noted to decline with increasing latitude (Giberson and Rosenberg, 1994). Giberson and Rosenberg (1994) point out that the range in degree day requirements span from greater than 2,500 in the southern United States (Hudson and Swanson, 1972) to only ~1,300 degree days in their study of South Indian Lake in northern Manitoba. Accordingly, it is most appropriate to compare the potential for Gardiner Dam to support *Hexagenia* to studies conducted at comparable latitudes; Dauphin Lake, Manitoba in particular (~1,900 degree days, Heise et al., 1987). As such, management of the Gardiner Dam would necessarily require increasing the instream temperature to sustain an additional ~200 degree days, or days with temperatures above 10 ° C within a two to four year period (estimated from Heise et al., 1987; Giberson and Rosenberg, 1994). Further, recorded minimum temperatures in a controlled laboratory setting necessary for the emergence of *H. limbata* are 12.0 – 14.5 ^{o}C (Fremling, 1973; McCafferty and Pereira, 1984) which is much greater than temperatures achieved here.

Additionally, achieving the temperature requirements for egg hatch of the native crayfish *O. virilis* (11 ^{0} C; Weagle and Ozburn, 1972) would be challenging downstream of the Gardiner Dam, as temperature for hatch here is delayed nearly 2 months relative to upstream or reference conditions. In contrast, however, the temperature at this time is optimal for the hatch of the invasive *O. rusticus* which can hatch at temperatures as low as 4 ^{0}C (Berrill and Arsenault, 1982).

Implications for the reduction in mayfly taxa and increase of midges may be most important in their loss as forage resources for fish in the South Saskatchewan River. Midges are common in large rivers and lakes around Saskatchewan (AquaTax, 2009), and are food for juveniles of walleye (*Sander vitreus* [Mitchill]), pike (*Esox lucius* Linnaeus), and lake sturgeon (*Acipenser fluvescens* [Rafinesque]). Although we currently find a great density of midges downstream of the Gardiner Dam, we do not have a measure of their value as a forage resource and overall biomass relative to what would naturally occur. In particular, further research should target quantifying the secondary production in these reaches to estimate ultimate change (positive or negative) in the quality of this habitat for fish.

Despite the impact of the Gardiner Dam on benthic communities in the current study, restoration of aquatic ecosystems downstream of reservoirs have been successful in other jurisdictions where increases in minimum flows improved benthic macroinvertebrate community structure reflecting un-impacted condition (Bednarek and Hart, 2005). In the current study it is difficult to discern the relative significance of decreased temperature, reduced sediment load and high flow fluctuation on the macroinvertebrate assemblage. However the presence of stenothermic midges (e.g., *Paracladius* sp., *Orthocladius* sp.) and the complete absence of temperature sensitive insects such as the burrowing mayflies (e.g., *Hexagenia limbata*, and *Ephoron album*) or crayfish suggests that temperature does play a significant role in structuring the patterns we observe; particularly since the tailwater maintains wetted width and does not dry during periods of low flow.

Obviously there are other factors besides temperature such as flow that changed with the construction of the Gardner dam. This makes assigning causation difficult, but based on the criteria proposed by Collier (2003) we have considerable evidence that the observed changes stem from a change in the thermal environment. Specifically, the strength of the association between the dam and the differences in the benthic community is large, and this relationship between dam-related cold-water stress has been demonstrated for many of these taxa in other studies (e.g., Lehmkuhl, 1972). Further, we observe that as the magnitude of the cold water stress decreases through SSR 5 to Saskatoon in the South Saskatchewan River the benthic community returns to reference condition, and ultimately, these results have a credible biological basis in which we demonstrate the mechanistic linkages between life history and depressed temperature.

In order to mitigate the impact Gardiner Dam has on the temperature of the South Saskatchewan River the tailwater temperature would need to be raised ~9 °C from the beginning of May to the end of July from its current regime. Based on continuous lake temperature profile data collected as part of a water quality study of Diefenbaker Lake in the 1980's (Saskatchewan Environment & Public Safety and Environment Canada, 1988), changing the level at which water is drawn through the turbines at Gardiner Dam will not have the potential to restore the natural temperature regime in the South Saskatchewan River. In particular, even if release from Gardiner Dam was from the surface of Diefenbaker Lake, maximum temperatures only reach ~20 ^{0}C, and only reach this temperature by the beginning of August when temperatures in the river too return to the natural regime. The important period of temperature difference between reference temperature regimes and that of the downstream sections of the South Saskatchewan River is between May and mid-July when the decreased temperature causes the biological differences observed in the current study. As such, we do not recommend altering the regime of discharge as it would not benefit the South Saskatchewan River and may in fact have detrimental impacts on the unique stenothermic benthic communities that are establishing in the tailwater of the Gardiner Dam.

Acknowledgements

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CHAPTER 4: MACROINVERTEBRATE COMMUNITIES IN A NORTHERN GREAT PLAINS RIVER ARE STRONGLY SHAPED BY NATURALLY OCCURRING SUSPENDED SEDIMENTS: IMPLICATIONS FOR ECOSYSTEM HEALTH ASSESSMENT

Produced in:

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Abstract:

Rivers are typified by considerable seasonal flow variability. For rivers that flow through alluvial deposits, fine sediment (<63μm) is readily suspended, especially during periods of high discharge. Assessment of the impacts to biota by anthropogenic stressors must therefore occur within the context of dynamic turbidity and background flow conditions. Using the Qu'Appelle River as a study system in southern Saskatchewan, we develop a model in which discharge is a principal determinant of in-stream suspended sediment. This relationship was explored with a case study showing that macroinvertebrate community structure is strongly correlated with suspended sediment gradients and ultimately predicted by discharge. Factors affecting sediment loads and ecosystem responses in managed systems should be considered so that in-stream water quantity and quality needs are met. This new understanding should allow for the development of improved ecosystem based flow management objectives.

Key words: macroinvertebrates, discharge, Northern Great Plains, turbidity, non-metric multidimensional scaling, rivers, total suspended solids

Introduction

Use of benthic macroinvertebrate communities as a measure of riverine health requires an understanding of the physical factors structuring those communities in the absence of human effects. Moreover, the range of abiotic variables (e.g., water chemistry, substrate type, sediment mobility, temperature range, and flow patterns) encountered in some regions, including the Northern Great Plains, differ from those in other regions and these differences result in distinctive macroinvertebrate communities (Matthews 1988). In the Northern Great Plains, turbidity, temperature, and discharge are highly variable, and benthic habitats are typically silt-dominated, whereas forested rivers and streams of neighboring landscapes have more stable in-stream habitats and greater benthic complexity (Matthews 1988). Lakes interrupt the continuum of some Northern Great Plains rivers and can lead to longitudinal re-setting of lotic ecosystems by returning the in-stream environment to one with lower turbidity, temperature, sediment deposition and less variable flow (Jones 2010). These lakes are sediment sinks (Dorava and Milner 2000, Arp et al. 2007), and their outflow has relatively low sediment content, which results in greater light penetration and increased primary production below than above the lake. Therefore, the relative influence of physical characteristics, such as suspended sediment, on the structure of benthic macroinvertebrate communities must be taken into account when developing measures of ecosystem health for Northern Great Plains rivers.

Total suspended solids (TSS) strongly affect benthic macroinvertebrate communities (Roy et al.

2003, Freeman and Schorr 2004, Chatzinikolaou et al. 2006). Elevated TSS can lead to increases in pollution-tolerant species (Chatzinikolaou et al. 2006), reductions in diversity (Roy et al. 2003, Evans-White et al. 2009), and decreases in pollution-indicating metrics, such as % Ephemeroptera, Plecoptera, Trichoptera (% EPT) (Freeman and Schorr 2004). Evans-White et al. (2009) found that turbidity reduced species richness of all feeding modes, but suggested that community responses to nutrients and turbidity differed so that disturbance-related metrics could be interchanged between these 2 potential anthropogenic perturbations. Studies of Great Plain macroinvertebrate communities suggest that traditional indicators for assessing perturbation by suspended sediment may be inappropriate in these rivers because they naturally support taxa (Phillips et al. 2008) that are more tolerant to TSS (Hoemsen 2015) than taxa in rivers with less variable temperature, discharge, and water chemistry (Matthews 1988).

Benthic macroinvertebrate-based biomonitoring programs designed to assess ecosystem health and evaluate human effects must be based on realistic expectations of metric responses that reflect human alteration (Dolédec et al. 1999). General indices, such as benthic macroinvertebrate abundance or taxonomic richness, can vary depending local physical habitat regardless of human activity (Statzner and Sperling 1993, Lenat 1993). For example, modified biomonitoring approaches for application to lowland streams in The Netherlands were needed because of underlying differences in the macroinvertebrate communities that comprised the diagnostic metrics (Tolkamp 1985). Reference-condition-based biomonitoring programs typically rely on expected ranges of invertebrate metrics (e.g., EPT). However, concentrations of TSS are highly variable across Northern Great Plains rivers (Matthews 1988), and most macroinvertebrate metrics have been developed in areas other than the Great Plains (e.g., Barbour et al. 1999) where natural TSS is lower and species are less tolerant of high TSS. Before community composition metrics used to evaluate human effects can be redefined for Northern Great Plains rivers, basic ecological data are needed to understand how benthic communities in this region are structured by natural processes.

We characterized the relationships between TSS and turbidity and used the discharge–turbidity relationship to evaluate the effect of turbidity on benthic macroinvertebrate communities to understand site-specific responses to discharge management practices. We hypothesized that turbidity explains variation in macroinvertebrate abundance, diversity, dominance, commonly-used metrics of biotic integrity, and the structure of benthic macroinvertebrate assemblages in Northern Great Plains rivers.

Methods

Study area

We conducted our study along ~600 km of the Qu'Appelle River in south-central Saskatchewan (Fig. 4.1; see Wiens 1987, Hall et al. 1999, Quinlan et al. 2002 for detailed descriptions) in the Moist Mixed Grassland and the Aspen Parkland ecoregions. The Qu'Appelle River flows in a glacial river valley formed through the Alluvial Plain during the retreat of the Wisconsin glacier ~14,000 y ago and the river's order does not change throughout its length. The benthic habitat in the Qu'Appelle River is typified by silt and soft sediment with rare patches of riffle and cobble through its length (Meissner et al. 2016).

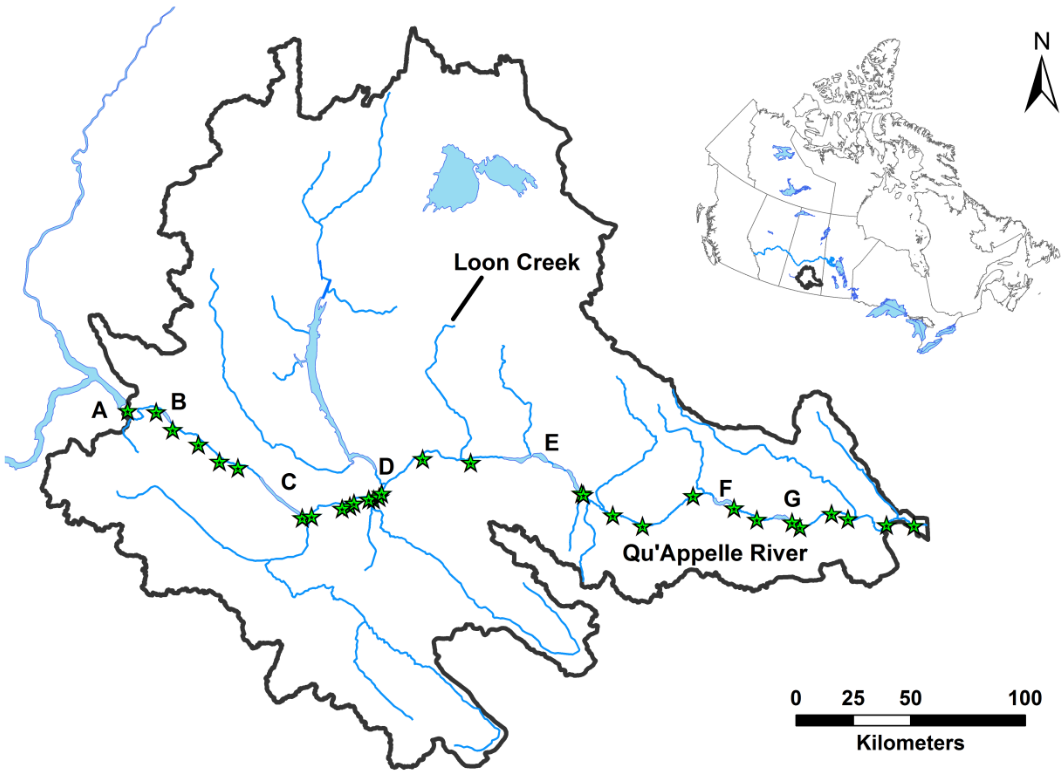

Figure 4.1. Qu'Appelle River watershed in Saskatchewan showing the stations in star symbols sampled during the summer of 2007 and Lake Diefenbaker (A), Eyebrow Marsh (B), Buffalo Pound Lake (C), Craven water control structure/Last Mountain Lake (D), Pasqua – Katepwa Lakes (E), Crooked Lake (F), and Round Lake (G).

Lake levels are managed by control structures at the outlets of waterbodies along the mainstream of the Qu'Appelle River. Flow in the headwaters is augmented by interbasin transfer of water from Lake Diefenbaker. Mean annual discharge received from Lake Diefenbaker in 2006 (the year before our study) was 2.23 m^3/s (Water Survey of Canada site 05JG006). Discharge at the furthest downstream gauging station near the confluence of the Qu'Appelle River with the Assiniboine River was 13.90 m^3/s (Water Survey of Canada site 05JK007).

Physiochemical and habitat variable assessment

We sampled 33 sites, chosen on the basis of distribution down the length of the river and access from road crossings, once between 31 July and 31 August 2007. At each site, we used a rapid habitat assessment approach (MoE and SWA 2012) and recorded habitat characteristics (% cover of macrophytes, vegetated banks, snags, and sediment types; categorical occurrence of algae, woody debris, and detritus based on visual estimates). We measured turbidity (NTUs) with a turbidimeter (model 2020; LaMotte, Chestertown, Maryland). We also obtained data on turbidity (LaMotte 2020 turbidimeter), TSS (dried mass), and discharge (*Q*) from the Water Survey of Canada (http://wateroffice.ec.gc.ca/mainmenu/historical_data_index_e.html) measured on 38 occasions from 2013 and 2015 downstream of the inflow of Loon Creek to better assess the relationships among these 3 variables.

Benthic macroinvertebrate assessment

We used standard D-frame nets (0.30 m, 500 μm mesh) to collect traveling kick-and-sweep samples along a transect at each site. Samples consisted of composite sweeps (10 s/sweep over an ~1 m^2 area) on the left bank, ¼ distance, ½ distance, ¾ distance, and right bank integrated into a single sample per transect. We concentrated samples by pouring them through a 500-μm-mesh sieve and immediately preserved them with 80% ethanol. Organisms were sorted with the aid of a stereoscope at 7× magnification. We subsampled the initial sample when the total number of organisms was estimated to be >1000 individuals by evenly spreading the samples on a 250-µm-mesh sieve and then removing ½ the sample to sort. We multiplied resulting abundances by the fraction removed to estimate original sample abundance. We were unable to use a Marchant-box style subsampler because of large quantities of macrophytes and filamentous algae in many samples. We identified specimens to the lowest possible taxon designation (usually genus and species, but family for Chironomidae, Ceratopogonidae, Stratiomyidae, Tabanidae, and Tipulidae, subcohort for Hydrachnidia, class for Oligochaeta, and phylum for Nematoda) with the aid of keys for North America (Merritt et al. 2008). Voucher series were deposited in both the Water Security Agency Invertebrate Voucher Collection (Saskatoon, Saskatchewan), and the Royal Saskatchewan Museum (Regina, Saskatchewan).

Data analyses

We used benthic macroinvertebrate data to calculate total abundance, taxon richness (number of unique taxa), Shannon diversity, Berger–Parker dominance, % EPT, habit groupings (% burrowers, sprawlers, clingers, climbers, swimmers; Merritt et al. 2008), and functional feeding groupings (% collector gatherers, predators, scrapers, filterers, detritivores, shredders; Merritt et al. 2008). We used generalized linear models (GLMs) to assess the relationships of these metrics to turbidity at each site. We log(x + 1)-transformed relative abundance data before calculating the taxon × taxon dissimilarity matrix (see below) of all benthic taxa. The relationships between turbidity and Shannon diversity and Berger–Parker dominance data both appeared curvilinear in preliminary inspection of variables, so we analyzed these metrics with a quadratic rather than a linear term in subsequent regressions.

We used nonmetric multidimensional scaling (NMDS) to visualize compositional changes in the benthic macroinvertebrate community along the gradient of turbidity. First, we selected the optimal distance measure based on highest r^2 of nonmetric fit between various approaches, e.g., Mahalanobis, Bray–Curtis, Gowers, and Kulczynski. Bray–Curtis distance metric was the optimal distance linking metric. We considered our NMDS solution to be a good representation of the community structure if stress was <0.10, but acceptable if it had a final stress <0.18 (Clarke and Warwick 2001). We chose the optimal number of dimensions to apply by including subsequent dimensions until we arrived at a solution explaining >85% of the variance and further addition of dimensions provided small reductions in stress (McCune and Grace 2002). The most stable solution of the NMDS analysis (stress = 0.16) of the abundant taxa (>5 individuals = 132 taxa) consisted of 2 dimensions.

All data analyses were done in R (version 3.0.1; R Project for Statistical Computing, Vienna, Austria) with the *glm*, *vegan* and *MetaMDS* packages.

Results

Discharge–turbidity relationship

Turbidity ranged between 0 and 78.2 NTU (Fig. 4.2). Turbidity was lower immediately downstream than upstream of lakes, but increased with distance from each lake until the river reached the next lake. Based on samples collected between 2013 and 2015 near Loon Creek in the Qu'Appelle River, turbidity and TSS (n = 38, r^2 = 0.92, p < 0.001; Fig. 4.3A) and discharge and turbidity (n = 38, r^2 = 0.54, p < 0.001; Fig. 4.3B) were significantly positively related. Therefore, we were able to use turbidity alone in our analyses with benthic macroinvertebrate metrics.

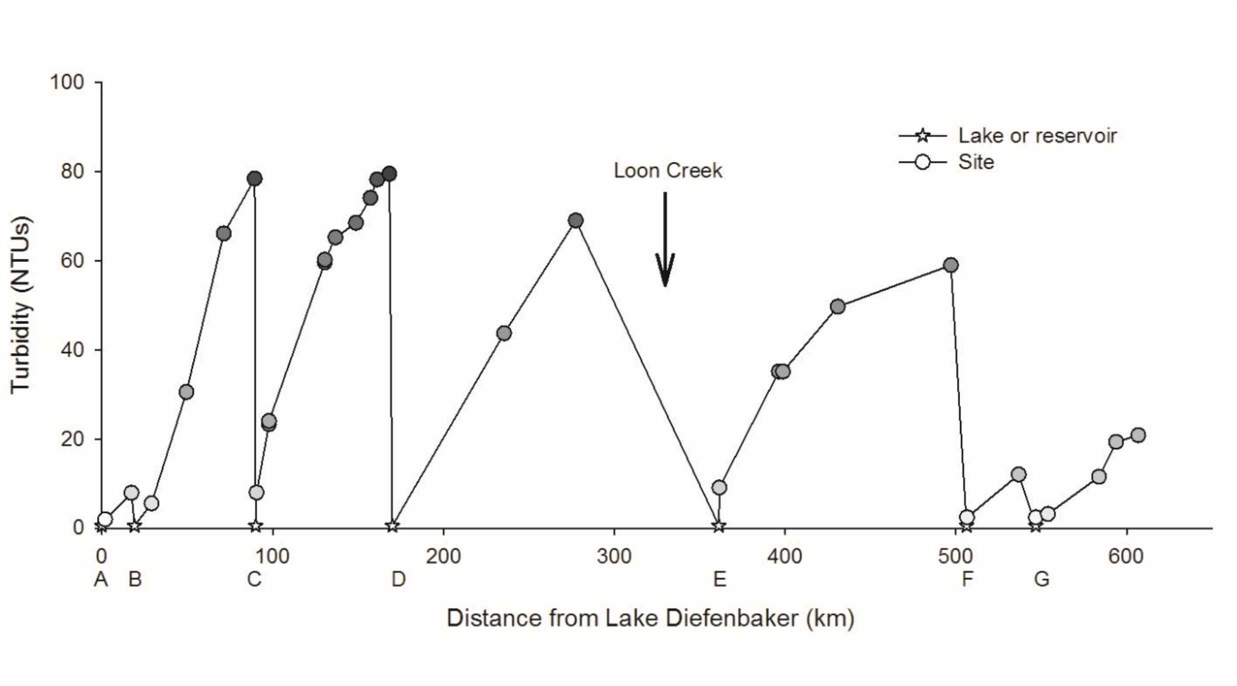

Figure 4.2. Turbidity and river distance of sites down the length of the Qu'Appelle River from its source at Lake Diefenbaker. Star symbols indicate position of a river-interruption at Lake Diefenbaker (A), Eyebrow Marsh (B), Buffalo Pound Lake (C), Craven water control structure/Last Mountain Lake (D), Pasqua – Katepwa Lakes (E), Crooked Lake (F), and Round Lake (G). Symbol shading indicates relative turbidity levels for graphical presentation and opacity is calculated as a proportion of highest turbidity measured in the study with highest level shaded black (e.g., 78.2 NTU = 100% opaque).

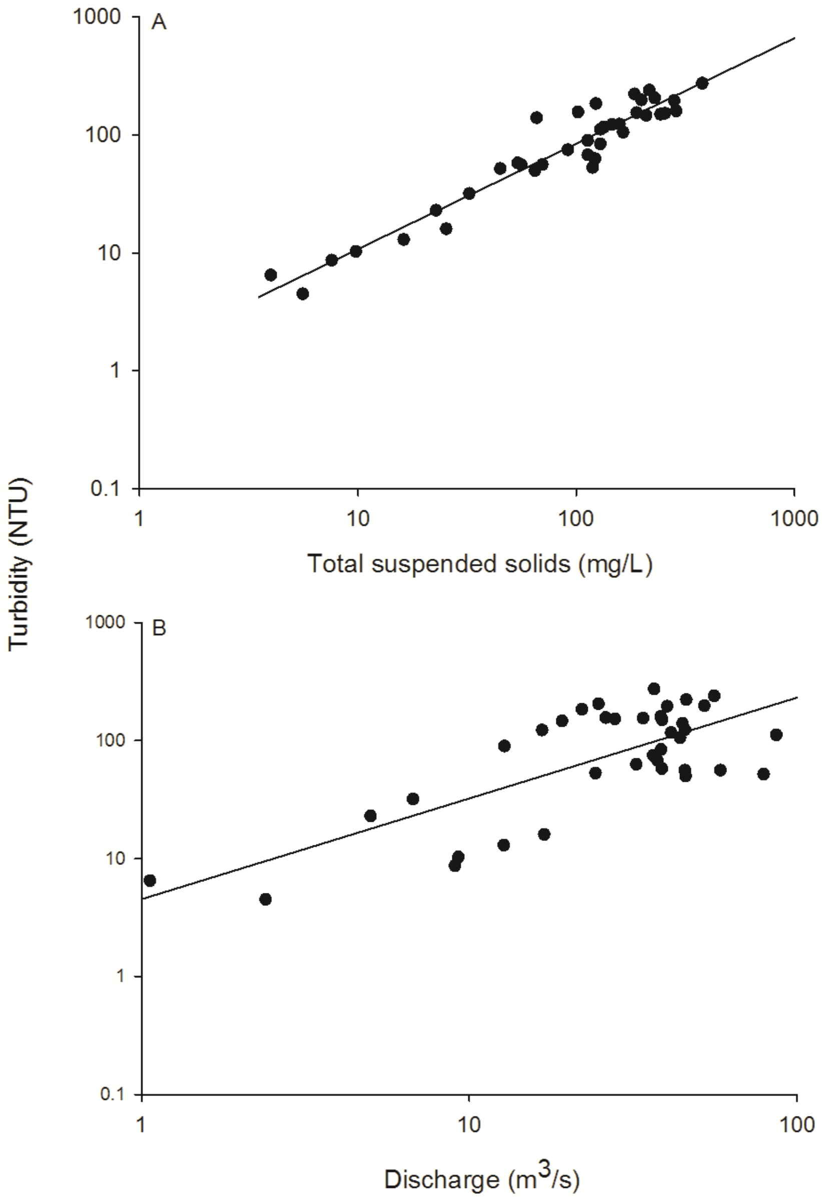

Figure 4.3. Relationship between turbidity (NTU's) and total suspended solids (mg/L; A) and turbidity (NTU's) and discharge (m^3/s; B) in the Qu'Appelle River downstream of Loon Creek from 2013 to 2015 (n = 38).

Benthic macroinvertebrate metrics

We collected 107,616 individuals representing 171 taxa. *Hyalella azteca*, *Gammarus lacustris*, Chironomidae, and *Simulium vittatum* were highly abundant at low-turbidity sites. *Hyalella azteca* and *G. lacustris* were the only amphipod species found, but we identified 13 distinct dipteran taxa. Other macoinvertebrate taxa with abundances >1000 individuals were *Physa* sp., Oligochaeta, Baetidae, *Hexigenia limbata*, Corixidae, and *Hydropsyche* sp. Hemiptera had the highest species richness (21 species, 20 in the family Corixidae). Ephemeroptera was represented by 18 unique taxa, Trichoptera by 16 taxa, and Plecoptera by 15 individuals in 2 species (*Acroneuria abnormis*, *Pteronarcys dorsata*). Coleoptera were represented by 17 unique taxa, Gastropoda by 11 taxa, and the Hirudinea by 9 taxa (Table S1; http://www.journals.uchicago.edu/doi/suppl/10.1086/689013).

Turbidity and total macroinvertebrate abundance were negatively related (Table 4.1). The relationship between Shannon diversity and turbidity was a negative quadratic function, with low diversity at low and high levels of turbidity (Fig. 4.4A, Table 4.1). However, the relationship between Berger–Parker dominance and turbidity was a positive quadratic function, with high dominance at low and high levels of turbidity (Fig. 4.4B, Table 4.1). *Hyalella azteca* and *G. lacustris* abundance were negatively related to turbidity (Fig. 4.5A, B, Table 4.2). Abundance of sediment-burrowing Ephemeroptera (Ephemeridae) was positively related to turbidity (Fig. 4.5C, Table 2), whereas abundance of Baetidae was negatively related to turbidity (Fig. 4.5D, Table 4.2). Abundances of Corixidae and Chironomidae were negatively related to turbidity (Fig. 4.5E, F, Table 4.2).

Table 4.1. General Linear Model of the effects of turbidity on common benthic macroinvertebrate metrics in Qu'Appelle River

Metric	**Response**	**df**	**MS**	**F**	***P***	**r2**
Total Abundance	-	1, 32	6.90	35.63	< **0.001**	**0.54**
Taxa Richness	No Change	1, 32	179.39	2.57	0.129	0.08
Shannon's Diversity *	- at low and high	1, 32	1.17	5.13	**0.012**	**0.25**
Berger-Parker Dominance *	+ at low and high	1, 32	0.15	6.26	**0.005**	**0.29**
% EPT Taxa	+	1, 32	12064.00	9.78	0.004	0.24
Habit Groupings						
% Burrowers	+	1, 32	9123.90	21.36	< **0.001**	**0.41**
% Sprawlers	No Change	1, 32	10.30	0.82	0.374	0.03
% Clingers	No Change	1, 32	2.30	0.00	0.951	0.00
% Climbers	No Change	1, 32	5.43	4.08	0.052	0.12
% Swimmers	-	1, 32	9777.40	19.75	< **0.001**	**0.39**
Functional Feeding Groupings						
% Collector Gatherers	No Change	1, 32	748.10	1.16	0.290	0.04
% Predators	No Change	1, 32	16.67	1.67	0.205	0.05
% Scrapers	No Change	1, 32	173.00	0.52	0.477	0.02
% Filterers	No Change	1, 32	375.80	0.89	0.354	0.03
% Detritivores	+	1, 32	1493.30	10.73	**0.003**	**0.26**
% Shredders	No Change	1, 32	0.95	1.59	0.217	0.05

* indicates a quadratic relationship

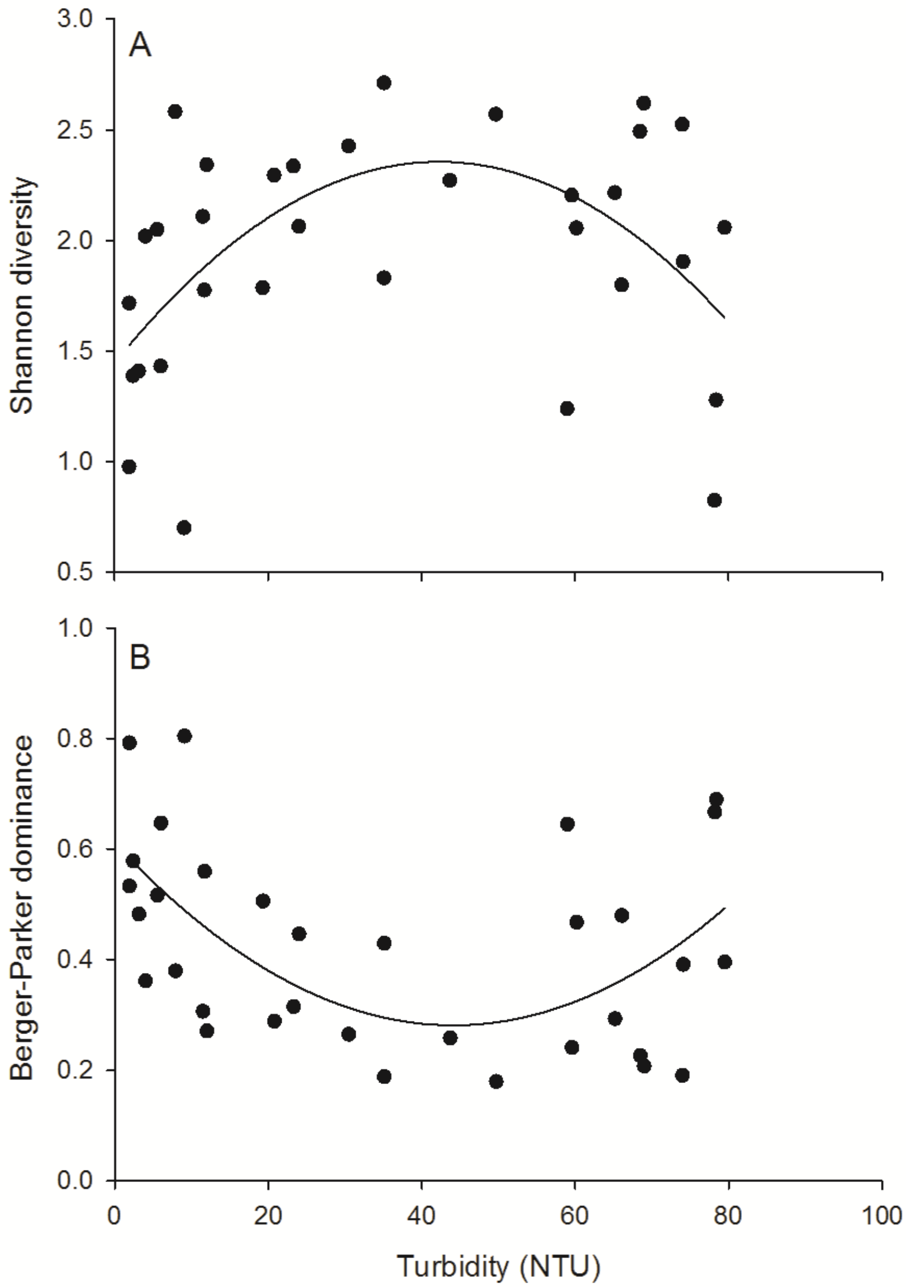

Figure 4.4: Quadratic relationships between Shannon's Diversity (A) and Berger-Parker Dominance (B) along a gradient of turbidity.

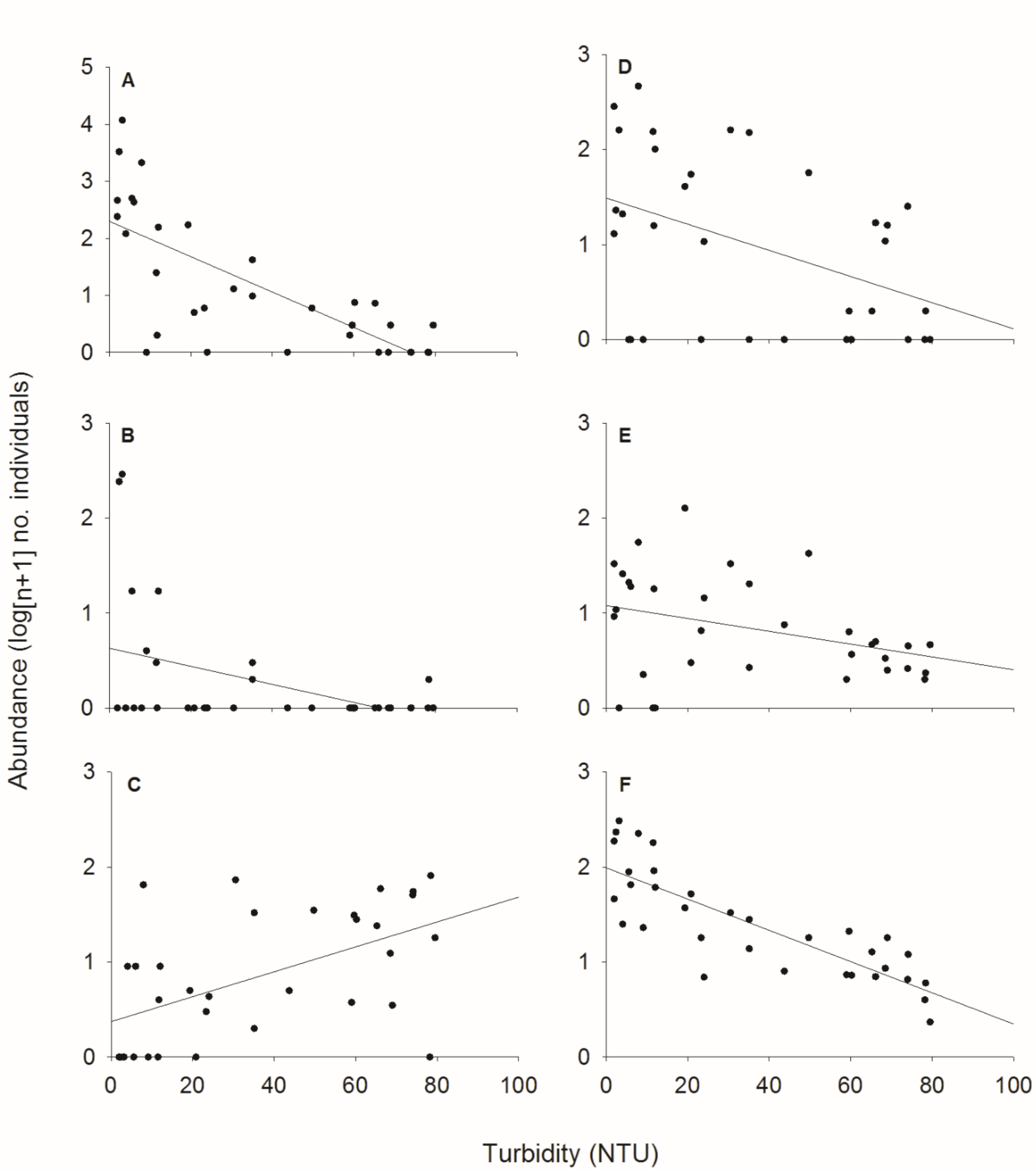

Figure 4.5. Relationships between log(x + 1)-transformed total abundance of *Hyalella azteca* (A), *Gammarus lacustris* (B), Ephemeridae (C), Baetidae (D), Corixidae (E), and Chironomidae (F) and turbidity. Linear relationships are shown where significant. Note the different log abundance scale for *H. azteca.*

Table 4.2. General Linear Model of the effects of turbidity on common benthic macroinvertebrate metrics of the Qu'Appelle River

Taxa	**Response**	**df**	**MS**	**F**	***P***	***r*²**
Oligochaeta	No Change	1, 32	0.413	0.98	0.330	0.03
Dogielinotidae						
Hyalella azteca	-	1, 32	26.980	47.32	**< 0.001**	**0.60**
Gammaridae						
Gammarus lacustris	-	1, 32	2.325	6.72	**0.010**	**0.17**
Hydropsychidae	No Change	1, 32	0.501	1.21	0.280	0.03
Baetidae	-	1, 32	4.872	7.10	**0.010**	**0.19**
Caenidae	No Change	1, 32	0.982	2.49	0.120	0.07
Ephemeridae	+	1, 32	4.412	12.98	**0.001**	**0.30**
Corixidae	-	1, 32	1.170	4.42	**0.040**	**0.12**
Chironomidae	-	1, 32	6.944	68.75	**< 0.001**	**0.69**

Macroinvertebrate community structure and turbidity were strongly related (Fig. 4.6). Approximately 60% of the variation in axis 1 and 46% of the variation in axis 2 of the NMDS were explained by turbidity (r^2 = 0.59, $F_{1,32}$ = 44.29, p < 0.001; r^2 = 0.46, $F_{1,32}$ = 26.15, p < 0.001; respectively).

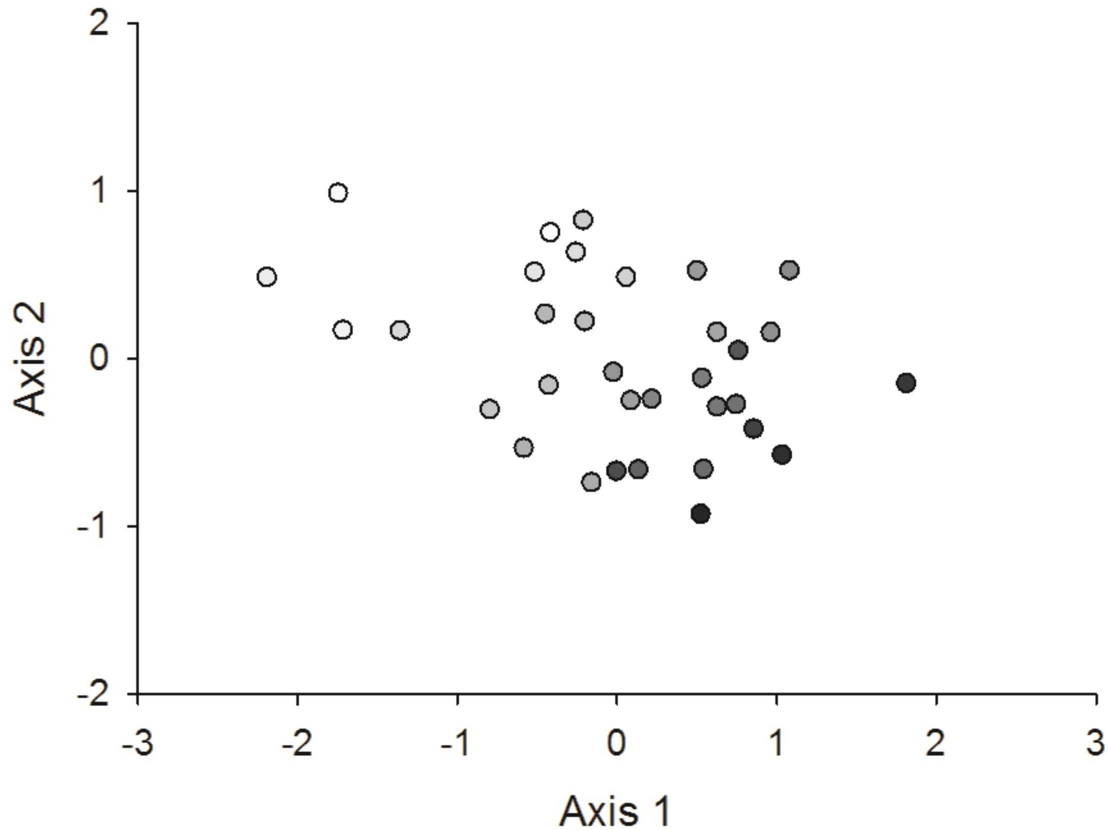

Figure 4.6. Site-level nonmetric multidimensional scaling (NMDS) ordination of benthic macroinvertebrate taxa. Axes 1 and 2 illustrate change in community structure with increasing turbidity. Symbol shading indicates relative turbidity levels for graphical presentation and opacity is calculated as a proportion of highest turbidity measured in the study with highest level shaded black (e.g., 78.2 NTU = 100% opaque).

The macroinvertebrate metrics, % burrowers, % swimmers, % detritivores, and % EPT also were strongly related to the turbidity gradient (Table 4.1). Percent burrowers had a positive relationship with turbidity, whereas % swimmers had a negative relationship with turbidity (Fig. 4.7A, B, Table 4.1). Percent detritivores and % EPT had positive relationships with turbidity (Fig. 4.7C, D, Table 4.1).

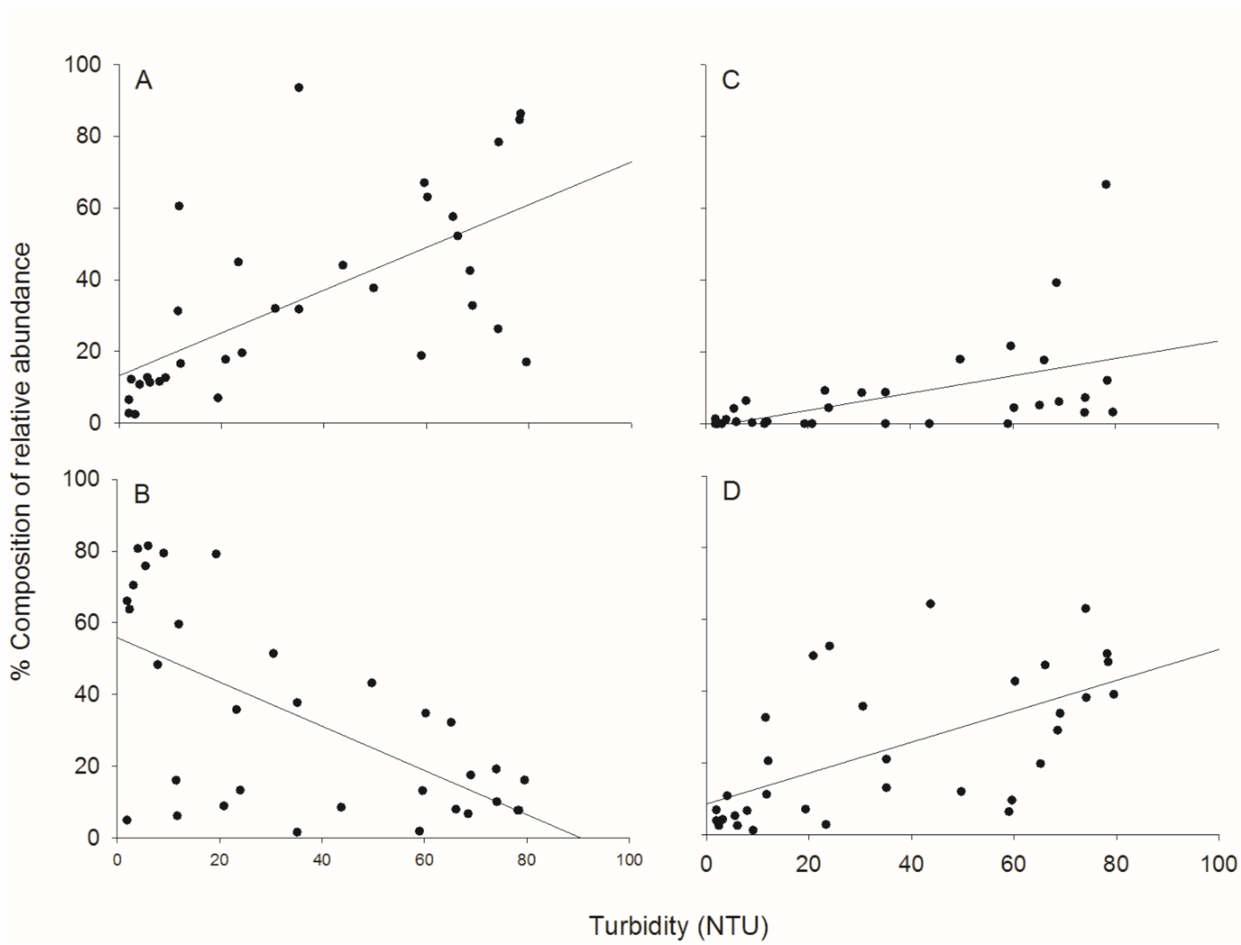

Figure 4.7. Relationships between relative abundance of burrowers (A), swimmers (B), detritivores (C), and Ephemeroptera, Plecoptera, and Trichoptera (EPT) (D) and turbidity in the Qu'Appelle River.

Aquatic macrophytes and algae

Percent cover of macrophytes was high at low-turbidity sites, but macrophytes were practically absent at sites where turbidity was >20 NTU (Fig. 4.8A). Benthic algae were abundant at low-turbidity sites and largely absent at high-turbidity sites (Fig. 4.8B). Algae were not abundant at any site where turbidity was >40 NTU.

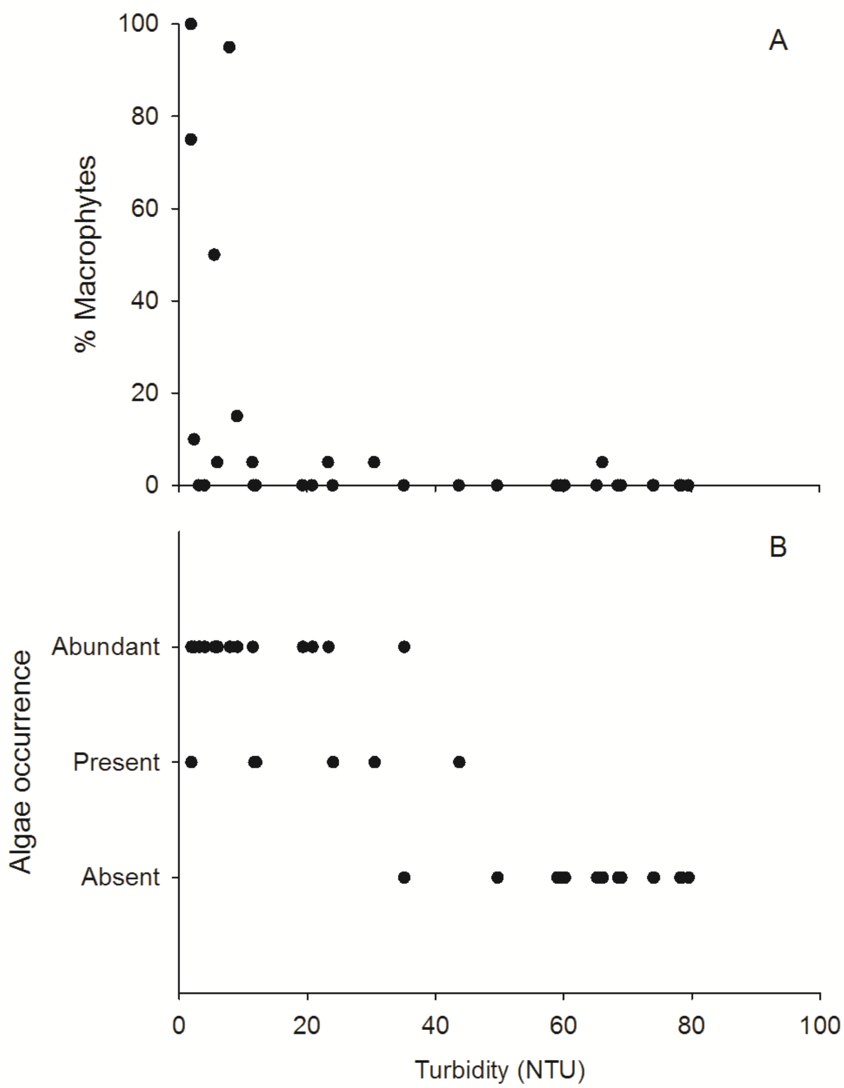

Figure 4.8. Scatterplots showing % cover of macrophytes (A) and algal abundance category (B) and turbidity at each site based on rapid site assessment.

Discussion

Relationships between turbidity, primary productivity, and benthic communities

The effects of turbidity on macroinvertebrate communities typically are attributed to the influence of turbidity on light (Jones 2010). In lotic systems, high turbidity limits light penetration and reduces primary production of periphyton (Steinman and McIntire 1990), phytoplankton (Hoetzel and Croome 1994), and macrophytes (Lloyd et al. 1987), thereby limiting productivity because less light is available for photosynthesis (Ryan 1991). Lloyd et al. (1987) found that increases of only 5 NTUs decreased primary production by 3 to 13% and that increases of 25 NTUs could reduce primary production up to 50%. Reduced production at the base of an autochthonous food web creates cascading effects through the food web and an overall decrease in energy available to the ecosystem. Kinetic effects of suspended sediment on individuals (Wood and Armitage 1997) also can affect benthic macroinvertebrates directly via physical harm (Rosenberg and Snow 1975, Fairchild et al. 1987). Direct kinetic effects are unlikely in the low-gradient Qu'Appelle River, but the responses we observed are consistent in the context of the light limitation that turbidity imparts to a river.

Abundant algae and macrophytes in low-turbidity conditions could be providing the habitat and foodweb base needed to support abundant *H. azteca*, *G. lacustris*, Baetidae, Corixidae, and Chironomidae. In contrast, the dominance of burrowers, detritivores, and burrowing mayflies at higher turbidities in this soft-bottomed river probably was related to lower primary productivity and lower habitat complexity in the absence of macrophytes and algae. All study sites had large quantities of deposited sediment (>90%), and the benthic community did not change as sediment was increased in experimental manipulations (Hoemsen 2015). Therefore, we infer that the differences in burrowing taxa among sites in our study were not a product of differences in sediment. The increase in % EPT taxa at high turbidities was not a contradiction of the expected decrease in turbidity-sensitive taxa (Freeman and Schorr 2004, Roy et al. 2003, Chatzinikolaou et al. 2006, Evans-White et al. 2009). Rather, it reflected the traits of the taxa included in this metric in Northern Great Plains rivers. For example, Ephemeroptera was represented by high abundances of burrowing mayflies, such as *H. limbata* and *Ephoron album*, or armored Caenidae (~34% of total EPT). Gammon (1970) found similar positive responses by *Tricorythodes* sp. to increasing suspended sediment.

We also found no marked decrease in diversity with increasing turbidity, a result that contrasts with those of other studies in other systems (e.g., Lemly 1982, Quinn et al. 1992, Evans-White et al. 2009). As turbidity increased in the Qu'Appelle River, *H. azteca*, *G. lacustris*, Baetidae, Corixidae, and Chironomidae became less abundant and burrowing mayflies became more dominant. Shannon diversity peaked at intermediate values of turbidity despite a strong change in community composition, and turbidity explained only ~25% of the variation in Shannon diversity. Thus, other environmental variables appear to be causing most of the variation in Shannon diversity. We propose that naturally turbid rivers, such as the Qu'Appelle, in the Northern Great Plains have a unique community structure whereby taxonomic diversity initially increases with turbidity (~0–50 NTUs) before decreasing at high levels (~60–78 NTUs), with a corresponding change in taxonomic composition across the turbidity gradient.

Turbidity controls primary production and biomass (Cloern 1987, Hall et al. 2015), and lakes and reservoirs reducing sediment load at their outflow (Jones 2010). Our results in the Qu'Appelle

River were consistent with this expected pattern. Wooton et al. (1996) and Myers et al. (2007) reported that the stable flow regimes and low sediment load in lake outflows reduce periphyton scouring and allow accumulation of biofilm and, ultimately, more food and habitat for benthic macroinvertebrates (Dorava and Milner 2000). Our results in the Qu'Appelle River also were consistent with this expected pattern in that we found high % cover of macrophytes and algae under the low turbidity conditions in outlet stretches of the Qu'Appelle River.

The discontinuum model proposed by Ward and Stanford (1995) and described by Jones (2010) predicts that the influence of lakes on a river decreases in the downstream direction. Our results support this suggestion that the influence of lentic systems interrupting a river continuum decrease with distance downstream from each lake as the river increasingly resembles a turbid system. The order of the Qu'Appelle River does not increase between lakes, so the effects of increased turbidity with distance from waterbodies on the benthic community can be distinguished from the stream-order-related increases in turbidity described in the river continuum concept (Vannote et al. 1980). As in other Northern Great Plains rivers, discharge and suspended sediments in the Qu'Appelle River are positively related (Hansen et al. 2016). Thus, managing stream flows or erosion will have direct consequences on downstream turbidity in these rivers. Strong TSS–turbidity relationships are common in prairie streams, and turbidity is often used as a proxy for TSS (e.g., Evans-White et al. 2009). The series of managed lakes in the Qu'Appelle River allowed us to quantify the influence of discharge on TSS, the relationship between TSS and turbidity, and the interaction between discharge-related in-stream sediment concentrations and benthic macroinvertebrate community structure and function.

Implications for biomonitoring

Biomonitoring programs in which community structure is used to assess human impacts require appropriate benchmark sites (Ode et al. 2016). The underlying turbidity regime of rivers should be considered in the selection of least-impaired benchmark sites and the application of anthropogenic disturbance metrics, especially in large rivers where few sites are in reference or least-impaired condition (Angradi et al. 2009a, b). Appropriate benchmark sites for naturally turbid rivers (e.g., Northern Great Plains Rivers) can be selected by including turbidity regime as an underlying physical feature of a river (in addition to other consequential natural and ubiquitous anthropogenic, e.g., agricultural runoff) when statistically evaluating biological groupings of reference sites (Bowman and Somers 2005). Our results highlight how, in a single river, the underlying position of a site along a discontinuum can affect community structure independently from stream order, soils, surficial geology, ecoregion, and other physical characters that might otherwise be tested as explanatory variables.

Metrics used in Northern Great Plains rivers must be relevant to their naturally occurring taxa. We found that biodiversity was highest at intermediate levels of turbidity and that % EPT, which is traditionally viewed as metric that decreases with increasing sediment load (e.g., Barbour et al. 1999), increased with turbidity because sediment-tolerant burrowing Ephemeridae mayflies increased as swimming organisms (e.g., *H. azteca* and *G. lacustris*) decreased.

In general, our study emphasizes the need to develop ecoregion-specific metrics for biomonitoring, consistent with practice in other jurisdictions, such as the European Union, Australia, and USA (summarized by Hering et al. 2006). Developers of multimetric approaches to macroinvertebrate-based biomonitoring need to include careful evaluation of metric responsiveness for accurate identification of impairment, as has been done with the European Water Directive Framework (Mondy et al. 2012). Considerations should include interactions of variables, such as discharge, nutrients, and organic C with turbidity. Highly turbid river systems have low primary productivity, so the expression of organic-pollution-related metrics may be less pronounced with increasing sediment resuspension and increasing discharge. However, river reaches at the outflow of lakes in a discontinuum might be more sensitive because they are characterized by low turbidity and higher primary production.

Acknowledgements

Field collection of the benthic macroinvertebrates and habitat variables over the course of 2008 were primarily performed by Kevin Kirkham and Deanne Schulz. Janet Halpin helped with sorting, and Dale Parker of AquaTax (Saskatoon, Saskatchewan) provided the bulk of the taxonomic identification. We thank Glen McMaster and 2 anonymous referees for providing comments on earlier drafts of this manuscript. Funding for the field work associated with this project was provided by the Water Security Agency of Saskatchewan Water Sustainability Fund and a grant from the Prairie Adaptation Research Collaborative. Support for manuscript preparation was provided by a Natural Sciences and Engineering Research Council PGS-D scholarship to IDP.

CHAPTER 5: SASKATCHEWAN CONDITION ASSESSMENT OF LOTIC ECOSYSTEMS: A MULTIVARIATE TOOL FOR ASSESSING THE INTEGRITY OF NORTHERN GREAT PLAINS WADEABLE RIVERS AND STREAMS

Abstract

The current study develops a multivariate and predictive model based on the reference condition approach for the Northern Great Plains region of Saskatchewan. Benthic macroinvertebrate communities and environmental abiotic data were collected at 280 reference sites and 10 test sites (waste water [n=3], reservoir [n=5], urban [n=1], and crop [n=1]). Reference sites were classified into groups characterized by similar macroinvertebrate communities. Three classifications were developed, using lowest possible taxonomic designation (typically genus). Benthic macroinvertebrate communities were reduced to descriptive metrics (total abundance, number of Coleoptera, % EPT, and % shredders) and related to the first axis of community ordination structure, then correlated to each other to simplify non-redundant metrics. Abiotic descriptors were systematically reduced through correlation analysis, and discriminant function analysis was used to develop a predictive model for reference sites based on stream order and ecoregion. This model predicted 68.7% of the sites correctly using cross-validation. Of the 10 test sites, two were stressed (one waste water and one urban site) while three were classified as impaired (one waste water and two reservoirs). This model is an effective tool that provides a practical means of evaluating biotic condition of streams in the Northern Great Plains.

Introduction

Communities of aquatic organisms and the traits they possess are often used to evaluate the biological condition of freshwater habitats as they are responsive to changes in environmental characteristics and express the ultimate ecological consequences of anthropogenic perturbation (Rosenberg and Resh 1993). Benthic macroinvertebrates in particular have been used in biomonitoring studies since the early 1900's when summary metrics of their sensitivity to organic pollution were incorporated into a European focused Saprobien system of river condition (Kolkowitz and Marsson 1908). Since this initial application many multimetric (e.g., Karr et al. 1986, McCormick et al. 2001, Klemm et al. 2003), Nearest Neighbour Analysis (e.g., Sarrazin-Delay et al. 2014), Bayesian (e.g., Qian et al. 2003), and multivariate (e.g., Wright et al. 1983, Hawkins et al. 2000, Reynoldson et al. 2001, Bowman and Somers 2006) approaches to biomonitoring have been developed and applied globally with the intention of defining thresholds of ecological impairment. However, regardless of method used, the definition of biological condition and the ability to assess impairment from healthy condition relies on an understanding of what community of organisms and their traits are expected in the absence of human activity (Ode et al. 2005). This reference condition is fundamentally important in the construction of any biomonitoring approach (Bowman and Somers 2005).

The Northern Great Plains in western Canada possess a landscape dominated by human activity, and the potential anthropogenic perturbation on lotic aquatic ecosystems lacks quantifiable tools for evaluation. Phillips et al. (2015) designed and applied a Test Site Analysis-based method for evaluating biological integrity in large river systems of the Northern Great Plains, but the methods and model provided are inappropriate for evaluating wadeable streams.

The current study presents a benthic macroinvertebrate-based multivariate biomonitoring tool based on least-impacted reference condition approach and Test Site Analysis condition assessment for wadeable rivers and streams in the Northern Great Plains. The steps in its development are as follows:

1) Characterize scope of inference for what type of waterbodies and human stressors are to be included in the study.
2) Second, it is necessary to identify the desired condition to be applied in evaluating waterbodies under human stress. If a biomonitoring tool is to be defensible in setting site specific objectives for community composition and biological end-points, the biological condition expected in the absence of human activity needs to be clearly and defensibly defined *a priori* (Stoddard et al. 2006).
3) Once reference condition has been established, the third step in this process is to classify biological communities at reference sites and determine what underlying abiotic characteristics best discriminate between classifications. This step is necessary in order to match test sites to groups of reference sites enabling comparison of communities that should be similar but for the anthropogenic stressor being investigated (Bowman and Somers 2005).
4) Next, summary metrics of the benthic macroinvertebrate community that capture significant community structure are summarized and reduced using correlation analysis.
5) Finally, test sites are designated to respective reference groupings using predictive membership assignment, and evaluated for biological condition using the metrics identified above and Test Site Analysis.

The objective of this study was to apply a reference condition approach using benthic macroinvertebrate communities to develop an ecosystem health monitoring tool for Northern Great Plains streams that can be applied to assess the effects of anthropogenic perturbation. This study evaluates the hypothesis that abiotic characteristics of rivers and streams can explain significant variation in the benthic macroinvertebrate communities present at reference sites of least human activity. Further, I hypothesize that these abiotic characteristics can then be used to assign test sites to appropriate reference groupings, and evaluate biological condition based on community metrics.

Methods

Study region

I selected sites in a stratified random design to cover the range of conditions in southern Saskatchewan (Fig. 5.1). First, I expanded the potential reference sites to cover all ecoregions to ensure inclusion of the range of landscape and climatic conditions. Overall, I conducted this study in the Northern Great Plains region of southern Saskatchewan, within the Cypress Upland, Mixed Grassland, Moist Mixed Grassland, Mid-Boreal Lowlands, Mid-Boreal Uplands, Aspen Parkland, and Boreal Transition ecozones (see http://www.biodiversity.sk.ca/eco.htm).

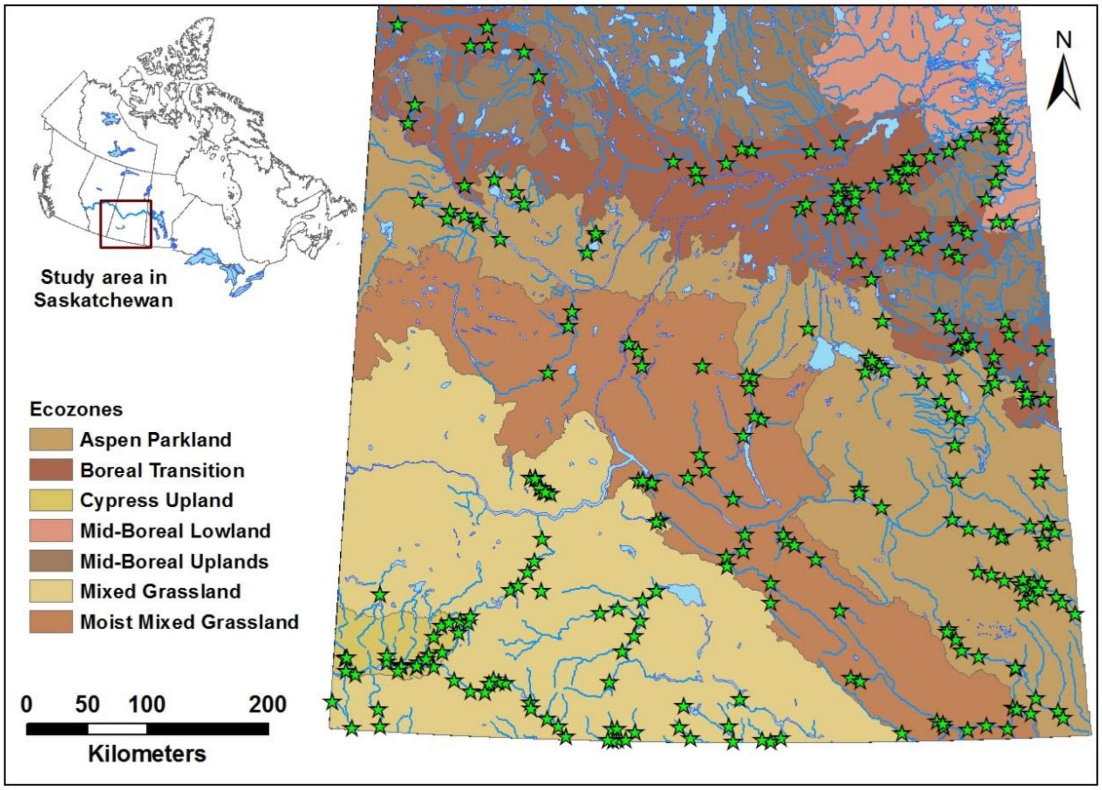

Figure 5.1. Study sites in southern Saskatchewan.

Second, I selected sites covering the range of stream order (Strahler 1964) from first order streams to seventh order streams. The large rivers of southern Saskatchewan are covered in Phillips et al. (2015). The extent of each stream type in southern Saskatchewan is estimated below (Saskatchewan Ministry of Environment 2006):

1) Ephemeral streams: Classified here as those of stream order 0 – 1, comprising 83,086 km of Saskatchewan south of and including the Boreal Transition Ecozone.

2) Temporary streams: Classified here as those of stream order 2 – 4, comprising 75,297 km of Saskatchewan south of and including the Boreal Transition Ecozone.
3) Perennial rivers. Classified here as those of stream order 5 – 7, comprising 15,140 km of river length in Saskatchewan south of, and including, the Boreal Transition Ecozone.
4) Large rivers: Classified here as those of stream order 8, comprising 1,214 km of river length in Saskatchewan south of and including the Boreal Transition Ecozone.

As such, the current study estimates stream health inferred to an estimated (83,086+75,297+15,140)/(83,086+75,297+15,140+1,214)=99.3% of the stream length in southern Saskatchewan.

I had two steps to site selection. To begin with, a series of consultations with provincial experts identified known least heavily impacted waterbodies and to choose sites that were distributed through as many ecoregions as possible. Reference waterbodies identified for sampling were then outlined on 1:250,000 maps and stream orders identified using the Strahler (1964) method. Two or three potential sites 100 m upstream of a road crossing were located in each potential reference stream within a subcatchment. A potential site was considered to be a stream reach with a longitudinal distance approximately six times its width (Newbury 1984). Only run reaches were sampled as they were the dominant habitat type and riffles were highly uncommon. Where riffles did occur they were typically cattle-crossings that received focused impact from livestock and would not likely reflect the instream conditions of the stream. The next step was field verification involving site visit and site assessment if no discernable impact was identified at the site. A total of 486 sites were visited; however, if sites were dry at time of visit they were not sampled or included further in the study.

Reference and test sites were visited only once, in autumn as it provided fewer populations of rapidly growing and reproducing multivoltine benthic macroinvertebrate taxa, and a greater proportion of univoltine taxa representing longer-term conditions in the waterbody. Further, autumn provided a period of low-water conditions across Saskatchewan allowing for greater access. The reference condition approach requires a sufficient number of sites to characterize the variability amongst waterbodies when applied to multivariate models. Reynoldson and Wright (2000) suggest that approximately 250 sites are required to characterize variability adequately and build predictive models, as such, I assembled a total of 280 potential reference sites and sampled them over a four year study period from 2006-2009, together with 19 test sites known to have human impact on water chemistry, hydrology, or habitat (Table 5.1).

Table 5.1. Landcover variables evaluated in watersheds upstream of sampling sites.

Landcover type	**Description**
Annual Cropland	Land used for annual cropping
Native Pasture	Includes native and seeded grazing land but not riparian areas
Improved Pasture	Includes native and seeded grazing land but not riparian areas
Hay	Land used for cut forage (alfalfa, clover, grass, mix)
Forest	Treed land including cutovers and forest burns
Wetland	Saturated landscapes with wetland vegetation species
Water	Permanent water bodies
Barren	Non-vegetated areas including badlands, salt/mud flats, industrial facilities
Built up	Urban and populated areas

Biological data

Traveling kick-and-sweep samples were conducted at each site along a transect. These consisted of composite sweeps (10 seconds per sweep over an area ~ 1 m) on the left bank, ¼ distance, ½ distance, ¾ distance, and right bank. All five position sweeps were integrated into a single sample per transect. Samples were concentrated using a 500-μm-mesh sieve and immediately preserved with 80% ethanol. Organisms were sorted using a stereoscope at 7× magnification. The initial sample was subsampled when the total number of organisms was estimated to exceed 1000 individuals, by evenly spreading the samples on a 250-μm-mesh sieve and then removing half the sample to sort. Resulting abundances were then multiplied by the fraction removed to estimate original sample abundance. Use of a Marchant-box style subsampler was not possible due to high amounts of macrophytes and filamentous algae in many samples. Specimens were identified to the lowest possible taxon designation (usually genus and species, but family for Chironomidae, Ceratopogonidae, Stratiomyidae, Tabanidae, and Tipulidae, subcohort for Hydrachnidia, class for Oligochaeta, and phylum for Nematoda) using keys for North America (Clifford 1991, Webb 2002, Webb et al. 2004, Merritt et al. 2008). Voucher series were deposited in both the Water Security Agency Invertebrate Voucher Collection (Saskatoon, Saskatchewan) and the Royal Saskatchewan Museum (Regina, Saskatchewan).

Environmental data

The evaluation of land use and physical characteristics of watershed upstream of each site were evaluated using the "Tabulate Area" tool in ArcMap version 10.1. The digital elevation model (DEM) applied to generate watershed polygons was derived from data at the Canadian Digital Elevation Data (CDED; Natural Resources Canada http://geogratis.gc.ca/api/en/nrcan-rncan/esssst/3A537B2D-7058-FCED-8D0B-76452EC9D01F.html). The process for evaluating watershed variables was as follows: 310 Point files were created in Excel for each biomonitoring site recoded in Universal Transverse Mercator, Zone 13 standardized. Flow networks for each site were created using the ARCGIS Network Analyst extension. We then used the ArcGIS ArcHydro Extension, along with the CDED DEM dataset to create watershed polygons for each of the 917 point locations. Due to the relative lack of accuracy of the DEM product and hydrology GIS dataset, the watershed polygons for a considerable number of the sites were obviously incorrect. A visual review of a sample of those polygons showed that the polygons generated look "reasonable" for approximately 70% of the sites. The remaining approximately 30% of polygons were visually obviously not correct. These remaining 30% of sites were manually corrected in ArcGIS to capture the accurate watershed polygons.

The polygons of watershed area were further reduced by overlaying the effective drainage area which may be expected to entirely contribute runoff to the main river channel during a flood with a return period of two years and considering "dead drainage areas" proposed by Godwin and Martin (1975). A complete discussion of the drainage boundary delineation methods can be found in Hydrology Report #104 (PFRA Hydrology Division 1983).

Calculations for landcover in each polygon were derived from the AFSC_56m_2006 dataset produced by the Agricultural Financial Services Corporation (AFSC 2006, See Table 5.1). Landcover variables were summarized as the % composition of the total landcover in the effective drainage area of a polygon.

Soil composition classification was evaluated using Agriculture and Agri-Food Canada's (2005) digital Saskatchewan Soil Resource Data, Saskatchewan Soil Information System Ver. 1 (SKSISv1) in the effective drainage polygons for each scale. Ecoregion membership and composition of each polygon was determined using the Ecoregions ArcGIS file available from Agriculture and Agri-Food Canada (http://sis.agr.gc.ca/cansis/nsdb/ecostrat/gis_data.html). Surficial geology of each polygon was quantified using 1:250,000 scale GIS map (250K_surficial) from Agriculture and Agri-Food Canada Quaternary geology map series showing surficial terrain deposits classified by depositional environment and geomorphology (2008). Soils, Ecoregion, and surficial geology variables were summarized as % composition of the total effective drainage area in each polygon (Table 5.2).

Table 5.2. Physical abiotic variables of reference sites, as well as whether or not they were retained in Discriminant Functions Analysis (DFA).

Variable	Units	Scale	Variable Type	Retained?
Landscape				
Surficial Geology	% Composition transformed to PCA Axis 1	WS	Continuous	Y
Soils composition	% Composition transformed to PCA Axis 1	WS	Continuous	N
Ecoregion	none	WS	Categorical	Y
Hydrology				
Effective Watershed Area	m^2	WS	Continuous	N
Stream Order	none	Site	Categorical	Y
Median Annual Volume	dam^3	Site	Continuous	N
Median Peak Flow	$m^3 \cdot s^{-1}$	Site	Continuous	N
Median Minimum Flow	$m^3 \cdot s^{-1}$	Site	Continuous	N
Cross Sectional Habitat Availability	m^2	Site	Continuous	N
Water Chemistry				
Specific Conductivity	$\mu S \cdot cm^{-2}$	Site	Continuous	N
Turbidity	NTU	Site	Continuous	Y
pH		Site	Continuous	N
Dissolved Oxygen	$mg \cdot L^{-1}$	Site		N
Habitat				
Site Habitat Condition	none	Site		Y

Hydrology variables for each site were determined using Water Survey of Canada Hydrometric Stations upstream of each site (https://wateroffice.ec.gc.ca/). A 1:2 year median value was calculated for each of the following variables: Minimum Mean Daily Discharge ($m^3 \cdot s^{-1}$), Peak Mean Daily Discharge($m^3 \cdot s^{-1}$), and Annual Mean Volume (dam^3). Finally, Effective Drainage Area (EDA, km^2) was estimated using the two year return period contributing area described above (PFRA Hydrology Division 1983), and Stream Order was determined based on 1:50,000 National Topographic Survey of Canada series maps (http://www.nrcan.gc.ca/earth-sciences/geography/topographic-information/maps/9767; Table 5.2).

Turbidity (NTUs), Dissolved Oxygen (mg • L^{-1}), pH, and Specific Conductivity (μS • cm^{-1}) were all measured at time of benthic macroinvertebrate collection for each site. However, Dissolved Oxygen was removed from further analysis as it is susceptible to variation in time of day or season while Specific Conductivity was removed due to concern it could be influenced by human activity. Further, pH was removed for further analyses as it was found to have too small a range to meaningfully explain variation in benthic macroinvertebrate communities (mean ± 1 SD, 8.33 ±0.58; Table 5.2).

Site habitat condition is a composite of Environmental Protection Agency (EPA) Rapid Bioassessment variables for streams and wadeable rivers (Barbour et al. 1999). Specifically, I included Embeddedness, Channel Flow Status, Sediment Deposition, Flow Alteration, and Bank Stability. Each of these variables is scored from 0-20 from low condition to optimal condition (see Barbour et al. 1999). All variables were then summed to a maximum score of 100 at each site that was collectively referred to here at Site Habitat Condition. The Cross Sectional Habitat Availability (CSHA) at each site was calculated as a variation on hydraulic radius, in an effort to estimate how much space was available to benthic macroinvertebrates at each site (Fig. 5.2).

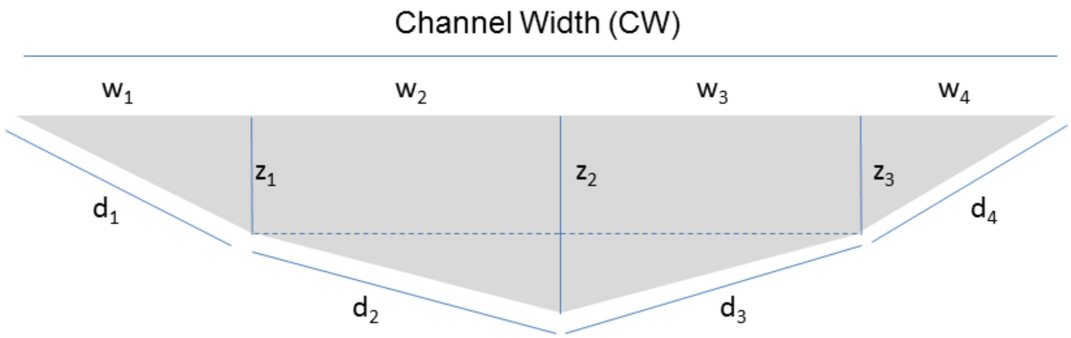

Figure 5.2. Cross Sectional Habitat Availability estimation.

To estimate this distance, I first measured the Channel Width (CW, m), then the depth of the channel at 1/3 Channel Width (z_1, m), ½ Channel Width (z_2, m), and 2/3 Channel Width (z_3, m) at time of site benthic macroinvertebrate collection. I then applied these dimensions in equations (Equations 1-4 below) to estimate benthic-distances (hypotenuses) for four sections of benthic habitat (see Fig. 5.2): as follows:

(Equation 1); $$d_1 = \sqrt{w_1^2 + z_1^2}$$

Where d_1 = benthic distance from left bank to 1/3 CW (m), w_1 = 1/3 CW (m), and z_1 = depth of channel at w_1 (m).

(Equation 2); $$d_2 = \sqrt{w_2^2 + (z_1^2 - z_2^2})$$

Where d_2 = benthic distance from 1/3 CW to ½ CW (m), w_2 = ½ CW - 1/3 CW (m), z_1 = depth of channel at 1/3 CW (m), and z_2 = depth of channel at ½ CW.

(Equation 3); $$d_3 = \sqrt{w_3{}^2 + (z_2{}^2 - z_3{}^2})$$

Where d_3 = benthic distance from ½ CW to 2/3 CW (m), w_3 = ½ CW - 1/3 CW (m), z_2 = depth of channel at ½ CW (m), and z_3 = depth of channel at 2/3 CW (m).

(Equation 4); $$d_4 = \sqrt{w_4{}^2 + z_3{}^2}$$

Where d_4 = benthic distance from 2/3 CW to right bank of channel (m), w_4 = 1/3 CW (m), and z_3 = depth of channel at 2/3 CW (m).

(Equation 5); $$\mathrm{CSHA} = d_1 + d_2 + d_3 + d_4$$

Where d_n =benthic distance (m) at each cross sectional division (Fig. 5.2).

Classification of reference site biological data

The first step in an RCA study design is to identify reference site biological community groupings at minimally disturbed sites. To begin with, taxa occurring in fewer than five sites, or those that had a total abundance of less than 10 individuals were removed from analyses of community composition as these community summaries (e.g., ordination and cluster analysis) are sensitive to rare taxa. Community data were $\log_{(n+1)}$ transformed prior to analyses, and three sites were removed from further analyses as they consisted of only Chironomidae and these sites would have led to poor characterization of community type.

To achieve these biological groupings, I clustered reference sites using the quantitative symmetric dissimilarity metric known as the Kulczynski distance linking metric (McCune and

Grace 2002) in order to relativize data points by sample unit totals since there was a large range in sample abundance totals (range = 18,464). This distance linking metric is a variation on the Bray-Curtis coefficient, but with a "built-in standardization" (Faith et al. 1987). Clustering of the reference sites was completed using an agglomerative hierarchical fusion method with complete linkage (Clarke and Warwick 2001).

Although Reynoldson and Wright (2000) recommended that only five sites may be necessary per biological grouping, I chose to increase the accuracy and precision of this RCA following the guidelines of Bowman and Somers (2005) to set a minimum cutoff of 20 sites for a biological grouping (Fig. 5.3).

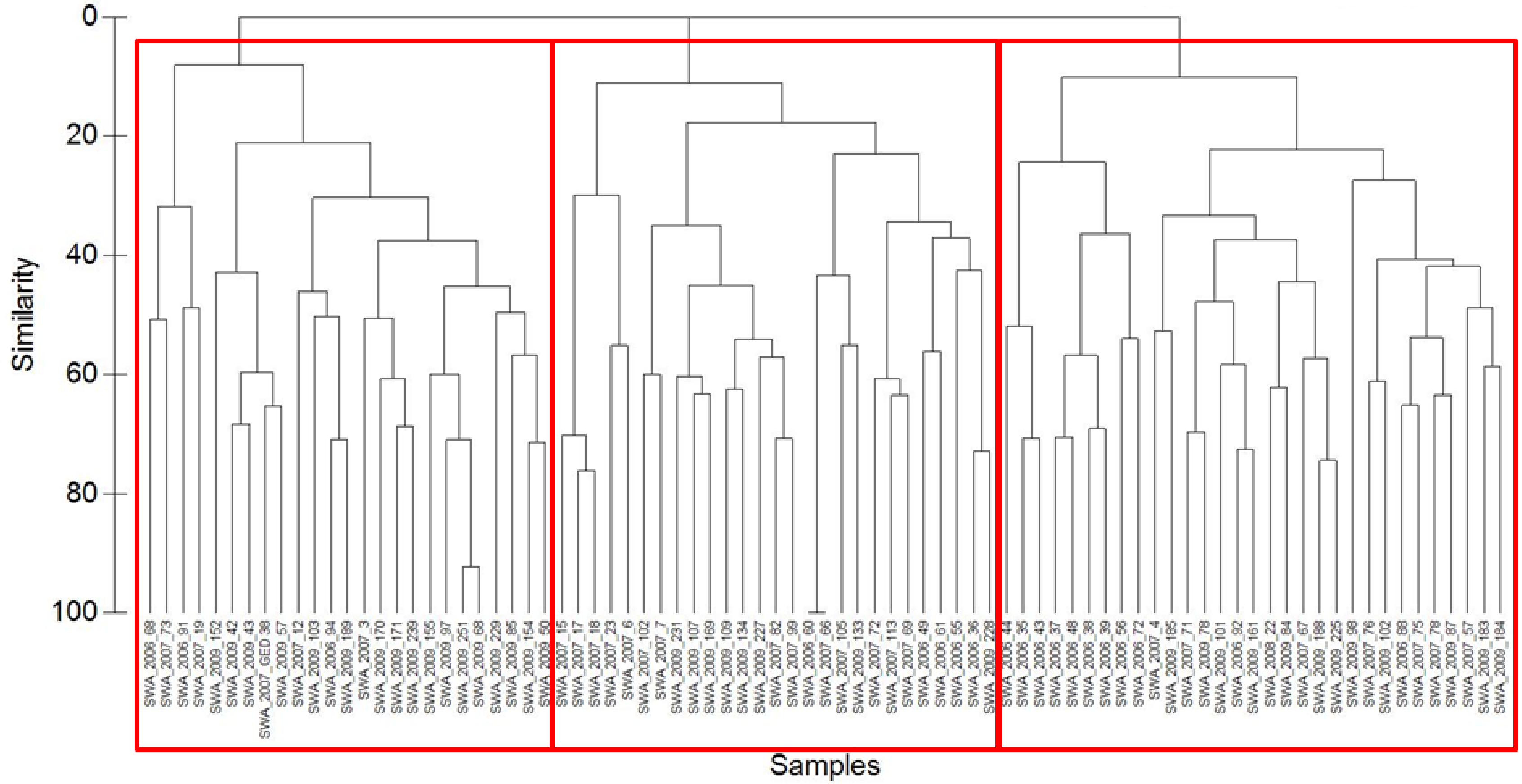

Figure 5.3. Cluster analysis of 83 reference sites in southern Saskatchewan. Data is $\log_{(n+1)}$ transformed, and the cluster analysis is completed using Kulczynski distance linking metric with complete linkage. Red boxes indicate the similarity cutoff used to separate the three biological groupings.

Reference site biological grouping abiotic discrimination

Secondarily, it is necessary to identify abiotic characteristics that can be used to distinguish reference biological communities and then subsequently use those abiotic characteristics to assign test sites to the most appropriate reference group. Of the 38 environmental abiotic variables collected in this study, I chose 14 of these variables to search for underlying relationships separating the reference site biological groupings. I only included variables most likely not to be influenced by human activity (Table 5.2). Basic correlations were used to determine redundancy of abiotic variables prior to further community-abiotic variable analysis to avoid false inflation of predictive capacity and multicollinearity of predictor variables. I used a cutoff of $r > 0.6$ to establish whether variables are autocorrelated, and then retained the variable with most biological relevance, established through best professional judgement.

Discriminant Functions Analysis was used to evaluate which of the remaining environmental abiotic variables best explain the separation between cluster analysis-based biological groupings. Due to unequal covariances in environmental variables, quadratic discriminant functions were used (Clarke et al. 2010).

Metric selection

I compiled commonly used benthic macroinvertebrate summary metrics which incorporate ecological knowledge about invertebrate assemblage responses to human stress (Resh and Jackson 1993). I initially examined 33 metrics using lowest-taxonomic designation level data (typically genus), but reduced the number of metrics by examining the relationships between each metric and the overall community structure. I began with number of Diptera, number of

Diptera families, number of EPT, % EPT, number of EPT genera, % collector gatherers, % detritivores, % filterers, % herbivores, % omnivores, % predators, % scrapers, % shredders, number of taxa, total abundance, Simpson's diversity, Shannon's diversity, Jaccard's evenness, number of Coleoptera, % Coleoptera, number of amphipods, % *Gammarus lacustris*, % *Hyalella azteca*, and NMDS community Axis 1 (Fig. 5.4). Next, I selected metrics for further inclusion in the model by conducting a preliminary ordination and ranking of the metrics from their correlation with the overall ordination structure, and then correlating metrics with all other metrics to reduce the number of redundant metrics (Reynoldson et al. 2001). I retained a metric if the probability of its correlation with the overall community structure was low ($p > 0.001$) and if it was not correlated with any of the remaining metrics.

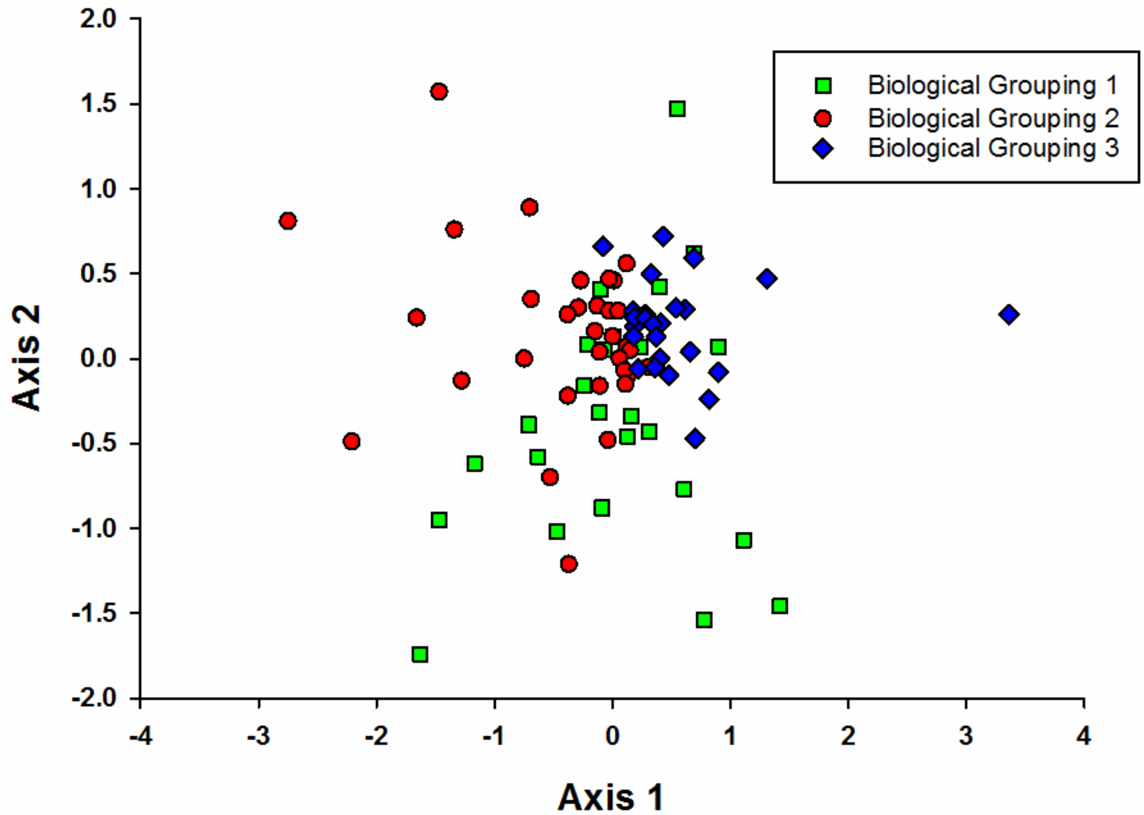

Figure 5.4. Non-metric multidimensional scaling ordination of reference sites based on community structure. $Log_{(n+1)}$ transformed, Manhattan Distance Linking Metric. Stress = 0.18

The metrics I used in the construction of a reference database are summarized in Table 5.3. The metric total abundance contributed most to the ordination and had the greatest amount of information and had the highest correlation with the ordination axis 1 (r = -0.539) (Table 5.3). Of the next highest ranked metrics, number of amphipods and number of EPT taxa were significantly correlated with total abundance and were therefore removed from further analyses. The next highest ranked metric to be included in subsequent analyses was number of Coleoptera, and I continued the correlation analyses on subsequent metrics until I produced a final list of four metrics (Table 5.3). This approach produced four non-redundant metrics and the : total abundance, number of Coleoptera, % EPT, and % shredders.

Table 5.3. Metrics and correlation of individual metrics with the first axis of the community ordination (ranked by contribution to the ordination structure).

Metric	**Correlation with ordination structure**	**Correlation of metric with total abundance**	**Probability level of correlation of metric with total abundance**
Total abundance	-0.539	1.000	0.000
number of Coleoptera	-0.445	0.274	0.012
% EPT	0.332	-0.162	0.143
% shredders	0.289	-0.171	0.123

Prediction of metric values at reference sites

The next step in evaluating condition in the reference condition approach model is to compare biological metrics at test sites to attributes of the reference site biological grouping with the closest underlying abiotic variables significantly discerning these biological groupings.

I then predicted membership of test sites to each of the biological groupings, using backwards DFA, and compared test site metric variables to those of their appropriate reference groupings using Test Site Analysis (Bowman and Somers 2006). Test Site Analysis formally evaluates the magnitude of difference between test sites and reference sites using a noncentral hypothesis test, defining the normal range as the probability region enclosing 95% of the reference sites (Kilgour et al. 1998). In this evaluation, a small probability ($p \leq 0.05$) suggests the test site is impaired, while a large probability (*p* > 0.05) suggests the site is not impaired (Kilgour et al. 1998). However, further to this I followed Bowman and Somers (2006) three-tiered condition applying the following conditions; impaired ($p \leq 0.05$), possibly impaired ($0.05 < p < 0.95$) and within reference condition.

Evaluation of metric performance

Ten test sites known to have human activities (waste water effluent [n=3], reservoirs [n=5], high urban development [n=1], and high crop agriculture [n=1]) were selected to evaluate the performance of the four metrics in discerning impact (Table 5.4). There is no *a priori* way to determine if these test sites are actually impaired, but were selected on known human development. However, applying these sites in comparison against reference site groupings will provide insight into whether or not the metrics selected can be used in evaluating disturbance. If none of the selected test sites exceed the variation in metrics at reference sites, then the model will be known to be ineffective in discerning impact.

Table 5.4. List of test sites with their respective stressors and biological grouping membership.

Test Site	Stressor	Site Code	River	Location	Biological Grouping
1	**Waste water**	SWA_2006_7	Wascana River	Regina	2
2		SWA_2009_33	Swift Current Creek	Swift Current	2
3		SWA_2006_31	Moose Jaw Creek	Moose Jaw	2
4	**Reservoir**	SWA_2007_57	Whitesand River	Theodore Dam	3
5		SWA_2006_26	Wood River	Thompson Dam	1
6		SWA_2006_90	Frenchman River	Eastend Reservoir	2
7		SWA_2008_14	Avonlea Creek	Avonlea	1
8		SWA_2006_15	Souris River	Rafferty Dam	2
9	**Urban**	SWA_2006_6	Wascana River	Regina Golf Course	2
10	**Crop**	SWA_2007_GED 63	Pipestone River Trib.	Moosomin	1

Backwards Discriminant Functions Analysis was used to assign test sites to their respective biological groupings of reference sites based on stream order and ecoregion described above. Test sites were then evaluated against respective biological groupings using Test Site Analysis.

Specifically, the Wascana River downstream of the Regina waste water treatment plant, Swift Current Creek downstream of the Swift Current waste water treatment plant, and the Moose Jaw Creek downstream of the City of Moose Jaw were compared to reference grouping 2 (Table 5.4). For potential impacts of reservoir operation, the Whitesand River downstream of Theodore Dam was compared to biological grouping 3, Wood River at the Thompson Dam and Avonlea Creek at the Avonlea water control structure were compared to biological grouping 1, while the Souris River downstream of the Rafferty Dam and Frenchman River at the Eastend Reservoir were compared to biological grouping 2 (Table 5.4). Finally, a single urban-dominated site at Regina in the Wascana River downstream of the city but upstream of the waste water treatment was evaluated against reference biological grouping 2, and a crop-dominated site on a Pipestone River tributary was evaluated against reference biological grouping 1 (Table 5.4).

Results

Reference Site Criteria

The final collection of 88 reference sites were characterized by <5% urban land use, <40% cropland land use, <50% pasture land use, < 70% total land under human influence in the upstream contributing watershed, as well as no greater than two landfills, oil wells, bridges or road crossings.

Classification of reference site biological data

Classification based on cluster analysis of the community data at reference sites resulted in three possible groups of 76 taxa found in the reference sites (Fig. 5.4). Groups 1, 2, and 3 consisted of 25, 33, and 25 sites respectively. Group 2 had the highest abundance, Group 3 the lowest abundance while Group 3 had the highest Shannon's Diversity, and Group 2 had the lowest Shannon's Diversity (Table 5.5). The most commonly occurring taxa in all groupings were Chironomidae, but the mayfly genus *Caenis* spp. occurred in high proportions in each grouping as well (Table 5.5).

Table 5.5. Taxa richness (mean number of taxa [±SD] per sample), total abundance (mean abundance [±SD] per sample), Shannon’s diversity (mean ±SD), Jaccard’s evenness (mean ±SD), and taxa considered as very common among the three groups of reference sites generated from cluster analysis of 76 taxa from southern Saskatchewan rivers and streams.

Group	**Taxa Richness**	**Total Abundance**	**Shannon's Diversity**	**Jaccard's Evenness**	**Common species (occurring in >60% sites in a group)**	**%**
1	16.6 (4.2)	1429.6 (1680.7)	1.6 (0.5)	0.6 (0.1)	Chironomidae	92
					Oligochaeta	80
					Hydrachnidia	80
					Hyalella azteca	76
					Ceratopogonidae	72
					Physidae	64
					Caenis spp.	60
2	14.6 (5.3)	2296.8 (3997.3)	1.4 (0.5)	0.5 (0.2)	Chironomidae	97
					Caenis spp.	91
					Hyalella azteca	88
					Enallagma/Coenagrion spp.	73
					Gammarus lacustris	67
3	15.9 (6.7)	326.6 (419.1)	1.9 (0.4)	0.7 (0.1)	Chironomidae	96
					Caenis spp.	84
					Heptageniidae	64

Underlying abiotic variable correlations

Initial correlations between abiotic variables were used to narrow down the number of variables retained for further analysis with the reference site community groupings (Table 5.2). First, I determined that the first PCA axis of surficial geology and the first PCA axis of soils were highly correlated (Pearson Correlation Coefficient $r = 0.989$, $p < 0.001$), thus I retained surficial geology in further analysis. For hydrology-associated abiotic variables, Median Annual Flow was highly correlated with both Median Peak Flow (Pearson Correlation Coefficient $r = 0.90$, $p < 0.001$) and Median Minimum Flow (Pearson Correlation Coefficient $r = 0.97$, $p < 0.001$), while Median Peak Flow and Median Minimum Flow were both correlated to each other (Pearson Correlation Coefficient $r = 0.80$, $p < 0.001$). In addition, Median Annual Volume was correlated with Effective Watershed Area (Spearman Rank Correlation Coefficient $r = 0.75$, $p < 0.001$), CSHA (Pearson Correlation Coefficient $r = 0.79$, $p < 0.001$), Stream Order (Spearman Rank Correlation Coefficient $r = 0.76$, $p < 0.001$). Stream Order was maintained for future analyses out of all the original hydrology-related abiotic variables as 1) they were all highly correlated with each other, 2) in future application Stream Order is the easiest data to collect from 1:50,000 maps, and 3) Stream Order has little chance of being affected by human activities and thus makes a good variable to match reference and test sites.

Of the remaining abiotic variables, Stream Order and Ecoregion were the only variables explaining a significant amount of separation between biological groupings in Fig. 5.4 (ANOVA, $F_{2,80} = 56.942$, $p < 0.001$). Stream Order and Ecoregion were then selected to apply in discriminant model predicting reference sites to biological groupings, and predicted 68.7% of the sites correctly using cross-validation.

Biological Grouping 1 was characterized primarily by Aspen Parkland, Cypress Hills Upland, and Mixed Grassland ecoregions, and all sites occurred in low first and second order streams (Table 5.6). In contrast, Biological Grouping 2 primarily occurred in Boreal Transition, Mixed Grassland, and Aspen Parkland ecoregions, with most of the sites in third and fourth order streams (Table 5.6). Finally, Biological Grouping 3 was nearly entirely characterized by Boreal Transition and Aspen Parkland ecoregions, as well as larger fourth and fifth order rivers (Table 5.6).

Table 5.6 Ecoregion and stream order site membership for Biological Groupings.

	Biological Groupings		
Physical feature	**1**	**2**	**3**
Ecoregion			
Cypress Hills Uplands	5	2	0
Mixed Grassland	8	11	2
Moist Mixed Grassland	1	2	1
Aspen Parkland	10	7	7
Boreal Transition	1	11	14
Mid Boreal Upland	0	0	1
Stream order			
1	18	0	1
2	7	4	1
3	0	14	4
4	0	12	11
5	0	3	8
6	0	1	0

Evaluation of Test Site condition

Of the ten test sites compared to their respective reference site biological groupings, one site was evaluated as impacted, two evaluated as stressed and the remaining seven test sites were evaluated to be healthy (Table 5.7). Essentially, this evaluation determined a 50% Type II error performance wherein a null hypothesis of “no human impact” would be retained when there is in fact human activity upstream of the site. The NMDS ordination axis 1 used in the analysis was not included in the final assessment as it was not sensitive to change.

Table 5.7. Probability of impairment at test sites known to be impacted from human activity compared against reference sites from their respective reference groupings.

	Anthropogenic Stressor									
	Waste water			**Reservoir**					**Urban**	**Crop**
Site:	1	2	3	4	5	6	7	8	9	10
Biological Grouping:	**2**	**2**	**2**	**3**	**1**	**2**	**1**	**2**	**2**	**1**
D in metric-based TSA	2.70	15.00	2.30	5.00	1.30	2.00	1.60	21.20	2.60	2.50
p-value for equivalence to reference	0.91	<0.001	1.00	0.02	1.00	1.00	1.00	<0.001	0.95	0.98
Total abundance	1.00	<0.001	1.00	<0.001	1.00	1.00	1.00	1.00	1.00	1.00
Number of Coleoptera	1.00	<0.001	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
% EPT	0.25	0.90	0.48	1.00	1.00	0.48	1.00	0.28	0.23	1.00
% Shredders	0.83	1.00	1.00	1.00	1.00	1.00	1.00	<0.001	0.97	1.00
Total <0.05	0	2	0	1	0	0	0	1	0	0

Discussion

The first null hypothesis that was tested in this study was that abiotic characteristics of rivers and streams would not explain significant variation in the benthic macroinvertebrate communities present at reference sites of least human activity. This hypothesis was rejected, as stream order and ecoregion predicted 68.7% of the sites correctly using cross-validation. This result is consistent with previous studies, where abiotic geographical predictors have been found to be useful in explaining waterbody to waterbody variation of benthic macroinvertebrate communities (Corkum 1989, Bailey et al. 1998). Applying easily measured abiotic geographical descriptors to predict components of variation in benthic macroinvertebrate communities can greatly improve biological assessments by refining the specific prediction of site-specific expected communities at a test site.

Similar to our findings, Reynoldson et al. (2001) found stream order and ecoregion variables predictive for reference site grouping classification in Fraser River Watershed sites. The Fraser River catchment in British Columbia has very different ecoregions and environmental conditions relative to that of southern Saskatchewan, suggesting these variables may have broader application as predictive of communities in the absence of human activity.

The development and application of benthic macroinvertebrate-based aquatic biomonitoring models has been provided across the globe, providing guidance on ecosystem health for most developed nations. Original work by Wright (1984) on the classification of lotic systems using benthic macroinvertebrates was adapted to the River Invertebrate Prediction and Classification System (RIVPACS) in the United Kingdom (Wright et al. 1991, Wright 1995). Multimetric and multivariate models have radiated and developed in Europe, Australia, and North America (E.g., Plafkin et al. 1989, Gerritsen 1995, Norris 1995, Reynoldson et al. 1995, Barbour et al. 1999, Barbour and Yoder 2000, Resh et al. 2000). In Canada specifically, the BEnthic Assessment of SedimenT (BEAST) approach formed the foundation of multivariate bioassessment models that have since expanded and fall under an umbrella of the Canadian Biomonitoring Network (CABIN). This network is the national biomonitoring program recommended by Environment Canada, and uses the reference condition approach as its common study design (Environment Canada 2012). My study was designed to provide a model for the assessment of aquatic ecosystem quality for the Northern Great Plains in Saskatchewan, filling a gap in model application across Canada in the CABIN program.

The current Test Site Analysis, reference condition approach model developed here provides a tool to evaluate ecosystem health of rivers and streams of the Northern Great Plains filling the gap in coverage provided by the CABIN program. Further, the use of simple abiotic variables to link test sites and reference groupings (stream order and ecoregion), combined with easy to calculate metrics of the benthic macroinvertebrate community allow for a simplified and easily usable condition assessment.

Hynes (1960) clarified that water has lost its natural qualities only when pollution is severe enough to cause changes that exceed the boundaries of natural variation. Applying a large number of metrics to evaluate human impact can increase the sensitivity of this natural variation in biomonitoring models, but including metrics that do not discriminate between impaired and reference conditions will decrease sensitivity. Future experiments to refine metrics forensic of human impact will assist the biomonitoring model developed here and enable better quantification of variation from natural condition.

Acknowledgments

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CHAPTER 6: ECOSYSTEM HEALTH MONITORING AND ASSESSMENT IN SASKATCHEWAN USING BENTHIC MACROINVERTEBRATES - CONCLUSIONS

Rivers and streams are a critical component of Northern Great Plains ecosystems (Dodds et al. 2004), and are an extremely valuable source of surface water in an otherwise primarily arid environment. Despite their importance, these waterbodies have traditionally received much less scientific attention than forested regions of North America (Matthews 1988). Of further importance, it has been estimated that approximately 95% of the native prairie has been lost through expansion of agriculture (Samson and Knopf 1994), and rivers dammed for water retention and hydropower production resulting in altered water chemistry and hydrology. This thesis provides insight into what best available natural communities may still exist, and how alterations in the physical characteristics of rivers and streams can push communities outside of their range of natural variation.

Specifically, I show that since the construction of a hydroelectric dam on a large NGP river densities of benthic macroinvertebrates have increased significantly through time relative to reference sites and pre-dam habitats. Further, these communities have changed through time to be different downstream of the reservoir in both soft and hard sediment. My findings indicate that although there is a notable loss in sensitive taxa such as mayflies and stoneflies, other midge taxa colonize this unique, cold water habitat and create a new community of benthic macroinvertebrates. Further, reaches downstream of the dam are significantly cooler than reference through the summer into August and do not reach the temperature optima of reference reaches. This cold-water release, or some other change in flow characteristics, changes abundance, diversity, % Orthocladiinae, and community composition. I used these metrics to compare test sites to reference sites and to quantify the impact of the Lake Diefenbaker reservoir on community metrics characteristic of temperature stress and develop a reference condition model that assesses current condition and can be used to monitor recovery through mitigation of these perturbations. My results have implications primarily for understanding and quantifying the ecosystem impacts of hydroelectric energy production, but also range expansion of cold-water tolerant taxa, the life-history of select groups of invertebrates, and ultimately the forage resources available to the fish assemblages of this river system. Future exploration of how these changes in community composition relate to secondary production and biomass would be valuable in assessing the impact of the reservoir on the ecosystem's ability to support fish populations. These results and model are already being adapted to monitor current pollution threats to aquatic ecosystems in the Northern Great Plains such as the 16Tan Husky Oil spill in the North Saskatchewan River in which 250,000 L of bitumen were accidentally released in July 2016 (Phillips 2017).

Further, as Northern Great Plains rivers are typified by considerable flow variability, particularly in the presence of water control structures, fine sediment (<63μm) is readily suspended, especially during periods of high discharge. Assessment of the impacts to biota by anthropogenic stressors must therefore occur within the context of dynamic turbidity and background flow conditions. I developed a model in which discharge is a principal determinant of in-stream suspended sediment. This relationship was explored with a case study showing that macroinvertebrate community structure is strongly correlated with suspended sediment gradients and ultimately predicted by discharge. Factors affecting sediment loads and ecosystem responses in managed systems should be considered so that in-stream water quantity and quality needs are met. This new understanding should allow for the development of improved ecosystem based flow management objectives.

Finally, I develop a multivariate and predictive model based on the reference condition approach for the Northern Great Plains region of Saskatchewan. Developed from benthic macroinvertebrate communities and environmental abiotic data were collected at 280 reference sites and 10 test sites reference sites were classified into groups characterized by similar macroinvertebrate communities. This model predicted 68.7% of the sites correctly using cross-validation. Of the 10 test sites, two were stressed (one waste water and one urban site) while three were classified as impaired (one waste water and two reservoirs). This model is an effective tool that provides a practical means of evaluating biotic condition of streams in the Northern Great Plains.

Philosophy implications of what reference means in the Northern Great Plains

Biomonitoring programs will always struggle to achieve the philosophical biological optima underpinning their condition objectives, no different in concept from the difficulties in setting reference criteria for the Northern Great Plains. The challenge is to characterize reference site criteria that retain sites with high biological integrity and achieve the conceptual ideal of theoretical reference condition that water management should strive to achieve. Truly pristine waterbodies untouched by human activities no longer exist in the strictest sense (Boivin et al. 2016). As such, any language inferring that we could compare current water quality and ecological health to that which would occur in the absence of human activity is pointless, and requires a more realistic view of what comparisons a biomonitoring program should be seeking to achieve (Ode et al. 2016). Although this change in perspective is often characterized as a "shifting baseline," and a lamentable disservice to the environment (Papworth et al. 2009), it is realistically all that may be possible today (Stoddard et al. 2006). There is even scientific argument for this human alteration of the environment being so great that the earth has passed into a new epoch because of it (Waters et al. 2016), and the environmental consequences captured in this "Anthropocene" are broad-reaching to global declines in biodiversity.

However, establishing site-specific objectives for community composition and diversity based on reference condition has advantages even if this condition is not pristine. Specifically, mitigation of human impact can even improve metrics of ecosystem health such as diversity to be greater than what is observed in comparable waterbodies. They may be significantly different than reference, but if they have metrics higher in quality than reference sites (e.g., higher in diversity) this may enable hypothesis generation for what water quality, habitat, or abiotic conditions could be promoted in other test sites to increase ecosystem health. This is particularly realistic to expect when working with reference sites that already have quite tolerant assemblages of benthic macroinvertebrates.

Consistent with my initial observations that benthic macroinvertebrate taxa in the Northern Great Plains are more tolerant than other regions where ecosystem health models have been developed (Fig. 1.1), reference sites in the Northern Great Plains were typified by taxa such as Chironomidae, Oligochaetes, and *Caenis* sp. mayflies (Table 5.5)—all of which are more tolerant to anthropogenic stress (Barbour et al. 1999). However, I was able to successfully develop a community classification of reference site groupings that can be used to evaluate human stress at test sites and set site-specific ecosystem health objectives. Though the communities of benthic macroinvertebrates in the Saskatchewan River system, the Qu'Appelle River system, and the wadeable streams of the Northern Great Plains are not without human influence, I have successfully constructed models for site specific objectives that can be used to begin management and allow hypothesis testing for mitigation of anthropogenic perturbation.

Though this model works in its present form, many questions and hypotheses remain. For instance, Chironomidae dominated the benthic macroinvertebrate community, and are a very diverse group with ~ 190 species known from the region from the South Saskatchewan River to Montana (Mason and Parker 1994, Parker and Glozier 2005, Phillips et al. 2013). A valuable future hypothesis for the expansion and refinement of the model developed here would be to identify chironmids to genera or species to help refine community classification and detect impacts. As a family, Chironomidae are represented in all functional feeding groups (Merritt et al. 2008) and cover the range of tolerances provided in Barbour et al. (1999).

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