

EFFECTS OF A HYDROPEAKING DAM ON RIVER HEALTH AND BENTHIC
MACROINVERTEBRATE SECONDARY PRODUCTION IN A NORTHERN GREAT
PLAINS RIVER

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By

Jordan Edward Mihalicz

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Head, School of Environment and Sustainability
University of Saskatchewan
Saskatoon, Saskatchewan S7N 5A8
Canada

OR

Dean
College of Graduate and Postdoctoral Studies
University of Saskatchewan
116 Thorvaldson Building, 110 Science Place
Saskatoon, Saskatchewan S7N 5C9
Canada

ABSTRACT

Hydroelectric dams represent an ever-growing portion of the global energy grid, and the number of operations practicing hydropeaking is on the rise. Early research shows that dams affect benthic macroinvertebrates (BMIs). In contrast, our knowledge of the impacts of hydropeaking on BMIs is limited; this contributes to gaps in our understanding of how hydropeaking affects structural and functional aspects of lotic ecosystem health. This is especially apparent in the large rivers of the Northern Great Plains where aquatic health and function are rarely measured. River health is often scored using metrics such as biotic indices (BI) and the ratio of Ephemeroptera, Plecoptera, and Trichoptera to Chironomidae (EPT/C), whereas functional measures include calculating secondary production are far less common. This dissertation sought to explore the effects of the hydropeaking E. B. Campbell dam on downstream BMI assemblages in the Saskatchewan River using BI, EPT/C, and estimating BMI mean daily secondary production while considering seasonality as a key factor.

BMI communities downstream of the dam had higher proportions of tolerant taxa relative to upstream sites regardless of season. This was reflected in elevated BI and lower EPT/C scores observed downstream. Canonical correspondence analysis (CCA) showed clear differences between upstream and downstream sites, and an analysis of similarities (ANOSIM) found the downstream BMI assemblage was significantly different than the one found upstream. A similarity percentages (SIMPER) analysis revealed that *Sigara lineata* (Hemiptera: Corixidae), *Hydropsyche sp.* (Trichoptera: Hydropsychidae), and Chironomidae (Diptera) were among the taxa that contributed most to differences between upstream and downstream locations.

Estimated mean daily secondary production and mean instantaneous growth rates for Chironomidae, *Sigara lineata*, *Hydropsyche sp.*, and *Stenonema sp.* (Ephemeroptera: Heptageniidae) ranged between 0.083 – 0.996 mg m⁻² day⁻¹ and 0.004 – 0.021 upstream, and 0.059 – 23.402 mg m⁻² day⁻¹ and 0.021 – 0.046 downstream, respectively. High production in downstream *Hydropsyche sp.* may be attributed to their proximity to the reservoir.

This dissertation demonstrated that hydropeaking may alter downstream BMI community composition and production, resulting in degraded river health. It was shown that seasonality can play a key role in dictating hydropeaking effects. Continued monitoring of downstream BMI assemblages will provide valuable information that can mitigate the impacts of hydropeaking operations on the downstream environment.

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CHAPTER 1: GENERAL INTRODUCTION

1.1 Hydroelectricity and Its Impacts

1.1.1 Hydroelectricity and known impacts on rivers

With increasing pressure to move from fossil fuels to renewable energy, hydroelectricity has proven to be a reliable source of power across the globe. As a result, most large rivers across the world have had at least one dam built along their length (Smokorowski et al. 2011). Despite generating relatively clean energy compared to fossil fuels, hydroelectric dams can create impacts on the downstream environment (Rosenberg et al. 1997). Fragmentation of rivers can affect the movement of fish and cause shifts in benthic macroinvertebrate (BMI) community structure (Jones 2013a).

The effects of dams on downstream biota have been documented for decades. Changes to the downstream environment are related to the type of operation in place, and impacts include altered thermal (Lehmkuhl 1972, Olden and Naiman 2010, Phillips et al. 2015) and flow regimes (Poff et al. 2007), water quality (Phillips et al. 2016), and benthic assemblages (Poff and Zimmerman 2010, Jones 2013b). Hydropeaking has received little attention compared to other types of regulation. This data gap is beginning to be filled, with recent studies highlighting BMI egg mortality due to dessication (Kennedy et al. 2016), hydropeaking-induced BMI drift (Schülting et al. 2016), hydraulic stress tolerance of BMIs (Leitner et al. 2016), as well as mitigation strategies to reduce impacts caused by hydropeaking (Bruder et al. 2016, Premstaller et al. 2017, Tonolla et al. 2017, Hauer et al. 2017). Hydropeaking is defined as the cycle of increasing and decreasing flow rate that closely follows electricity demand, often referred to as ramping (Armanini et al. 2014). In

particular, hydropeaking operations in the Northern Great Plains of Canada remain poorly studied.

1.1.2 Hydroelectricity in Saskatchewan

The Saskatchewan River (Saskatchewan, Canada) consists of two main stems, the North Saskatchewan River and South Saskatchewan River, which converge to form the mainstem Saskatchewan River. It is one of North America's largest, with mean annual discharge for the South Saskatchewan River at Saskatoon of $268 \text{ m}^3 \text{ s}^{-1}$ and $289 \text{ m}^3 \text{ s}^{-1}$ in the North Saskatchewan River at Prince Albert (https://wateroffice.ec.gc.ca/search/historical_e.html). In fact, Saskatchewan is derived from the Plains Cree word *kisiskâciwan*, meaning "swift flowing water." As such, parts of the river have been exploited for their electricity-generating potential, and several hydroelectric operations have been built along its length including the Gardiner Dam on the South Saskatchewan River and the hydropeaking E.B. Campbell Dam on the mainstem Saskatchewan River.

1.2 Benthic Macroinvertebrates (BMIs) and River Health

1.2.1 River health

The concrete definition of river health has been and continues to be debated by many scholars (Karr 1991, Meyer 1997, Karr 1999, Bunn et al. 1999) with some considering it as being analogous to human health (Norris and Thoms 1999). However, unlike human health, the health of a river probably cannot be determined by a single 'check-up' and requires multiple samples over a period of time as well as comparisons to parts of the river that are deemed 'healthy.' A variety of methods exist for determining river health, including assessing the biotic integrity of the system from a structural and/or

functional perspective (Karr 1999). River health assessments are largely based on indicators, which can include the presence or absence of certain organisms, river chemistry, and species community structure (Karr 1991, Karr 1999). Frequently used indicator taxa include fish and BMIs, the latter being more common due to their ubiquity, relative ease of sampling, and sensitivity to environmental impairment (Lamberti and Berg 1995, Huryn and Wallace 2000, Mandaville 2002, Benke and Huryn 2006). The literature suggests that the majority of river health assessments have been performed using structural (e.g. water quality, taxonomic composition) rather than functional indicators (e.g. decomposition rates, primary and secondary productivity) (Gessner and Chauvet 2002, Young and Collier 2009). However, an increasing number of researchers are recognizing the value of using functional indicators, suggesting that they may be able to detect low levels of environmental damage that structural indicators are unable to distinguish (Buffagni and Comin 2000, Gessner and Chauvet 2002, Young et al. 2008, Young and Collier 2009). Therefore, approaches that use both structural and functional indicators are likely to provide a more accurate assessment of river health (Yates et al. 2014). Multivariate approaches have proven to be more effective than multimetric methods at assessing ecological integrity (Reynoldson et al. 1997), and the number of river health studies applying multivariate analyses have been on the rise, especially in Canada (Reynoldson et al. 2001, Horrigan and Baird 2008, Yates and Bailey 2010, Medeiros et al. 2011) and Australia (Chessman et al. 2010, Cortez et al. 2012). Additionally, there is a paucity of literature that has examined whether relationships exist between river health and functional indicators such as BMI productivity.

1.2.2 BMI secondary production

Several methods have been proposed for determining the secondary production of BMIs over the years (e.g. Benke 1993), most of which incorporate relationships between the length and mass of the collected specimens (Benke et al. 1999, Benke and Huryn 2006). Among these techniques are cohort and noncohort, the latter of which has several specific methods for calculating production including the size-frequency method and the instantaneous growth method (Benke and Huryn 2006). Calculating the instantaneous growth rate of BMIs is particularly useful as it can be used to track changes in production over time. This method utilizes equations that ultimately express annual production as a single value in $\text{g m}^{-2} \text{y}^{-1}$. These values are commonly used to estimate the growth rate of a macroinvertebrate population by dividing the production values by the biomass of the population, commonly expressed as P/B (Huryn and Wallace 2000, Benke and Huryn 2006). While the production of single species populations has been determined for a wide variety of macroinvertebrate taxa, production estimates for entire freshwater benthic communities are encountered less frequently in the literature (Morin and Dumont 1994). However, an increasing number of studies have estimated macroinvertebrate community production in recent years (Buffagni and Comin 2000, Stagliano and Whiles 2002, Entekin et al. 2007). Annual community production values for BMIs differ widely, ranging from 10 to almost 1000 grams of dry mass $\text{m}^{-2} \text{year}^{-1}$; most values are between 10 and 50 g dry mass $\text{m}^{-2} \text{year}^{-1}$ (Benke 1993, Huryn and Wallace 2000, Benke and Huryn 2006). BMI community P/B ratios have a similar range from <1 to >100, with most being <6 (Benke and Huryn 2006). The majority of studies on BMI secondary production have focused on small, wadeable streams (Buffagni and Comin 2000, Stagliano and Whiles

2002, Entekin et al. 2007), whereas few have considered large rivers due to a variety of factors, especially the complexity of sampling.

1.2.3 History of using BMIs as indicators

Owing to their ubiquity and ease of capture, BMIs have long been used for monitoring the integrity of aquatic ecosystems (Cairns and Pratt 1993). Modern biomonitoring in North America using BMIs originated with the large-scale river surveys conducted by Ruth Patrick and her colleagues starting in 1948 (Cairns and Pratt 1993). Since then, a variety of metrics have been developed to assess the health of aquatic systems using benthic communities including the ratio of disturbance-sensitive Ephemeroptera, Plecoptera, and Trichoptera (EPT) to tolerant Chironomidae (EPT/C), the percentage of EPT in the community (%EPT), and the Hilsenhoff Biotic Index (BI) (Mandaville 2002).

1.2.4 Impacts of hydroelectric dams on BMIs

How impoundments affect BMIs in downstream environments has been the subject of many studies (Lehmkuhl 1972, Jones 2013a, Phillips et al. 2015, Phillips et al. 2016). Lehmkuhl (1972) as well as Phillips et al. (2015) have documented how the hypolimnetic release of the Gardiner Dam changed the thermal regime in the South Saskatchewan River and caused the disappearance of many sensitive BMI taxa (Ephemeroptera, Plecoptera, Trichoptera, etc.). Others have examined how modified flow conditions from hydropeaking operations have altered BMI assemblages downstream (Armanini et al. 2014, Kennedy et al. 2016, Holzapfel et al. 2017). Typical effects include catastrophic BMI drift and increased egg mortality (Kennedy et al. 2016), both of which may contribute to a shift toward a disturbance-tolerant downstream BMI

community. However, the majority of hydropeaking studies in temperate regions do not incorporate seasonality and are usually conducted in late summer/autumn. This type of sampling design would miss any taxa that emerge in the early spring and summer, including some stoneflies (Plecoptera) and mayflies (Ephemeroptera).

Based on the literature, there is a clear gap concerning the impacts of hydropeaking power generation facilities on BMI productivity and biodiversity. Whether relationships exist between productivity, biodiversity, and river health is also unclear, as are the potential effects of altered flow regimes on these relationships.

1.3 Thesis Objectives

1.3.1 Effects of a hydropeaking dam on downstream benthic macroinvertebrates

One of the primary objectives of this study was to elucidate whether the daily and seasonal fluctuation in discharge from the hydropeaking E.B. Campbell Dam affects the downstream BMI community. This was examined by sampling several locations both upstream and downstream of the dam using a seasonal sampling design. Locations upstream of the dam were considered as reference sites, whereas downstream reaches represented potentially impacted sites. BMI assemblages were evaluated using several established metrics to determine river health. Any differences between upstream and downstream communities were determined using multivariate analyses. I hypothesized that the hydropeaking operation of the E.B. Campbell Dam may contribute to the alteration of the BMI community downstream and that river health was degraded downstream of the dam. With my findings regarding impacts of this hydropeaking dam, I reviewed key literature to help understand how hydropeaking dams differ in their impacts from other dam operations.

1.3.2 Daily secondary production of benthic macroinvertebrates in a Great Plains River influenced by a hydropeaking dam

Secondary production of BMIs is a common measure of riverine ecosystem function (Benke and Huryn 2006). In chapter 3, I assess whether hydropeaking affected the secondary production of downstream BMI communities relative to those found upstream. Specimens had their body length measured, which were then converted to an individual mass using established length-mass regressions (Benke 1999). Daily secondary production was calculated using the instantaneous growth method by first estimating the instantaneous growth rate of each population (Benke and Huryn 2006). Estimates for daily secondary production were compared to those found in the literature for related species. I hypothesized that hydropeaking favored production of only tolerant taxa downstream and that BMI daily secondary production was lower at locations downstream of E.B. Campbell Dam where hydropeaking was most severe.

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CHAPTER 2: EFFECTS OF A HYDROPEAKING DAM ON DOWNSTREAM BENTHIC MACROINVERTEBRATES

2.1 Abstract

As more hydroelectric dams regulate rivers to meet growing energy demands, there is rising concern about downstream effects, including impacts on downstream benthic macroinvertebrate (BMI) communities. Hydropeaking is a common hydroelectric practice where short-term variation in demand for electricity is met by changing flows. Hydropeaking leads to large and often rapid fluctuations in discharge and water level, and as such, impacts are expected to differ from impoundments with more stable flow regimes. There are key knowledge gaps on the ecosystem impacts of hydropeaking in large rivers, the seasonality of these impacts, and whether dams can be managed to help lessen impacts. The E.B. Campbell Dam on the Saskatchewan River (Saskatchewan, Canada) is a hydropeaking dam that causes daily fluctuations in water levels downstream, despite relatively new minimum flow requirements implemented in 2004. In this study, we assessed how patterns of hydropeaking affect abundance, taxonomic richness and relative tolerance of BMIs. We aimed to capture key seasonal changes during the relatively brief ice-free season, sampling from May to September 2014 at eight locations above and below the dam. Reaches immediately (<2km) downstream of the dam in a lentic-impact zone generally had high densities of BMIs and comparable taxonomic diversity relative to upstream locations, but were characterized by higher biotic index scores and lower ratios of sensitive (Ephemeroptera, Plecoptera, Trichoptera) to tolerant (Chironomidae) taxa. While biotic index and EPT/C scores remained high for the next 28 km downstream, these metrics returned to upstream values at the furthest downstream location (53 km). The magnitude of effect also varied with seasonal changes in discharge.

Downstream BMI community structure was most different from upstream locations during the period of May – July when the mean flow was higher than later in summer (August – September). Understanding the effects of river regulation on BMI biodiversity and river health has implications for mitigating the impacts of hydropeaking dams on downstream ecosystems. While we demonstrated that a hydropeaking dam may contribute to a significantly different downstream BMI assemblage, we emphasize that seasonality is a key consideration. The greatest differences between upstream and downstream locations occurred in spring, suggesting standard methods of late summer and fall sampling may underestimate ecosystem-scale impacts.

2.2 Introduction

At present, a large majority of the world's river systems have at least one dam somewhere along their length (Smokorowski et al. 2011), with more planned for the future (Zarfl et al. 2015). While it is a clean, renewable energy source compared to oil, gas, and coal burning, hydroelectric power also creates environmental impacts (Rosenberg et al. 1997). The effects of dams on rivers have been well documented over the last several decades, from changes in river thermal (Olden and Naiman 2010, Phillips et al. 2015) and flow regimes (Poff et al. 2007) to altered biological assemblages (Poff and Zimmerman 2010) and water quality (Phillips et al. 2016). While a number of recent studies have focused on flow alterations caused by hydropeaking operations and their effects on instream benthic macroinvertebrate communities (Jones 2013a, Jones 2013b, Armanini et al. 2014, Kennedy et al. 2016), the impacts of hydropeaking operations are poorly known, as compared to dam impacts more generally.

Benthic macroinvertebrates (BMIs) have been widely recognized as indicators of ecosystem integrity due to their wide tolerance spectrum to a variety of environmental disturbances (Cairns and Pratt 1993, Bonada et al. 2006, Bailey et al. 2014). Sensitive taxa such as most mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) tend to decrease in abundance and diversity in impacted rivers, while relatively tolerant taxa, including many chironomid species and oligochaetes, remain. Several metrics have been developed that utilize BMIs to quantify aquatic health, including the percentage of Ephemeroptera, Plecoptera, and Trichoptera (%EPT), the ratio of EPT to Chironomidae (EPT/C), and the Modified Hilsenhoff's Biotic Index (BI; Plafkin et al. 1989 in Mandaville 2002). These metrics are often used in assessing the health of wadeable streams and small rivers, but are rarely applied to large river systems (Jackson et al. 2010). The EPT/C metric has been widely used for evaluating the effects of general environmental disturbances (Karr 1991, Hannaford and Resh 1995, Odum et al. 2012). In a recent study, Alvial et al. (2012) showed that the Family Biotic Index, which uses BMI tolerance values at the family level, could be used to examine the effects of mining activities on BMIs in Chilean rivers. However, it is uncertain if the BI metric can indicate physical stress from hydropeaking, as its calculation uses taxa tolerance values that were developed for organic pollution.

Large rivers in the North American Great Plains naturally experience predictable fluctuations in discharge and depth throughout the year, often becoming swollen with snowmelt and mountain runoff in the late spring and returning to baseflow by late summer (Poff 1996). Dams regulate these fluctuations, often attenuating flood conditions by discharging less water over a longer period than the natural spring meltwater surge.

Hydropeaking refers to the rapid changes in discharge used by hydroelectric facilities to produce power during daily peak demand. Despite knowledge that patterns in BMI diversity and abundance are seasonally dependent (Linke et al. 1999), studies that have examined the impact of hydropeaking dams on BMI communities (e.g. Jones 2013a, Jones 2013b) are often done during the late summer, presumably to capture the highest diversity and later life stages of BMIs. This sample design does not capture the univoltine BMIs that develop and emerge in the spring and early summer, such as winter stoneflies and many mayfly species. How seasonal variations in flow overlaid by hydropeaking affect BMI life histories is poorly characterized.

Here we assessed the potential effects of a daily hydropeaking dam on downstream BMI communities by comparing five downstream locations with three upstream reference locations sampled monthly during the ice-free season in 2014. We hypothesized that the BMI assemblages immediately downstream of the dam are affected by the hydropeaking operations and that river health, as calculated using BMI metrics such as EPT/C and BI, is compromised at these locations through a combination of abrupt changes in flow, considerable fluctuations in water level, and repeated wetting and drying of river substrate (Kennedy et al. 2016). We also examined the potential for seasonal variation in the effects of hydropeaking by evaluating BMI assemblages across five months that varied considerably in flow conditions. Given the large number of extant dams, the common use of hydropeaking, and ongoing dam construction in many regions (Zarfl et al. 2015), understanding the effects of hydropeaking is a key step towards better understanding costs and benefits of alternative flow management regimes.

2.3 Methods

2.3.1 Study area

The Saskatchewan River basin in Canada is one of North America's largest river basins (405,864 km²), spanning three provinces and includes one of the largest freshwater deltas in the world (Partners for the Saskatchewan River Basin 2009). It is a sand-dominated river that begins in the Rocky Mountains of Alberta and discharges into Lake Winnipeg in Manitoba. The river is divided into the North Saskatchewan and South Saskatchewan rivers, which merge east of the city of Prince Albert to form the mainstem Saskatchewan River (Fig. 2.1). Ice cover on the river typically lasts from late November to April, although this can vary annually. Two large hydro dams were commissioned along the river system: the Gardiner Dam in 1967 on the South Saskatchewan River and the hydropeaking E.B. Campbell Dam in 1963 on the mainstem of the Saskatchewan River. Together, these dams alter the seasonal and daily flow regime downstream (Schindler and Donahue 2006, Gober and Wheeler 2014). The hydropeaking E.B. Campbell dam formed the Tobin Lake reservoir, and from 1963 to 2004, the dam operated in accordance with electricity demand, causing sudden changes in river depth downstream and occasionally stranding fish. This prompted Fisheries and Oceans Canada to establish a minimum flow requirement of 75 m³s⁻¹ for the E.B. Campbell operation as a way to mitigate changes in water level. However, the river downstream continues to experience daily changes in discharge and depth due to hydropeaking practices (Figs. 2.2 and 2.3); these changes attenuate downstream (Euteneier 2002) but are observable as far as 60 km from the dam.



Fig. 2.1: The portion of the Saskatchewan River system sampled in this study. Blue dots indicate upstream (reference) reaches, red dots are downstream (test) locations, and the split dot indicates the sampling location immediately downstream of E.B. Campbell Dam. The inset illustrates the sample area's location in Saskatchewan, Canada.

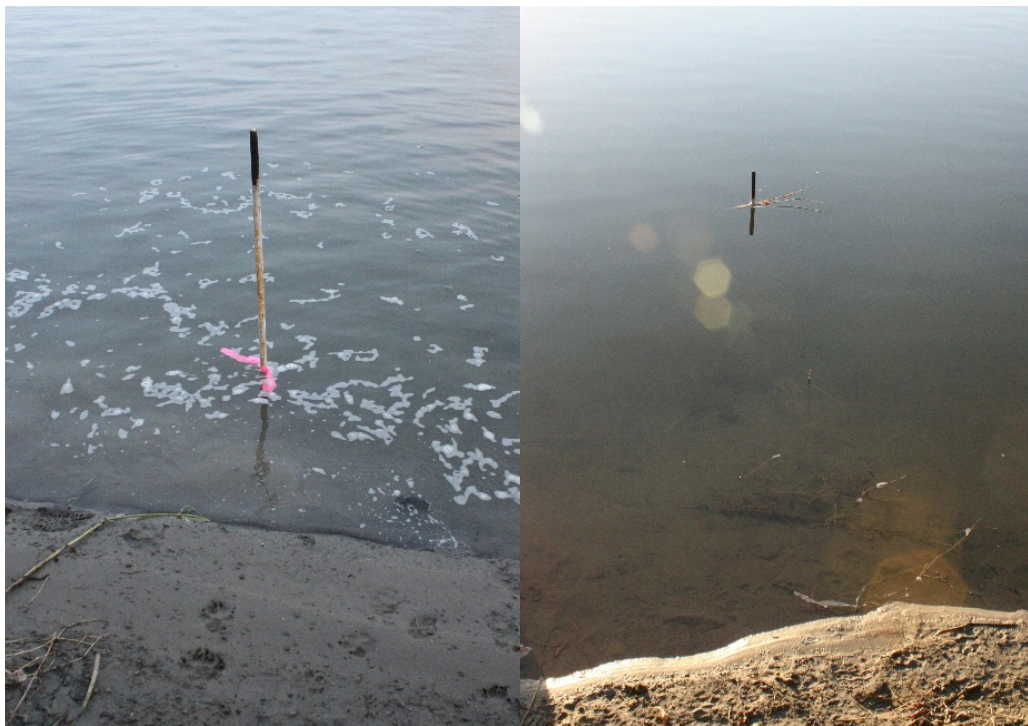


Fig. 2.2: Images taken at DS4, the sampling location furthest downstream (+53 km) from E.B. Campbell dam, on the evening of September 23 (top) and the morning of September 24, 2014 (bottom). The pole was used to illustrate the effect of daily hydropeaking on water levels at downstream locations and was not disturbed during the time between the two images.

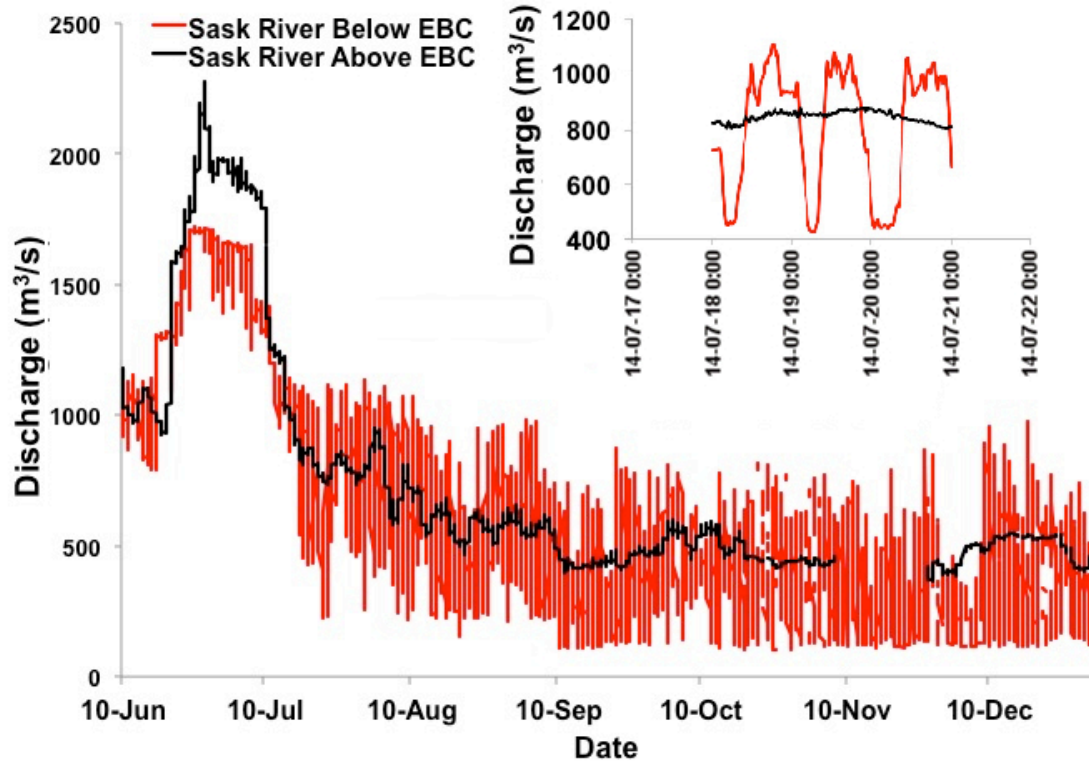


Fig. 2.3: Discharge data for the Saskatchewan River above and below E.B. Campbell Dam for June-December of 2014 (Environment Canada gauges 05KD007 and 05KD003). Note the peak in discharge that occurs from June to July due to the spring freshet. The erratic changes in discharge downstream of the dam are the result of hydropeaking. Data from 12:00 am on July 18 to 12:00 am on July 21, 2014 illustrates the daily peaks and troughs in discharge experienced by the river downstream due to hydropeaking (inset).

A total of eight locations were sampled along the Saskatchewan River system during the ice-free season of 2014: three upstream locations were chosen as reference areas, while five downstream locations were selected ranging from immediately below the dam (2 km) to ~50 km downstream (Fig. 2.1). Selection of sampling locations was based on the premise that the hydrological impacts of hydropeaking attenuate with increasing distance from the dam (e.g. Moog 1993). Samples were taken once per month from May-September and corresponded to seasonal changes in flows and associated hydropeaking, as higher discharges occur during the spring freshet (May to mid-July) compared to the daily hydropeaking schedule followed later in the season (mid-July to

September) (Fig. 2.3). This period constitutes the bulk of the ice-free season (typically from April to November), and covers the period where water temperatures exceeded 12°C, above which most BMI taxa can grow and complete their life cycles. The location immediately below the dam is in a spillway channel (hereafter labeled “SW”), a part of the original river channel that was bypassed during construction. This channel normally consists of a series of small, isolated pools with little or no flow, except when discharges from the reservoir exceed the capacity of the power station ($\sim 1000 \text{ m}^3\text{s}^{-1}$), at which point the spillway gates are opened and these pools fully connect and flow. During the months of May and June 2014, the channel had high flow as the dam was releasing water from the spring melt, whereas the water returned to pools during the months of July, August, and September. Due to time constraints, we were unable to sample at DS1 in June.

2.3.2 BMI sampling, processing

The sampling methods in this study followed a modified protocol for large rivers as described in the Saskatchewan Northern Great Plains Ecosystem Health Assessment Manual (MoE and SWA 2012). Benthic macroinvertebrates were collected using a standard D-frame kicknet with a 500 μm mesh and 0.3 m opening. Each sampled location was divided into 3 sub-locations, each 100 m apart. These sub-locations were sampled in a downstream-upstream direction to avoid disturbing those that had not yet been sampled. Samples were taken from shore to the deepest wadeable depth or until 1 minute of sampling time elapsed. The net was positioned so that the opening faced upstream while the substrate was disturbed. The entire contents of each sweep were preserved using 95% ethanol. BI and EPT/C scores were calculated for all 3 sub-locations before calculating mean BI and EPT/C scores. Macroinvertebrates were identified to

genus or, when practical, to species. Saskatchewan-specific keys were used to identify Ephemeroptera (Webb 2002), Plecoptera (Doddall 1976), Trichoptera (Smith 1984), and Hemiptera (Brooks and Kelton 1967). All other taxa were identified using Merritt et al. (2008). Sample area was estimated using the size of the net (0.3 m) and the total length of each sampling transect, which was then used to estimate BMI densities.

2.3.3 Metrics for assessing river health

River health was estimated using taxa tolerance values from the Modified Hilsenhoff Biotic Index (Plafkin et al. 1989, in Mandaville, 2002) to calculate an overall biotic index (BI) for running waters score using the following formula:

$$BI = \frac{\sum x_i t_i}{n} \quad (\text{Eq. 1})$$

where x_i is the number of individuals of a species, t_i is the tolerance value of a species (Barbour et al. 1999), and n is the total number of individuals. A low BI score indicates low levels of environmental stress (typically pollution) as there are more sensitive taxa present, whereas a high BI score is indicative of a stressed environment with a high proportion of tolerant taxa. The tolerance values for macroinvertebrates were originally based on their resistance to organic pollution (Plafkin et al. 1989, in Mandaville 2002). In contrast, the use of EPT/C has been used extensively and is known for its applicability to a variety of environmental disturbances (Mandaville 2002). An EPT/C score was also calculated for each location, using the ratio of sensitive taxa (Ephemeroptera, Plecoptera, and Trichoptera) to Chironomidae (a relatively tolerant group) plus one (EPT/C+1) to account for areas without chironomids. To assess community diversity, a Shannon's diversity score was calculated for each location.

2.3.4 Water sampling and analysis

Water samples were taken for total nitrogen (TN), total phosphorus (TP), and dissolved organic carbon (DOC). Unfiltered water was collected for TN and TP, whereas water was filtered through a syringe filter (0.7 μm) for DOC. Samples were frozen immediately upon returning from sampling. To estimate total suspended solids (TSS), a water sample was taken subsurface at the deepest wadeable depth and filtered through a pre-weighed glass fibre filter (GFF) to a maximum of 1000 mL. Samples were frozen, then TSS was determined gravimetrically, after drying to a constant weight. Other water quality parameters such as pH, conductivity, and turbidity were measured on-site using probes (pH and conductivity, Hannah Instruments) and portable devices (turbidity, Lamotte). Temperature loggers were deployed at all locations, except the first location immediately downstream of the spillway (DS1), to record changes at 15 minute intervals. Various habitat parameters were recorded during each visit, including substrate composition percentages, relative water level, and percent abundance of macrophytes.

Chlorophyll *a* was quantified for both periphyton and phytoplankton. Periphyton was sampled by first removing from the water, then scrubbing the surface of, submerged rocks, wood, sticks, or vegetation (listed from most preferred to least preferred substrate) using a toothbrush and a disc with a predetermined cutout of known area. If no submerged objects were in an area, a syringe tube (25 mm diameter) was used to collect benthic sediment and approximately 5 mm of the top layer was scraped off and transferred to a small Ziploc bag containing a small amount of water. A total of 3 periphyton replicates were taken from each location. The water in each Ziploc bag was then filtered through GFFs and frozen prior to analysis. Phytoplankton was sampled by

taking a subsurface sample of water from the deepest wadeable depth in a graduated cylinder by inverting the cylinder as it was placed under the surface, then turning it right side up once it was completely submerged. Samples were filtered onto GFFs and frozen. Both attached and suspended algal samples were analysed by extracting chlorophyll from the GFFs using hot ethanol followed by analysis on a benchtop Trilogy fluorometer (Turner Designs).

2.3.5 Statistical analysis

Several approaches were used to compare BMI assemblages among locations: canonical correspondence analysis (CCA), analysis of similarities (ANOSIM), and similarity percentages (SIMPER). The primary objective of these methods was to perform upstream versus downstream comparisons. CCA was done using R (version 3.4.2; R Project for Statistical Computing, Vienna, Austria) with the *vegan* and *lmom* packages. The ANOSIM and SIMPER analyses were done using PRIMER Version 6.1.13 (PRIMER-E software, Plymouth, United Kingdom; Clarke and Warwick, 2001). The scientific community remains divided on whether or not to remove rare taxa from statistical analyses, with authors arguing for and against it. One of the main reasons for removing rare taxa from multivariate datasets is that they can create ‘noise’ that might obscure otherwise clear patterns (Gauch 1982, Norris et al. 1982, Marchant 1990, Reece et al. 2001). The main counterargument is that their removal can negatively impact otherwise significant differences in the dataset (Cao et al. 2001). Prior to performing these analyses, we chose to adjust the community matrix by removing rare taxa that had a total abundance of ≤ 5 and had an occurrence of ≤ 4 in the matrix. Non-benthic taxa were completely removed from the dataset. Using this subset, the data were $\log_{(n+1)}$

transformed and used to calculate a taxa-by-taxa dissimilarity matrix using the Bray-Curtis dissimilarity metric. Other metrics were considered, including Gower and Euclidean; however, the former approach underestimates community dissimilarity by upweighting uncommon taxa and downweighting abundant taxa, whereas the latter technique considers BMI absence (zeros) to be a similarity among communities (Clarke and Warwick 2001). Furthermore, the Bray-Curtis dissimilarity metric is commonly used for analyzing BMI assemblages (Clarke and Warwick 2001, Phillips et al. 2015), and so was deemed the most appropriate choice for our study.

ANOSIM was used to compare average rank similarities of the benthic communities upstream and downstream of E.B. Campbell dam. The output of the analysis includes a calculated R-statistic, which varies between 0 and 1, and the significance value of that statistic. The latter is obtained by testing whether the observed numbers differ from a distribution based on the null hypothesis (no differences between upstream and downstream benthic communities; Clarke and Warwick 2001). A high R-statistic indicates a larger difference between the two assemblages. To evaluate which taxa were most responsible for any dissimilarity between upstream and downstream locations, a family-level similarity percentages (SIMPER) analysis was performed. All tests were done separately for each month of analysis. As our main purpose was to identify whether hydropeaking affects the BMI assemblage downstream, SW was omitted from the CCA, ANOSIM, and SIMPER analyses as it was not affected by hydropeaking.

2.4 Results

2.4.1 BMI community analyses

A total of 67,506 individuals from 237 different invertebrate taxa were collected over the course of this study. Average macroinvertebrate density ranged from 39 to 2,477 individuals m^{-2} (Fig. 2.4) and number of taxa at each location ranged from 13 to 53 (Fig. 2.5). Mayflies were the most common taxa, with 17 families overall (Fig. 2.5), and ranged in abundance from 2 to 8,816 per location. On the whole, taxonomic richness tended to increase throughout the sampling season, with the greatest change shown at SW, 2 km downstream of the dam (Fig. 2.5). Further downstream (8-30km from dam), richness was generally lower relative to sites US1 to US3 and SW. In contrast, DS4 (+53 km) was comparable to upstream locations.

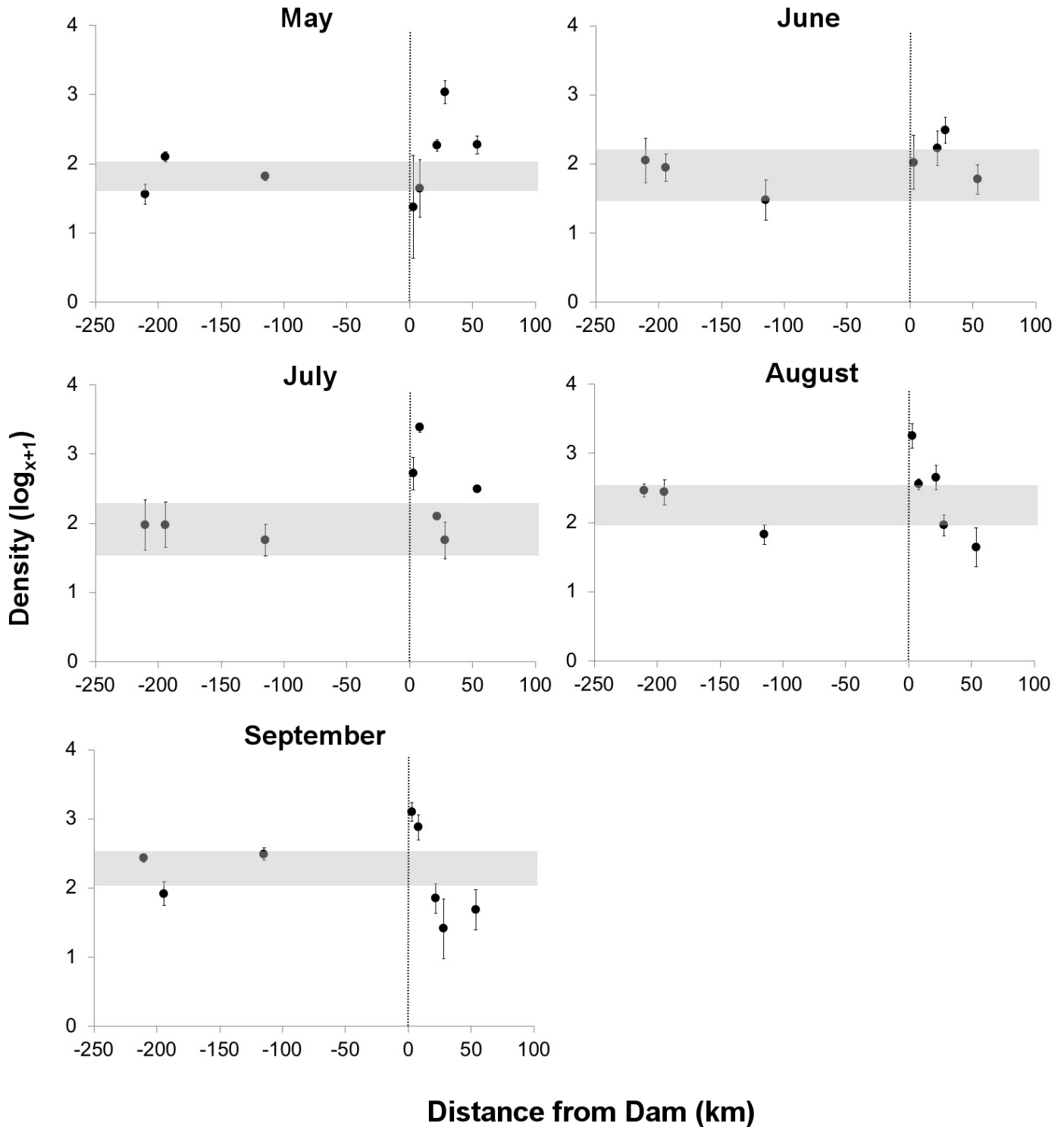


Fig. 2.4: Scatterplots illustrating BMI density (\log_{x+1}) across the sampling area versus the distance of each location from E.B. Campbell Dam, shown as a vertical dotted line, in kilometers. Distances upstream/downstream of the dam are depicted as negative/positive numbers, respectively. Grey boxes indicate the 95% CI for the upstream locations. Distances from the dam for each location are as follows: US1 (-210 km), US2 (-194 km), US3 (-114 km), SW (+2 km), DS1 (+8 km), DS2 (+21 km), DS3 (+28 km), and DS4 (+53 km).

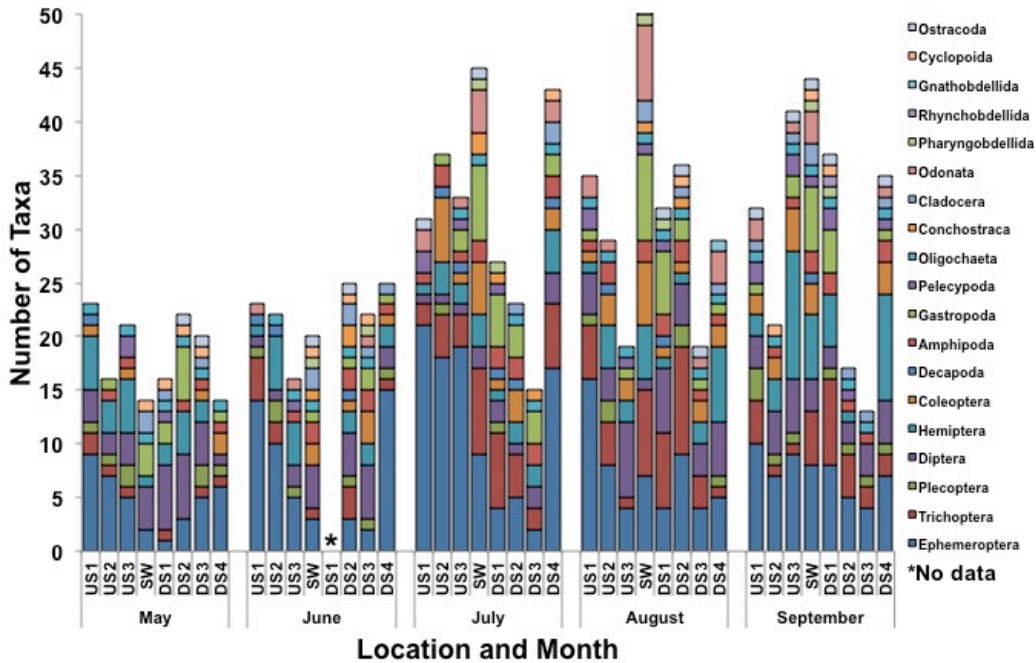


Fig. 2.5: A stacked column graph illustrating the number of BMI orders present at each location across the months of May to September 2014.

2.4.2 Water quality

In addition to changes in flow regime, the biophysical environment appeared to differ below the dam. In the river upstream of the dam, higher turbidity, TSS, and suspended chlorophyll *a* concentrations were typically observed relative to downstream (Table 2.1). In contrast, the downstream reaches appeared to have more benthic chlorophyll *a* and marginally higher DOC values compared to upstream. Higher concentrations of benthic chlorophyll immediately below the dam were observed between July and September (Table 2.1). Total phosphorus was marginally higher at upstream locations, whereas pH, conductivity, and total nitrogen were similar among locations both upstream and downstream. Mean daily temperature appeared to fluctuate more at upstream locations compared to the regulated regime observed downstream (Fig. 2.6).

Additionally, warmer and cooler temperatures were recorded upstream from May-July and August-September, respectively, relative to downstream locations (Fig. 2.6).

Table 2.1

WATER QUALITY DATA FOR ALL LOCATIONS

| | | US1 (-210 km) | US2 (-194 km) | US3 (-114 km) | SW (+2 km) | DS1 (+8 km) | DS2 (+21 km) | DS3 (+28 km) | DS4 (+53 km) |
|----------------------|------|------------------|------------------|------------------|---------------|----------------|-----------------|-----------------|-----------------|
| pH | May | 8.1 | 8.2 | 8.2 | 8.0 | 8.1 | 8.1 | 8.1 | 8.1 |
| | June | 8.5 | 8.6 | 8.7 | 8.7 | --- | 8.8 | 8.9 | 8.6 |
| | July | 8.6 | 8.6 | 8.8 | 8.9 | 8.4 | 8.4 | 8.7 | 8.7 |
| | Aug | 8.6 | 8.8 | 8.3 | 8.5 | 8.1 | 8.8 | 8.7 | 8.5 |
| | Sept | 8.7 | 8.5 | 8.7 | 9.4 | 8.3 | 8.7 | 8.6 | 8.5 |
| Mean ± 1 SD | | 8.5 ± 0.2 | 8.5 ± 0.2 | 8.5 ± 0.3 | 8.7 ± 0.5 | 8.2 ± 0.2 | 8.6 ± 0.3 | 8.6 ± 0.3 | 8.5 ± 0.2 |
| Conductivity (µS) | May | 476 | 488 | 490 | 448 | 463 | 464 | 473 | 385 |
| | June | 409 | 466 | 322 | 332 | --- | 323 | 323 | 266 |
| | July | 440 | 462 | 336 | 291 | 478 | 370 | 473 | 270 |
| | Aug | 466 | 438 | 446 | 365 | 441 | 464 | 461 | 295 |
| | Sept | 460 | 474 | 425 | 359 | 496 | 501 | 499 | 481 |
| Mean ± 1 SD | | 450 ± 27 | 466 ± 18 | 404 ± 72 | 359 ± 58 | 470 ± 23 | 424 ± 75 | 446 ± 70 | 339 ± 93 |
| Turbidity (NTU) | May | 26.9 | 11.4 | 23.8 | 4.6 | 11.8 | 8.9 | 4.5 | 11.3 |
| | June | 37.5 | 7.2 | 39.6 | 2.2 | --- | 3.8 | 2.1 | 8.7 |
| | July | 16.1 | 7.8 | 12.9 | 3.5 | 4.1 | 4.7 | 3.9 | 13.1 |
| | Aug | 13.2 | 6.2 | 10.1 | 5.6 | 3.1 | 3.7 | 3.4 | 4.4 |
| | Sept | 6.3 | 5.4 | 8.6 | 2.7 | 3.5 | 2.7 | 2.7 | 5.3 |
| Mean ± 1 SD | | 20.0 ± 12.3 | 7.6 ± 2.3 | 19.0 ± 13.0 | 3.7 ± 1.4 | 5.6 ± 4.1 | 4.8 ± 2.4 | 3.3 ± 0.9 | 8.6 ± 3.7 |

| | | | | | | | | | |
|---|------|-----------------|-----------------|-----------------|-----------------|-----------------|-----------------|----------------|-----------------|
| TSS (mg/L) | May | 112.0 | 39.0 | 83.8 | 8.3 | --- | 17.1 | 7.6 | 75.0 |
| | June | 106.7 | 21.0 | 110.8 | 8.1 | --- | 4.6 | 3.6 | 36.0 |
| | July | 71.4 | --- | 31.7 | 3.7 | 4.2 | 9.5 | 6.9 | 68.1 |
| | Aug | 27.6 | 9.9 | 28.5 | 3.6 | 4.0 | 4.6 | 12.8 | 9.4 |
| | Sept | --- | 10.4 | 29.7 | 7.5 | 3.7 | 3.0 | 0.9 | 5.9 |
| Mean \pm 1 SD | | 79.4 \pm 39.0 | 20.1 \pm 13.6 | 56.9 \pm 38.1 | 6.2 \pm 2.4 | 4.0 \pm 0.3 | 7.8 \pm 5.8 | 6.4 \pm 4.5 | 38.9 \pm 32.1 |
| Chlorophyll a, benthic (mg/m ²) | May | 1.3 | 0.3 | 0.2 | 2.7 | 25.5 | 2.3 | 2.5 | 8.1 |
| | June | 1.4 | --- | 0.3 | 1.2 | --- | 8.1 | 13.8 | 4.5 |
| | July | --- | 5.0 | 1.5 | 54.0 | 12.9 | 6.5 | 4.9 | 1.9 |
| | Aug | 3.9 | 23.5 | 1.7 | 29.7 | 104.5 | 5.1 | 23.1 | 2.0 |
| | Sept | 4.2 | 24.0 | 9.3 | 29.8 | 53.4 | 80.9 | 9.7 | 14.3 |
| Mean \pm 1 SD | | 2.7 \pm 1.6 | 13.2 \pm 12.3 | 2.6 \pm 3.8 | 23.5 \pm 22.0 | 49.1 \pm 40.6 | 20.6 \pm 33.8 | 10.8 \pm 8.1 | 6.2 \pm 5.2 |
| Chlorophyll a, suspended (μ g/L) | May | 17.0 | 2.7 | 9.1 | 4.2 | 7.3 | 9.1 | 4.8 | 2.7 |
| | June | 16.6 | 6.9 | 13.0 | 6.0 | --- | 6.3 | 7.4 | 5.6 |
| | July | 11.9 | 15.0 | 12.6 | 8.7 | 2.9 | 2.7 | 1.8 | 2.1 |
| | Aug | 9.0 | 3.8 | 12.5 | 4.0 | 1.1 | 2.2 | 2.7 | 5.0 |
| | Sept | 11.2 | 6.2 | 9.3 | 1.6 | 3.7 | 1.9 | 2.2 | 2.6 |
| Mean \pm 1 SD | | 13.1 \pm 3.5 | 6.9 \pm 4.8 | 11.3 \pm 1.9 | 4.9 \pm 2.6 | 3.8 \pm 2.6 | 4.4 \pm 3.1 | 3.8 \pm 2.3 | 3.6 \pm 1.6 |
| Total Phosphorus (mg/L) | May | 0.09 | 0.05 | 0.07 | 0.02 | 0.02 | 0.02 | 0.01 | 0.03 |
| | June | 0.07 | --- | 0.08 | --- | --- | 0.02 | 0.02 | 0.04 |
| | July | --- | --- | 0.03 | 0.03 | 0.02 | 0.02 | 0.02 | 0.08 |
| | Aug | 0.04 | 0.02 | 0.03 | 0.03 | 0.03 | 0.02 | 0.02 | 0.03 |
| | Sept | 0.03 | 0.01 | 0.01 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 |

| | | | | | | | | | |
|---------------------------------|------------------------|------------------------|-----------------------------|------------------------------|------------------------------|--------------------------|---------------|-----------------------------|-----------------|
| Mean \pm 1 SD | | 0.06 \pm 0.03 | 0.03 \pm 0.02 | 0.04 \pm 0.03 | 0.03 \pm 0.01 | 0.02 \pm 0.01 | 0.02 | 0.02 | 0.04 \pm 0.02 |
| Total Nitrogen (mg/L) | May | 2.0 | 0.6 | 0.7 | 1.1 | 1.0 | 0.8 | 0.7 | 0.9 |
| | June | 0.4 | --- | 0.4 | --- | --- | 0.5 | 0.5 | 0.5 |
| | July | --- | --- | 0.8 | 1.0 | 0.6 | 0.6 | 0.6 | 1.0 |
| | Aug | 0.2 | 0.6 | 0.2 | 1.0 | 0.2 | 0.4 | 0.2 | 0.5 |
| | Sept | 0.4 | 0.7 | 0.5 | 1.0 | 0.5 | 0.3 | 0.4 | 0.5 |
| Mean \pm 1 SD | | 0.8 \pm 0.8 | 0.6 \pm 0.1 | 0.5 \pm 0.2 | 1.0 \pm 0.1 | 0.6 \pm 0.3 | 0.5 \pm 0.2 | 0.5 \pm 0.2 | 0.7 \pm 0.2 |
| Dissolved Organic Carbon (mg/L) | May | 5.8 | 3.5 | 5.1 | 6.7 | 5.9 | --- | 4.4 | 8.0 |
| | June | --- | 3.5 | 5.8 | --- | --- | 5.0 | 4.4 | 8.7 |
| | July | 4.8 | 3.6 | --- | 11.9 | 5.8 | 5.3 | 5.0 | --- |
| | Aug | 3.5 | 3.7 | 4.8 | 12.4 | 5.3 | 5.1 | 5.4 | 7.7 |
| | Sept | --- | --- | --- | --- | --- | --- | --- | --- |
| Mean \pm 1 SD | | 4.7 \pm 1.2 | 3.6 \pm 0.1 | 5.2 \pm 0.5 | 10.3 \pm 3.2 | 5.7 \pm 0.3 | 5.1 \pm 0.2 | 4.8 \pm 0.5 | 8.1 \pm 0.5 |
| Dominant Substrate | Sand (35%), silt (25%) | Sand (30%), silt (25%) | Fine silt (60%), clay (40%) | Boulders (35%), cobble (40%) | Boulders (30%), cobble (35%) | Gravel (20%), sand (35%) | Sand (85%) | Sand (40%), fine silt (60%) | |

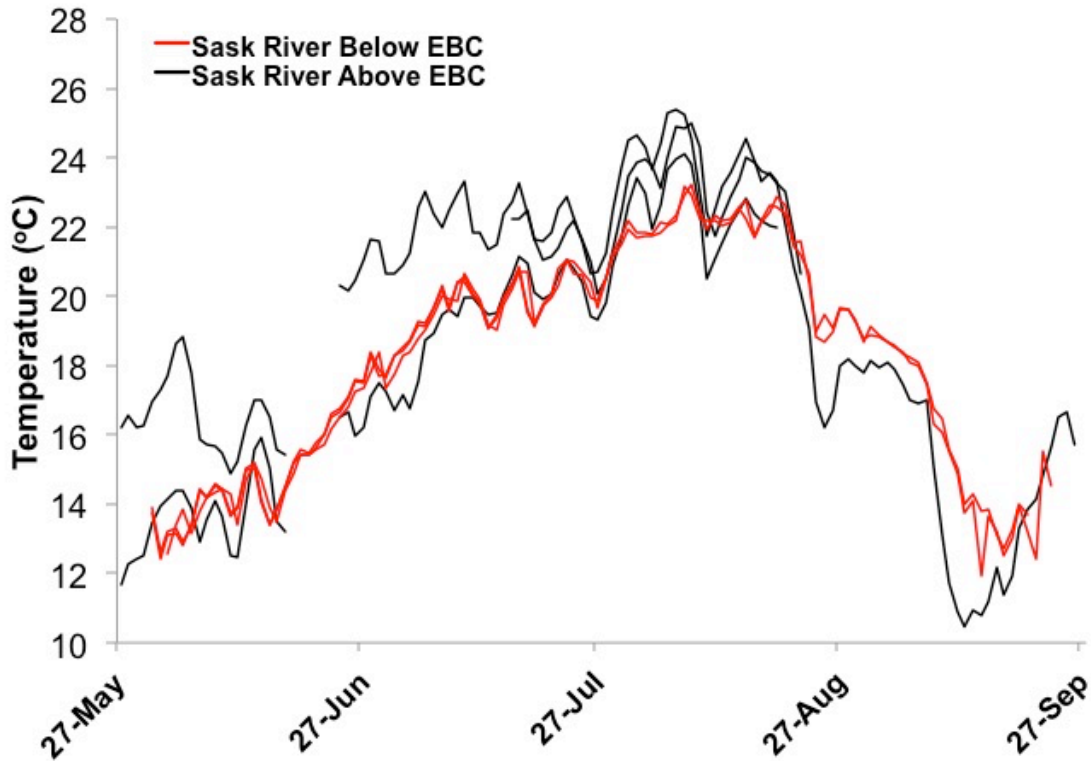


Fig. 2.6: Daily mean temperature of the Saskatchewan River both upstream and downstream of E.B. Campbell Dam for the period of May 27 – September 26, 2014. Temperature was measured upstream at US1 (-210 km), US2 (-194 km), and US3 (-114 km) while downstream measurements were taken at DS2 (+21 km), DS3 (+28 km), and DS4 (+53 km).

2.4.3 River health metrics

BMI densities at upstream locations were relatively similar throughout the sampling season, whereas taxa had sharply increased densities at locations immediately below the dam (SW and DS1; +2 and +8 km) from July-September (Fig. 2.4), largely due to an increase in tolerant taxa. Further downstream (+21 to +53 km), densities were similar to those found upstream (Fig. 2.4).

Biotic index (BI) values were generally higher at SW (+2 km) and DS1 to DS3 (+8 to +28 km) (Fig. 2.6). Mean BI values ranged from 3.40 to 5.64 upstream of the dam across seasons and were higher below the dam (range 4.45 to 7.65), especially in May

and June but this varied by location. DS4 (+53 km) had BI values that were within the range (95% confidence interval) of those observed at the upstream reaches.

Taxonomic richness at locations below the dam was comparable to those upstream (Fig. 2.5). Shannon's diversity scores, however, were generally higher at upstream locations compared to those immediately downstream of the dam, with only DS4 (+53 km) having values comparable to those upstream (Fig. 2.8).

Mean EPT/C+1 values were highest upstream of the dam (0.16 to 43.42 versus 0.03 to 7.39 downstream) and, like the BI, had values comparable to the upstream reference locations at the location furthest from the dam (Fig. 2.9). In a comparison of monthly BI to EPT/C+1 scores, a general negative correlation was observed in the first three months of sampling, but not August and September (Fig. 2.9).

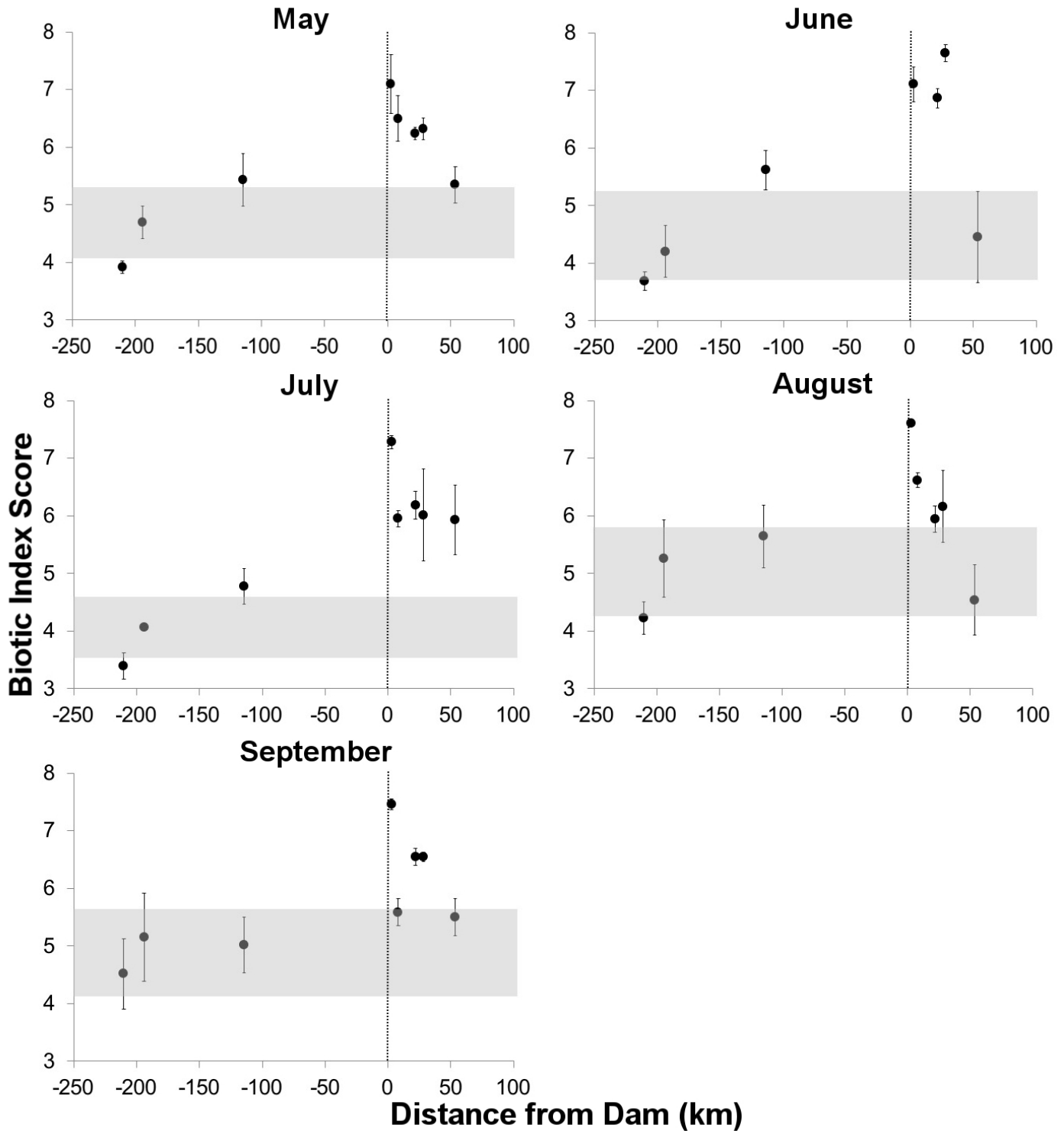


Fig. 2.7: Scatterplots showing the biotic index score for each location from May-September 2014 versus the distance of each location from E.B. Campbell Dam. Negative/positive distances represent kilometers upstream/downstream of the dam. Grey boxes represent the 95% CI for upstream BI scores. The vertical dotted line indicates the location of the dam. Distances from the dam for each location are as follows: US1 (-210 km), US2 (-194 km), US3 (-114 km), SW (+2 km), DS1 (+8 km), DS2 (+21 km), DS3 (+28 km), and DS4 (+53 km).

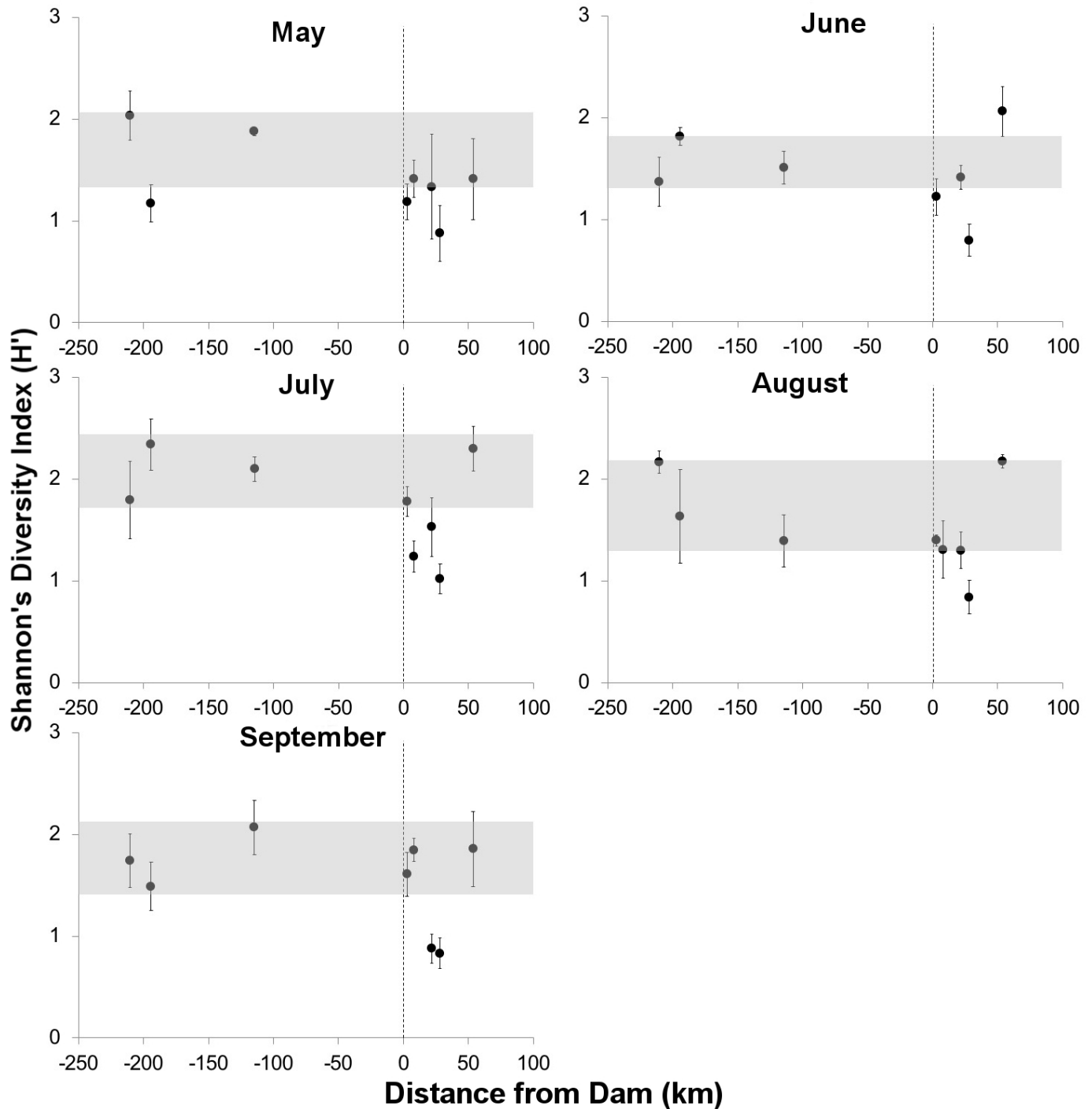


Fig. 2.8: Monthly scatterplots from May-September 2014 comparing the Shannon's diversity index scores of each sample location to distance in kilometers from E.B. Campbell dam. Negative/positive distances represent kilometers upstream/downstream of the dam. The E.B. Campbell Dam's location is represented by a vertical dotted line. Grey boxes indicate the 95% CI for upstream diversity scores. Distances from the dam for each location are as follows: US1 (-210 km), US2 (-194 km), US3 (-114 km), SW (+2 km), DS1 (+8 km), DS2 (+21 km), DS3 (+28 km), and DS4 (+53 km).

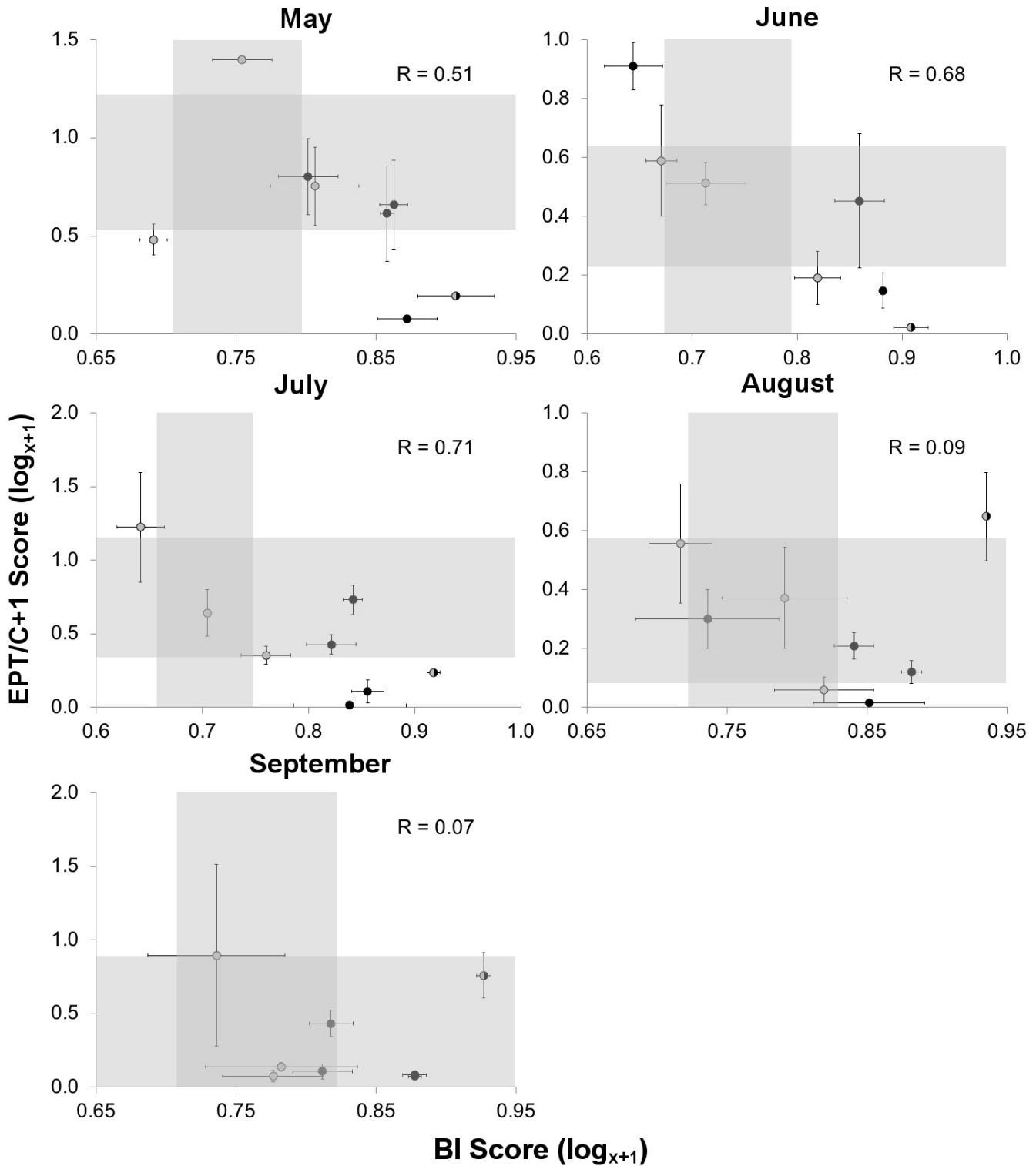


Fig. 2.9: Scatterplots depicting the BI and EPT/C+1 scores for sampled locations along the Saskatchewan River for the months of May-September 2014. Note the negative correlation between the two metrics from May-July that breaks down during the months of August and September. Grey points represent upstream locations, black points are downstream, and the split point indicates the spillway site immediately below the dam. Grey boxes indicate the 95% CI for upstream BI and EPT/C+1 scores.

The benthic assemblage at SW (+2 km) and DS1 (+8 km) varied from other locations, with high densities of tolerant taxa including amphipods and *Caenis* mayfly larvae from July-September. As these taxa are typically associated with lentic environments, the reach of the river downstream where these species were present was deemed the 'lentic impact zone.' While several species of *Sigara* were collected at downstream locations, *S. lineata* was only found at upstream locations and at the furthest downstream location from the dam. To assess whether high amphipod densities at downstream locations were the main influence for increased BI scores, the scores for all locations were calculated in the absence of amphipods. This had little effect on BI scores across all locations, indicating that although they were present in high densities, amphipods were not the primary driver of BI scores.

2.4.4 Statistical interpretations

In our CCA of the upstream and downstream BMI communities, we found that locations upstream of the dam were associated with small substrate grain sizes as well as higher TSS, turbidity, and suspended chlorophyll *a* concentrations compared to downstream locations, where benthic chlorophyll *a* concentrations were higher and substrate grain sizes were larger (Fig. 2.10). The ordination illustrates that the upstream locations are separated from those sampled downstream, and that those found immediately below the dam form distinct clusters, whereas further locations (DS4; >50 km) become more similar to upstream communities (Fig. 2.10). The ANOSIM results support these findings, as upstream and downstream locations had significantly different assemblages throughout all months even when including the furthest downstream location (Table 2.2). According to our SIMPER analysis, Corixidae (Hemiptera),

Baetidae (Ephemeroptera), Chironomidae (Diptera), and Hydropsychidae (Trichoptera) were among the families that contributed the most to differentiating upstream locations from those downstream (Table 2.2).

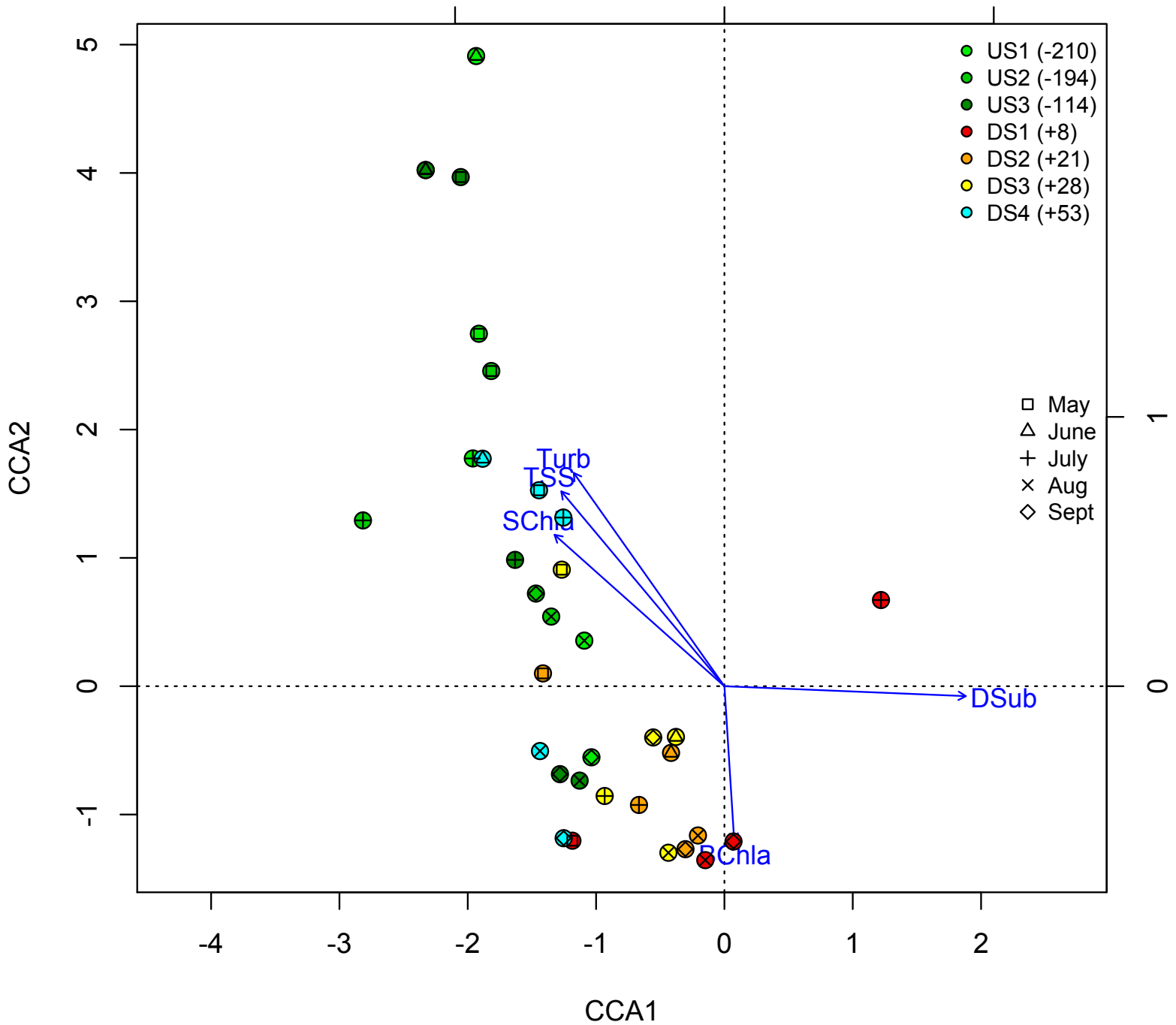


Fig. 2.10: Canonical correspondence analysis illustrating the differences in benthic macroinvertebrate community structure at 7 locations along the Saskatchewan River. Locations upstream of E.B. Campbell dam are shown in green, whereas those downstream are red, orange, yellow, and light blue. Numbers in brackets represent distance upstream (negative) and downstream (positive) from E.B. Campbell Dam in kilometers. Turb = turbidity, TSS = total suspended solids; SChla = suspended chlorophyll *a*; BChla = benthic chlorophyll *a*; DSub = dominant substrate

Table 2.2

MONTHLY ANOSIM RESULTS AND SIMPER DISSIMILARITY PERCENTAGES FOR SELECTED TAXA

| May | | | | June | | | |
|-----------------|---|--------------------|----------------------|---|--------------------|--------------------|----------------------|
| ANOSIM | <i>R</i> -statistic = 0.394 <i>p</i> = 0.001 | | | <i>R</i> -statistic = 0.166 <i>p</i> = 0.029 | | | |
| Family | U/S Mean Abundance | D/S Mean Abundance | Percent Contribution | Family | U/S Mean Abundance | D/S Mean Abundance | Percent Contribution |
| Corixidae | 2.91 | 0.56 | 16.69 | Corixidae | 2.48 | 0.78 | 14.55 |
| Baetidae | 2.67 | 2.96 | 14.42 | Chironomidae | 2.79 | 2.18 | 13.60 |
| Perlodidae | 1.58 | 0.48 | 9.61 | Baetidae | 2.31 | 2.09 | 10.30 |
| Oligochaeta | 0.48 | 1.59 | 8.75 | Oligochaeta | 1.09 | 0.82 | 9.49 |
| Chironomidae | 1.68 | 2.78 | 8.56 | Metretopodidae | 1.32 | 0.23 | 9.09 |
| Metretopodidae | 1.51 | 0.40 | 8.15 | Perlodidae | 0.93 | 0.81 | 6.91 |
| Baetiscidae | 0.89 | 0.39 | 5.56 | Heptageniidae | 0.90 | 0.33 | 6.59 |
| Leptophlebiidae | 0.15 | 0.70 | 4.62 | Lymnaeidae | 0.00 | 0.67 | 4.93 |
| Dolichopodidae | 0.12 | 0.66 | 4.02 | Hydropsychidae | 0.48 | 0.15 | 3.67 |
| Lymnaeidae | 0.12 | 0.51 | 2.98 | Baetiscidae | 0.00 | 0.45 | 3.44 |

| July | | | | August | | | |
|-----------------|---|--------------------|----------------------|---|--------------------|--------------------|----------------------|
| ANOSIM | <i>R</i> -statistic = 0.297 <i>p</i> = 0.002 | | | <i>R</i> -statistic = 0.291 <i>p</i> = 0.004 | | | |
| Family | U/S Mean Abundance | D/S Mean Abundance | Percent Contribution | Family | U/S Mean Abundance | D/S Mean Abundance | Percent Contribution |
| Baetidae | 2.77 | 2.96 | 9.25 | Corixidae | 4.00 | 1.01 | 10.38 |
| Chironomidae | 2.55 | 4.34 | 8.31 | Baetidae | 2.98 | 0.75 | 8.44 |
| Corixidae | 2.81 | 2.22 | 8.07 | Hydropsychidae | 1.51 | 2.94 | 8.27 |
| Hydropsychidae | 1.20 | 1.85 | 6.84 | Chironomidae | 4.22 | 4.99 | 7.24 |
| Polymitarcyidae | 1.80 | 0.00 | 6.63 | Heptageniidae | 1.93 | 2.06 | 7.16 |
| Heptageniidae | 2.09 | 0.66 | 6.07 | Dogielinotidae | 0.82 | 2.08 | 6.32 |
| Caenidae | 1.69 | 0.56 | 5.82 | Oligochaeta | 1.16 | 1.82 | 5.83 |
| Dogielinotidae | 0.45 | 1.50 | 4.59 | Leptohiphidae | 1.23 | 0.37 | 4.12 |
| Oligochaeta | 0.47 | 0.67 | 4.32 | Ephydriidae | 0.85 | 0.06 | 3.46 |
| Leptohiphidae | 0.90 | 1.04 | 3.49 | Caenidae | 0.00 | 1.29 | 3.43 |

MONTHLY ANOSIM RESULTS AND SIMPER DISSIMILARITY
 PERCENTAGES FOR SELECTED TAXA (CONTINUED)

| September | | | |
|-----------------------------|-----------------------|-----------------------|-------------------------|
| <i>R</i> -statistic = 0.217 | | | |
| <i>p</i> = 0.011 | | | |
| Family | U/S Mean Abundance | D/S Mean Abundance | Percent Contribution |
| Corixidae | 4.66 | 1.23 | 14.05 |
| Chironomidae | 3.74 | 4.65 | 7.93 |
| Hydropsychidae | 1.24 | 2.50 | 7.85 |
| Dogielinotidae | 0.91 | 2.11 | 6.59 |
| Oligochaeta | 1.91 | 1.27 | 6.59 |
| Dytiscidae | 1.61 | 0.06 | 5.99 |
| Heptageniidae | 1.15 | 1.39 | 5.39 |
| Leptophlebiidae | 1.50 | 0.91 | 5.30 |
| Baetidae | 1.06 | 0.90 | 4.27 |
| Sphaeriidae | 1.01 | 0.33 | 3.37 |

2.5 Discussion

Large rivers are among the most impacted freshwater ecosystems in the world (Nilsson et al. 2005, Poff et al. 2007). Hydroelectric dams are common along these systems, and although the effects of some dams on benthic macroinvertebrate assemblages have been previously described (Lehmkuhl 1972, Phillips et al. 2015, Phillips et al. 2016), the concerns of hydropeaking operations have only recently been acknowledged, despite the large number of dams that practice hydropeaking (e.g. Jones 2013a, Kennedy et al. 2016). The scale and scarcity of large rivers like the Saskatchewan have made it difficult to quantify the effects of anthropological disturbances, and traditional reference condition approaches often cannot be applied to these systems (Phillips et al. 2015). Additionally, the majority of hydropeaking studies have not considered the possible effects of seasonality on benthic communities as most of them are conducted in the late summer months (August-September) when the extent of flow variation can be high and the mean daily flows relatively low. This study utilized approaches for collecting benthic macroinvertebrates in large rivers (MoE and SWA 2012) and assessed the effects of hydropeaking on the system's health (Mandaville 2002) across much of the ice-free season. Key findings included altered benthic assemblages below the dam along with increased BI and decreased EPT/C scores compared to upstream locations, indicative of deteriorated river health. Seasonality in hydropeaking was reflected in the changes to downstream BMI community tolerance, density, and diversity.

Immediately downstream of the dam in the spillway channel (SW; +2 km), the benthic assemblage consisted mainly of tolerant taxa usually found in lentic environments

(e.g. amphipods, *Caenis* mayfly larvae). These taxa were found in very high abundance from July-September after flow through this reach had ceased, but were absent from May-June samples when this channel was used as a spillway to accommodate high flows. Unlike the other downstream reaches, SW was not subjected to daily hydropeaking from July-September, resulting in little to no flow and much higher abundances of lentic taxa. Although it does not reflect daily hydropeaking, SW is still subjected to seasonal changes in flow and more likely reflects changes that would occur when large reservoirs are used primarily for extraction (e.g. irrigation supply) and only release water during extreme high flow events.

Because amphipods have relatively high tolerance to environmental disturbances and were found in high densities immediately below the dam, these areas had correspondingly high BI values. Surprisingly, the removal of amphipods did little to change the BI scores across all locations, even when their numbers were in the thousands. This exploratory analysis provided insight into how the downstream benthic communities had changed in composition in terms of their tolerance to disturbance and that the dominant taxon was not solely responsible for that change. The 'lentic impact zone' extended several kilometers downstream, and lentic taxa abundance generally decreased with distance from the dam. Even so, some of these typically lentic taxa were observed at DS4, 53 kilometers downstream. The macroinvertebrate community at the next downstream location, DS1 (+8 km), consisted of high densities of taxa in the filter-feeding functional group (e.g. Hydropsychidae) from mid- to late summer, similar to findings in the regulated Magpie River system (Jones 2013a). As proposed by Richardson

and Mackay (1991), these filter-feeding communities are probably sustained by plankton that originated in the reservoir (Table 2.1).

Further downstream (>20 km from the dam), BMI abundance and diversity were much lower. Macrophytes, which often harbor high BMI abundance relative to bare substrate in deeper parts of the channel (Needham 1934), were absent at sites between 21 and 53 km downstream, likely due to the change in substrate (cobble and coarse gravel to sand) and the rapid fluctuations in water level resulting from daily hydropeaking (J. Mihalicz, pers. obs.). Bejarano et al. (2018) reviewed the effects of hydropeaking on riverine plants and concluded that abrupt changes in water level and flow have a marked effect on vegetation in the riparian zone. This suggests that the absence of macrophyte growth at our downstream locations is likely due to hydropeaking. Two locations in particular, DS2 (+21 km) and DS3 (+28 km), showed a notable decrease in BMI density and diversity (Fig. 2.4 and 2.5) as well as high BI (Fig. 2.7) and low EPT/C scores (Fig. 2.9) in August and September. These changes may be effects of daily hydropeaking at these two reaches, where the fluctuation in discharge is most severe in the late summer. Food sources and habitat complexity were mainly limited to occasional allochthonous inputs such as leaves and woody debris (J. Mihalicz, pers. obs.), and the taxa found at these far-downstream locations included shredders (e.g. Plecoptera) and scrapers (e.g. Heptageniidae), but relatively few filter-feeders. It is recommended that the lentic impact zone be accounted for in future projects and sampling designs that assess the effects of impoundments on river health, as its size may vary annually and seasonally.

Water quality parameters were found to differ substantially between upstream and downstream locations, especially chlorophyll *a* (benthic and suspended) and turbidity

(Table 2.1). These differences may contribute to the change in benthic assemblage composition observed below the dam, as seen in the CCA (Fig. 2.10). Higher benthic chlorophyll *a* concentrations at locations immediately below the dam, likely owing to greater light penetration in clearer waters, translate to greater food source availability for BMIs, which may explain the dense populations of tolerant taxa found there. Lower turbidity values downstream of the dam are likely due to the loss of particulate matter through sedimentation in the reservoir. This corresponds to lower concentrations of suspended chlorophyll *a*. These changes highlight how additional physical alterations to river habitat resulting from hydropeaking can consequently affect water quality and, ultimately, the biotic community (Melcher et al. 2017).

A large proportion of the BMIs collected in this study were aquatic insects (>68%). Of these, over 99% were emergent taxa. In the northern Great Plains, it is believed that the majority of emergent aquatic insects probably have univoltine (Ephemeroptera (Clifford 1982), Trichoptera (Smith 1984), Plecoptera (Dosdall 1976)) or semivoltine life cycles (some Ephemeroptera (Clifford 1982) and Plecoptera (Dosdall 1976)). Aquatic insects with the former type of life cycle have a cohort that emerges as adults once per year, usually in the ice-free season, whereas those with semivoltine life cycles spend several years as larvae before emerging. Information on Saskatchewan Diptera and Hemiptera life cycles could not be found, although it is probable that most species are univoltine. Clifford (1982) found two types of univoltine life cycles among Saskatchewan Ephemeroptera: summer and winter. Species of the summer type overwinter as eggs and hatch in the spring with rapid growth during the summer months. In contrast, eggs of winter species hatch soon after being laid and the larvae are the

overwintering stage. Similarly, Dossall (1976) found that some stonefly species in Saskatchewan emerge from late winter to early spring just as the ice begins to melt, while others emerge throughout May and June. Additionally, some winter stonefly nymphs enter a state of diapause through the summer months, burrowing into the sediment to avoid warm water temperatures. Clearly, both types of Plecoptera would probably not be collected in summer samples, and this was the case in our study. Given the disproportionate effects of hydropeaking on EPT (Kennedy et al. 2016), we suggest that future studies consider the effects of hydropeaking during the winter and early spring when the eggs and young larvae of these species are probably the most vulnerable.

Although literature on the effects of hydropeaking on river biota has recently begun to appear more frequently (e.g. Jones 2014, Kennedy et al. 2016, Melcher et al. 2017) the macroinvertebrates in many studies are collected in late summer or early autumn. In doing so, it is probable that many emergent insect species are not accounted for, and thus any effects of hydropeaking on these taxa remain unknown. Benthic assemblages are known to vary not only from one year to the next, but seasonally as well (Peterson et al. 2017). Our study incorporated a seasonal sampling design capturing much of the ice-free season, allowing collection and assessment of taxa whose presence varied seasonally. As a result, we were able to observe changes in the benthic community from one month to another, and found that the assemblage downstream of the dam was altered with high densities of disturbance-tolerant taxa relative to upstream locations. As the composition of the downstream macroinvertebrate communities changed from one month to the next with the emergence of some taxa, the overall tolerance of the community remained relatively high compared to upstream reaches. Yet differences were greatest in

the months May to July. This suggests that late summer/autumn sampling of BMI communities may underestimate the general effects of hydropeaking. The interplay between mean flow conditions, changes in extent of hydropeaking, and biotic sensitivity should be considered in individual rivers when considering sampling design.

The results of our study have identified that the BMI assemblage downstream of E.B. Campbell dam is significantly different than the one found upstream. This is not surprising, as the impacts of dams on BMI assemblages are already well understood. For example, studies of the Gardiner Dam on the South Saskatchewan River have found that its hypolimnetic releases cause thermal changes that affect the river for hundreds of kilometers and thus alter the BMI communities found there (Lehmkuhl 1972, Phillips et al. 2015). However, we have found evidence suggesting that daily hydropeaking may further alter BMI assemblages with its rapid changes in discharge. This was observed at DS2 (+21 km) and DS3 (+28 km) in August and September where daily hydropeaking was most severe because of a low-pitched shoreline that results in large changes in water level. At these two locations, BMI diversity and density were low, and BI scores were high relative to DS1 (+8 km) and DS4 (+53 km) (Fig. 2.4, 2.5, 2.7). Our CCA shows that these two sites tend to be similar to each other yet distinct from other downstream sites (Fig. 2.10). It was equally important that we assessed which taxa were responsible for driving the differences not only between the upstream and downstream reaches, but also between locations within each treatment. While some species of Corixidae were collected downstream of the dam, *Sigara lineata* was only found at upstream sites and at the location furthest downstream from the dam. Although Mandaville (2002) does not list a tolerance value specific to *S. lineata*, its presence among sensitive taxa upstream and its

absence downstream suggests that this species is less tolerant to disturbance.

Additionally, it is highly probable that the majority of the larval corixids collected at US1 (-210 km), US2 (-194 km), and US3 (-114 km) were *S. lineata*, but we were unable to confirm this as there is presently no way to identify most larval Corixidae. In contrast, Chironomidae and Docielinotidae were found in very high densities below the dam. These two taxa are highly tolerant to environmental disturbance, the latter being a mainly lentic species that helped define the lentic impact zone found downstream.

Despite our best efforts, studying large rivers presents a unique set of challenges. Their size can make it difficult to quantify the effects of impacts both longitudinally and laterally. In the present study, macroinvertebrates were sampled from one side of the river in the near-shore zone, but not from deeper parts of the channel. The deeper water species would not have been collected with our methods, and the use of other techniques and instruments would be required to sample them (e.g. Peterson grab sampler, Hess sampler). Others have found that disturbance-sensitive taxa are more likely to be found in deeper areas of the channel below hydropeaking facilities, whereas tolerant species tend to inhabit the edge habitat (Jones 2013b). This speaks to key questions regarding sampling design to understand impact. Our methods may be overly sensitive, showing a greater proportion of tolerant taxa than if the whole channel were sampled. This can be beneficial in assessing impacts, but may overestimate them at the ecosystem scale. Our coverage of spring-summer seasonality shows that high discharges in the spring can make it difficult for most taxa to inhabit near-shore areas downstream of the dam due to high flow and a lack of refugia, whereas later in the season low flows allow tolerant species to take over these habitats. However, effort associated with monthly sampling is large.

Finally, the impact of a dam varies spatially due to attenuation of hydropeaking from upstream to downstream, again, a key consideration in sampling design.

Determining river health often requires the use of several metrics to assign a score to the reach in question. As we discovered in our assessment, BI scores for reaches in the Saskatchewan River system appear to be negatively correlated with their EPT/C (Fig. 2.9). However, an important drawback of using EPT/C values is its use of specific taxa. This is illustrated in Fig. 2.9, where the negative correlation breaks down in August and September due to high densities of *Caenis* mayflies in those months. Their presence caused the EPT/C metric to indicate healthy river conditions, despite the BI metric suggesting otherwise (Fig. 2.9). An important consideration is that the sampling location in the spillway was the primary cause for this effect; this reach is less comparable to the other river locations and resembled a lentic ecosystem. Jackson et al. (2010) documented a similar phenomenon in the Mississippi River, where oligochaete worms and hydropsychid caddisfly larvae dominated many reaches of the river and affected a variety of techniques used to calculate river health. Our study, in conjunction with several others (Silveira et al. 2005, Borisko et al. 2007, Marchetti et al. 2011), suggests that the tolerance values for organic pollution compiled by Mandaville (2002) for use with the Modified Hilsenhoff BI may be applicable to environmental disturbances more generally in aquatic systems, and that metrics considering the entire benthic community may be preferred when determining ecosystem integrity.

The present study has illustrated that the hydropeaking E.B. Campbell Dam supports a downstream BMI assemblage that has high density and comparable species richness relative to that upstream, but this assemblage is shifted to one characterized by

tolerant, lentic-associated taxa. A key piece of environmental legislation in Canada, the Fisheries Act, currently assesses impacts of industrial operations on aquatic environments based their effects on fishes, namely through their habitat including provision of food (Fisheries Act, 1985). As a result, this legislation would view the increased abundances of BMIs downstream of the dam as a positive effect while ignoring the change in the assemblage and, more importantly, overall tolerance to disturbance. Despite the establishment of a minimum flow requirement in 2004, discharge and water depth continue to change on a daily basis with effects apparent as far as 53 km downstream (Fig. 2.2). Hydropeaking is an important means of matching power production to power requirements; however, we have found evidence that hydropeaking may contribute to the alteration of downstream biotic communities. Minimum flow requirements have many benefits, but more work is required to understand how to best manage dams to better mimic the natural flow regime, especially in systems dominated by multiple control structures that have competing demands for water. Integrated systems approaches that allow trade-offs among industrial and ecological uses will help mitigate current impacts of hydropower and hydropeaking and maintain its importance as part of the renewable energy portfolio.

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CHAPTER 3: DAILY SECONDARY PRODUCTION OF BENTHIC MACROINVERTEBRATES IN A GREAT PLAINS RIVER INFLUENCED BY A HYDROPEAKING DAM

3.1 Abstract

Secondary production in benthic macroinvertebrates (BMIs) is a common measure of aquatic ecosystem function. However, the use of such functional measures in aquatic biomonitoring is often overshadowed by the application of structural indicators, especially in northern environments. The effects of hydropeaking on secondary production and growth rates of BMIs are poorly understood. The E. B. Campbell Dam on the Saskatchewan River (Saskatchewan, Canada) is a hydropeaking facility that causes daily changes in flow and water level downstream. Here, we examine the effects of hydropeaking on daily secondary production and growth rates of several BMIs from the Saskatchewan River. We calculated mean daily secondary production and instantaneous growth rates for *Sigara lineata* (Hemiptera: Corixidae), *Hydropsyche* sp. (Trichoptera: Hydropsychidae), *Stenonema* sp. (Ephemeroptera: Heptageniidae), and Chironomidae (Diptera) from eight sites located upstream and downstream of E. B. Campbell Dam. Mean daily secondary production ranged from 0.059 to 23.402 mg m⁻² day⁻¹ and mean instantaneous growth rates ranged from 0.004 to 0.046. These estimates were within the range found in the literature. Upstream populations of *S. lineata*, *Stenonema* sp., and Chironomidae exhibited higher mean daily secondary production compared to downstream populations, whereas the opposite was true for *Hydropsyche* sp.. We suggest that suspended zooplankton supplied by the reservoir supports the highly productive population of *Hydropsyche* sp. downstream of the dam, and that hydropeaking may explain the absence of *S. lineata* at most downstream locations as well as the sudden appearance of *Hydropsyche* sp. at some downstream locations due to drift. Understanding

the effects of hydropeaking on BMI secondary production and growth rates may be crucial in establishing environmental flows that are implemented to meet ecological and societal needs.

3.2 Introduction

Benthic macroinvertebrates (BMIs) have long been recognized as indicators of environmental change due to their wide spectrum of tolerance to disturbance (Cairns and Pratt 1993, Bonada et al. 2006, Bailey et al. 2014). Consequently, BMIs are often used in determining the biotic integrity of aquatic ecosystems (Norris and Thoms 1999, Mandaville 2002). While structural indicators of biotic integrity are often used in assessing disturbance in rivers (Norris and Thoms 1999), functional metrics have received relatively little attention. Riverine ecosystem functions include processes such as biochemical oxygen demand, microbial respiration rates, gross primary production, nutrient uptake, and BMI secondary production (Buffagni and Comin 2000, Benke and Huryn 2006, Young et al. 2008). To effectively determine ecological integrity in rivers and other freshwater systems, methods that assess a combination of structure and function have been recommended (Dolbeth et al. 2012, Yates et al. 2014). The use of leaf litter breakdown rates in determining aquatic health is a common functional measure, largely due to the relative ease of its analysis (Gessner and Chauvet 2002, Young et al. 2008). This metric is often paired with BMI community structure (McKie and Malmqvist 2009). To date, few studies have used BMI secondary production as a functional indicator of environmental disturbance.

Secondary production is generally defined as the “formation of animal biomass over time” (Huryn and Wallace 2000). Several methods to estimate BMI secondary

production have been developed and are grouped into two main categories: cohort and size-based or non-cohort (Benke and Huryn 2006, Dolbeth et al. 2012). Cohort methods are used when distinct cohorts can be followed in a population through time, and methods include the increment summation, Allen curve, and removal summation approaches; in contrast, size-based methods can be utilized when BMI growth is not synchronous and/or cohorts cannot be distinctly recognized within a population and these include the size-frequency and instantaneous growth methods (Benke and Huryn 2006, Dolbeth et al. 2012). BMI secondary production has been described in the literature for decades, although the majority of studies take place in small streams (Berg and Hellenthal 1991, Céréghino et al. 1997, Barahona et al. 2005, Entekin et al. 2007, Bouchard and Ferrington 2009). In contrast, little work has been done to determine BMI secondary production in large river systems in temperate climates, probably due to the difficulty associated with sampling these large systems with distinct seasonality in temperature and flow. The application of secondary production in measuring human disturbance in aquatic systems is also rare.

Currently, a large proportion of the world's rivers have their hydrology altered by impoundments (Tharme 2003, Smokorowski et al. 2011). Hydroelectricity is one of the most popular renewable energy sources being used in the world today, and new projects continue to be planned for the future (Zarfl et al. 2015). Although hydroelectric dams provide a renewable, clean source of energy compared to fossil fuels, they can create environmental impacts. Effects of dams have been studied for decades and include changes in downstream biotic assemblages (Jones 2013, Chapter 2), as well as thermal effects (Lehmkuhl 1972, Olden and Naiman 2010, Phillips et al. 2015) and changes in

flow regimes (Poff et al. 2007). Dams can especially affect BMI community structure, and this is reflected in the wealth of research on their impacts (Lehmkuhl 1972, Gore 1977, Moog 1993, Armanini et al. 2014, Phillips et al. 2015, Kennedy et al. 2016).

While impoundments can affect downstream BMI assemblages through altered flow (Kennedy et al. 2016) and thermal regimes (Lehmkuhl 1972, Camargo and Voelz 1998, Phillips et al. 2015), there is a paucity of literature that describes how dams can affect BMI production (Céréghino et al. 1997). This is especially apparent for hydropeaking dams, and although the flow fluctuations caused by hydropeaking have been shown to change BMI community structure (Kennedy et al. 2014, Holzapfel et al. 2017, Chapter 2), how hydropeaking affects BMI secondary production is poorly understood.

This study investigated BMI secondary production in a large river in the Northern Great Plains of Saskatchewan, Canada. Our primary objectives were to compare the secondary production of a suite of common BMI taxa in this river to those found in the literature, and to examine whether the dynamic flow regime imposed by a hydropeaking dam is associated with differences between upstream and downstream BMI secondary production. We conducted this exploratory analysis to add to the currently sparse literature on large river BMI production, especially in the Northern Great Plains.

3.3 Methods

3.3.1 Study area

The Saskatchewan River system crosses three provinces in Canada, beginning in the Rocky Mountains of Alberta and discharging into Lake Winnipeg in Manitoba. The Saskatchewan portion is divided for most of its length into the North and South

Saskatchewan River, which merge east of the city of Prince Albert to form the mainstem Saskatchewan River (Fig. 2.1). Further east of the confluence is Tobin Lake, a reservoir created by the hydropeaking E.B. Campbell dam. Construction of the dam was completed in 1963, and its operation was largely based on electricity demand. Less water was released during times of low electricity consumption, resulting in drastic changes in discharge and water level downstream. This led to occasional fish stranding events, which prompted Fisheries and Oceans Canada to impose a mandatory daily minimum flow requirement of $75 \text{ m}^3\text{s}^{-1}$ on the operation in 2004 (Partners for the Saskatchewan River Basin 2009). Despite these minimum flow requirements, the hydropeaking nature of the dam means the river downstream of the E.B. Campbell dam continues to experience large changes in flow and water depth on a daily and seasonal basis (Fig. 2.3). In addition to these changes in river flow, when discharge from the dam falls below $1000 \text{ m}^3\text{s}^{-1}$, the reach in the spillway channel (hereafter referred to as SW) contracts to a series of small, interconnected pools with little to no flow. Unlike the rest of the downstream river, SW is not subjected to daily hydropeaking.

3.3.2 BMI sampling, processing

Sampling took place from May to September 2014 on a monthly basis. Three of the sites were located upstream of the E.B. Campbell dam and were treated as reference sites, whereas five sites were sampled downstream (Fig. 2.1). At each location, the area was divided into three separate sub-locations. Collection began at the sub-location furthest downstream, with subsequent samples taken in an upstream direction to avoid disturbing the benthos at the next sample point. Macroinvertebrates were collected using a $500\mu\text{m}$ D-frame kicknet to the deepest wadeable depth and/or until one minute had

elapsed. The total distance sampled was measured in meters from the shore to the final resting place using a tape measure. Large debris was removed from the sample after washing any macroinvertebrates off into the net. The entire sample was then preserved in 95% ethyl alcohol.

Macroinvertebrates were identified to the lowest practical designation using taxonomic keys specific to Saskatchewan (Webb 2002 for Ephemeroptera, Dossdall 1976 for Plecoptera, Smith 1984 for Trichoptera, Brooks and Kelton 1967 for Hemiptera). For all other taxa, the keys provided by Merritt et al. (2008) were used. Some taxa were difficult to resolve to genus-level, so these were identified to family only (e.g. Chironomidae, some larval Corixidae).

3.3.3 Species selection for assessing secondary production

In calculating daily production, taxa were selected if they were 1) present both upstream and downstream, and 2) present for at least 2 consecutive months. Only four taxa met these criteria: Chironomidae (Diptera), *Sigara lineata* (Hemiptera: Corixidae), *Hydropsyche sp.* (Trichoptera: Hydropsychidae), and *Stenonema sp.* (Ephemeroptera: Heptageniidae). For each month, histograms were used to identify and separate groups of similarly sized individuals into cohorts (Fig. 3.1).

3.3.4 Calculating daily secondary production and growth rates

Macroinvertebrate secondary production was determined using equations for the instantaneous growth rate method outlined by Benke and Huryn (2006). First, the growth rate (g) was calculated using:

$$g = \frac{\ln(W_{t+\Delta t}/W_t)}{\Delta t} \quad (\text{Eq. 2})$$

where W_t is the mean mass of an individual at time t , $W_{t+\Delta t}$ is the mean mass of an individual at time $t + \Delta t$, with Δt representing the length of the time interval. The mass of an individual was estimated by first measuring its body length using an ocular micrometer, then converting that length to a mass using established length-mass equations (Benke et al. 1999). Once the instantaneous growth rate was determined for a taxon, daily secondary production (P_d) was calculated using:

$$P_d = g\bar{B} \quad (\text{Eq. 3})$$

where \bar{B} is the mean population biomass (g/m^2) calculated over two consecutive dates. For each taxon, length frequency histograms were used to determine how many cohorts were present (Fig. 3.1). Daily production and instantaneous growth rates were calculated for each cohort from month to month. The sum of the daily production for all cohorts resulted in a total daily production value for each taxon. For our comparisons of upstream vs. downstream, we first calculated total production for each location by summing all values from May – September. We then determined the mean growth rate for each location. Finally, we calculated the mean total production and growth rate for the upstream (US1, US2, and US3) and then downstream (SW, DS1, DS2, DS3, and DS4) locations where each taxon was present.

3.3.5 *Water sampling and analysis*

Secondary production can be affected by temperature, nutrient concentrations, and algal abundance (Benke and Huryn 2006); hence, these parameters were monitored using sensors, and discrete samples. Water temperature was measured at regular intervals using deployed HOBO probes at all upstream and 4 of 5 downstream locations. Nutrient and algal data were obtained using methods described in the previous chapter (Table 2.1).

3.3.6 Literature Review

We performed an extensive review of the available literature on BMI secondary production in running waters. To do this, we utilized the Web of Knowledge database and searched with the following terms: “invertebrates” or “insects” and “freshwater” or “stream” or “river” or “lake” or “pond” or “wetland” and “growth rate” or “P/B” or “production/biomass” or “secondary production” or “biomass turnover rate.” We then searched within the results for those taxa in the same genus (or family for Chironomidae) as the four taxa for which we calculated secondary production to draw comparisons. Studies were considered relevant to our research if they were carried out in a running body of water and secondary production was estimated, especially if the effect of a dam was examined. The review returned a total of 721 studies with the initial search terms (searches completed May 6-8, 2017). Searching within the results for “hydropsyche” yielded 13 papers, nine of which were relevant to our study. Performing the same search with “sigara” or “corixidae” yielded just two papers, only one of which was relevant and was used to compare to our calculated production values for *S. lineata*. Following the same procedure with “stenonema” retrieved one relevant study, and searching for “chironomidae” returned 63 studies, of which nine were relevant. Annual production values and growth rates from these studies were converted to mean daily production values and instantaneous growth rates, respectively; these are compiled in Table 3.1.

3.4 Results

3.4.1 Daily secondary production and growth rates for selected species

In total, 67,506 BMIs from 237 different taxa were collected in our study. From these, we identified 15,830 Chironomidae, 3,415 *S. lineata*, 5,304 *Hydropsyche sp.*, and

785 *Stenonema sp.* Density ranged from 4 to 1,371 individuals m⁻² for Chironomidae, 0 to 345 m⁻² for *S. lineata*, 0 to 1036 m⁻² for *Hydropsyche sp.*, and 0 to 223 m⁻² for *Stenonema sp.* Production values for *Hydropsyche sp.*, and *Stenonema sp.* were calculated assuming the population at each location was a single cohort that was sampled from May – September. For Chironomidae and *S. lineata*, several cohorts were identified throughout the season (Fig. 3.1). Secondary production values were determined for each cohort from one month to the next, and mean daily secondary production was based on the summation of these values.

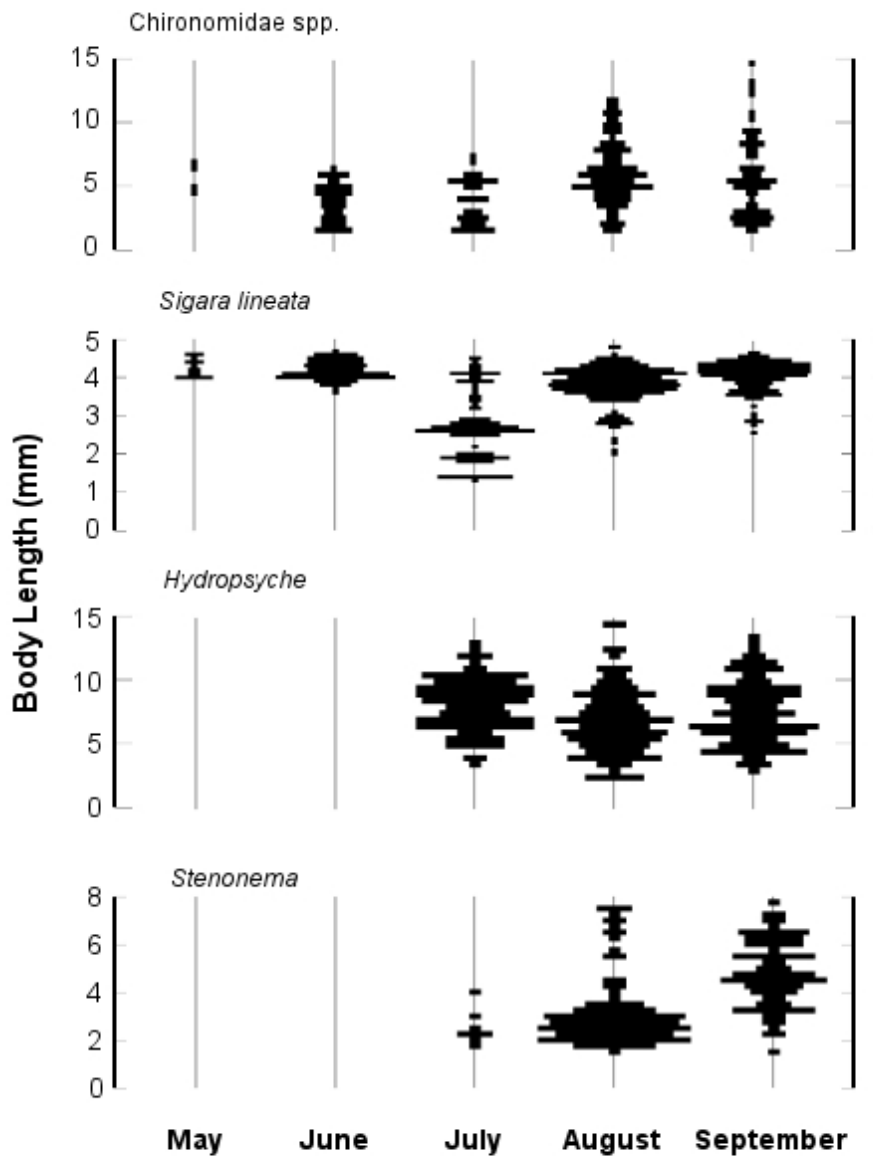


Fig. 3.1: Example size frequency histograms for the taxa in this study from specific sites for the months of May to September, 2014. Lengths for Chironomidae were taken from US2, *S. lineata* from US1, *Hydropsyche sp.* from DS1, and *Stenonema sp.* from US1. Bar width corresponds to the percentage of specimens with that body length.

Daily secondary production and instantaneous growth rates for Chironomidae, *S. lineata*, *Hydropsyche sp.*, and *Stenonema sp.* ranged from 0.059 to 23.402 mg m⁻² day⁻¹ and 0.004 to 0.046, respectively, and are summarized in Table 3.1. For Chironomidae, *S.*

lineata, and *Stenonema sp.*, mean daily production appeared to be higher upstream of the dam relative to downstream. However, daily production in *Hydropsyche sp.* shows the opposite trend, with higher production at downstream reaches. Growth rates for *S. lineata* were almost identical for populations above and below the dam, whereas higher growth rates were observed for Chironomidae, *Hydropsyche sp.*, and *Stenonema sp.* downstream (Table 3.1).

Table 3.1

MEAN DAILY SECONDARY PRODUCTION AND MEAN INSTANTANEOUS GROWTH RATES
FOR SELECTED TAXA

| Location | Species | Mean Daily Production, P ($\text{mg m}^{-2} \text{ day}^{-1}$) | Mean Instantaneous Growth Rate, g (P/B) | Source |
|--------------------------------------|--|---|--|---------------------------|
| Saskatchewan River, upstream | <i>Sigara lineata</i> | 0.824 ± 0.034 | 0.020 ± 0.003 | This study |
| Saskatchewan River, downstream | | 0.059 ± 0.006 | 0.021 ± 0.003 | |
| Rambla Salada Stream, Spain | <i>Sigara selecta</i> | 3.507 | 0.038 | Barahora et al. 2005 |
| Saskatchewan River, upstream | <i>Hydropsyche</i> <i>sp.</i> | 0.083 ± 0.033 | 0.004 ± 0.002 | This study |
| Saskatchewan River, downstream | | 23.402 ± 1.430 | 0.024 ± 0.004 | |
| Kintore Creek, Canada | <i>Hydropsyche</i> <i>slossonae</i> | 6.712 – 32.932 | 0.009 – 0.012 | Sallenave and Day 1991 |
| | <i>H. betteni</i> | 7.397 – 11.479 | 0.009 – 0.010 | |
| | <i>H. bronta</i> | 0.137 – 15.781 | 0.020 | |
| | <i>H. sparna</i> | 0.003 – 2.329 | 0.0003 – 0.018 | |

| | | | | |
|--------------------------------|---|--|--|------------------------|
| Ogeechee River, USA | <i>Hydropsyche rossi</i> <i>H. incommode</i> | 14.082 – 108.452 0.107 – 0.238 | 0.038 – 0.043 0.035 | Benke and Wallace 1997 |
| Saskatchewan River, upstream | <i>Stenonema sp.</i> | 0.996 ± 0.075 | 0.026 ± 0.003 | This study |
| Saskatchewan River, downstream | | 0.601 ± 0.024 | 0.046 ± 0.002 | |
| Ogeechee River, USA | <i>Stenonema exiguum</i> <i>S. integrum</i> <i>S. modestum</i> <i>Stenonema spp.</i> | 0.453 – 1.392 0.367 – 1.066 1.378 – 4.165 0.109 – 0.306 | 0.048 0.047 – 0.049 0.047 – 0.048 0.042 – 0.043 | Benke and Jacobi 1994 |
| Saskatchewan River, upstream | Chironomidae | 0.975 ± 0.095 | 0.021 ± 0.004 | This study |
| Saskatchewan River, downstream | | 0.706 ± 0.188 | 0.034 ± 0.006 | |
| Ogeechee River, USA | Chironomidae | 62.071 – 224.266 | 0.592 – 0.644 | Benke 1998 |
| Tundra pond, Alaska | <i>Chironomus spp.</i> | 0.011 | 0.010 | Butler 1982 |

3.4.2 Literature review findings

Production and growth rates found in this study were well within the range found in the literature. Mean daily production and growth rates for Chironomidae, calculated from annual values, ranged from 0.011 mg m⁻² year⁻¹ and 0.010 in an Alaskan tundra pool (Butler 1982) to 224 mg m⁻² year⁻¹ and 0.644 in the Ogeechee River in Georgia, USA (Benke 1998). Sallenave and Day (1991) reported mean daily production as low as 0.003 mg m⁻² day⁻¹ and a mean growth rate of 0.0003 for *Hydropsyche sparna* in Kintore Creek (Ontario, Canada), whereas Benke and Wallace (1997) found values as high as 108 mg m⁻² day⁻¹ for production and a mean instantaneous growth rate of 0.043 for *H. rossi* in the Ogeechee River (Georgia, USA). Benke and Jacobi (1994) observed daily production from 0.109 to 4.16 mg m⁻² day⁻¹ and growth rates from 0.042 to 0.049 in three species of *Stenonema* in the Ogeechee River (Georgia, USA). Finally, Barahona et al. (2005) calculated a mean daily production value of 3.507 mg m⁻² day⁻¹ and mean instantaneous growth rate of 0.038 for *Sigara selecta* in the Rambla Salada Stream in Spain.

3.5 Discussion

Large river systems are vital for society as they provide a variety of environmental services including irrigation, hydroelectric power, recreation, and cooling for industrial operations. Functional measures such as secondary production and leaf litter breakdown rates have been viewed as superior to structural metrics in assessing ecosystem health in response to the above stressors (Young et al. 2008). Yet BMI secondary production studies are often conducted in small, wadeable streams rather than large rivers (Berg and Hellenthal 1991, Barahona et al. 2005, Entekin et al. 2007). Our study estimated the secondary production of four taxa found above and below a

hydropeaking dam in a large Great Plains river for much of the ice-free season. Key findings included high production values for *Hydropsyche* sp. below the dam, apparently lower production for *S. lineata* below the dam relative to upstream locations, and differing growth rates for *Stenonema* sp. (higher), Chironomidae (higher), and *S. lineata* (lower) at downstream reaches.

In our literature review, we found that studies on secondary production in corixids are rare, regardless of the type of water body. Identification keys for larval Corixidae are scarce which may explain why corixids are not commonly used in estimating secondary production. The larval stages of the corixid used in our study, *Sigara lineata*, have not been described. However, we are confident in the characteristics used to identify larval corixids as *S. lineata*, including patterns on the wing pads and dorsal abdominal surface (Konopko 2014). Our estimation of daily secondary production and the instantaneous growth rate of *S. lineata* above ($0.932 \pm 0.060 \text{ mg m}^{-2} \text{ day}^{-1}$, 0.025) and below ($0.059 \pm 0.005 \text{ mg m}^{-2} \text{ day}^{-1}$, 0.021) E. B. Campbell dam were for the months of July, August, and September 2014. As mentioned previously, literature estimating secondary production in Corixidae is rare; the only study we found in our review was that of Barahona et al. (2005) in which annual secondary production and growth rates were calculated for *S. selecta*. Understandably, the production and growth rates calculated by Barahona et al. (2005) for *S. selecta* collected from the subtropical Rambla Salada stream in Spain were higher than what we found for *S. lineata* in a northern temperate river (Table 3.1). The climate in the region where *S. selecta* was studied (mean annual temperature 18°C) allowed growth throughout the year; in contrast, the growing season in Saskatchewan is much shorter, typically lasting from April to October. Additionally, Barahona et al.

(2005) described *S. selecta* as being multivoltine with four cohorts throughout the year, whereas we observed a single generation of *S. lineata* in our study. While water temperature is known to affect growth rate and total production in aquatic insects, river temperatures at locations above and below the dam were similar (Fig. 2.6), which suggests that additional factors limit the downstream population of *S. lineata*.

Downstream growth rates of *S. lineata* were similar to those found upstream, suggesting suitable habitat conditions below the dam. This could imply dispersal limitation in downstream *S. lineata*.

The high production observed for *Hydropsyche sp.* at downstream locations in our study might be due to their proximity to the dam. Macfarlane and Waters (1982) found that filter feeders such as hydropsychid caddisflies are prevalent below reservoirs as they feed on the large amount of zooplankton released to the downstream environment. While downstream secondary production in *Hydropsyche sp.* was certainly impressive in our study ($23.402 \pm 1.430 \text{ mg m}^{-2} \text{ day}^{-1}$), higher production in the genus *Hydropsyche* has been observed elsewhere (Benke and Wallace 1997). Benke and Wallace (1997) calculated the secondary production of BMIs in the Ogeechee River and found values ranging from 5,140 to 39,585 $\text{mg m}^{-2} \text{ y}^{-1}$ for *Hydropsyche rossi*, which translates to a mean daily production range of 14.08 to 108 $\text{mg m}^{-2} \text{ day}^{-1}$. It is likely that the downstream production observed for *Hydropsyche sp.* in our study was due to the combination of large densities and the input of zooplankton from the Tobin Lake reservoir, as the instantaneous growth rate was similar to that calculated for other species (Table 3.1).

Daily production and growth rates for *Stenonema sp.* were well within the range found in the literature (Table 3.1). Interestingly, *Stenonema sp.* exhibited higher production at upstream locations even though our calculations suggested that downstream populations have a much higher growth rate (Table 3.1). The likely cause for this observation is the higher density of individuals found at locations above the dam relative to downstream reaches. The grazing behaviour of *Stenonema sp.* may explain why populations below the dam may have an elevated growth rate, as large mats of benthic algae were observed there, indicated by high benthic chlorophyll-a concentrations (Table 2.1). Kennedy et al. (2016) highlighted that the loss of EPT at sites downstream of hydropeaking facilities was due to females depositing their eggs in shallow regions. The rapid fluctuation of water due to hydropeaking exposed the eggs to the air, causing them to desiccate. McCafferty and Huff (1974) observed adult female *Stenonema spp.* ovipositing eggs on stream surfaces, but did not indicate whether this occurred across the entire surface or in an isolated region (e.g. shallow riparian areas or deeper parts of the channel). We suggest that hydropeaking may limit the downstream population of *Stenonema spp.* due to repeated daily drying of river substrate, which might cause their eggs to desiccate.

Changes in flow regime have been shown to influence BMI secondary production by altering community structure (Hauer and Benke 1991, Scholl et al. 2016). In our study, changes to the river resulting from hydropeaking were most apparent at DS2 and DS3, where the water level changed rapidly on a daily basis from July to September. Fewer BMIs were collected from DS2 and DS3 during this time; as a result, we were unable to calculate secondary production and growth rates for most BMIs at these

locations, with Chironomidae being the exception. Chironomids were highly abundant at DS2 and DS3 throughout the sampling period. Mean daily chironomid production was $0.917 \pm 0.513 \text{ mg m}^{-2} \text{ day}^{-1}$ at DS2 and $0.655 \pm 0.026 \text{ mg m}^{-2} \text{ day}^{-1}$ at DS3. These values comprised 25.9% and 18.6%, respectively, of total downstream chironomid production for the sampling period. Mean instantaneous growth rates for Chironomidae were 0.016 ± 0.003 at DS2 and 0.022 ± 0.001 at DS3. As chironomids are tolerant to environmental disturbance, it is unsurprising that they were productive at these locations.

Conversely, *Hydropsyche spp.* is known to be less tolerant to degraded conditions, and yet the population found at DS2 was quite productive (daily production $1.255 \pm 0.221 \text{ mg m}^{-2} \text{ day}^{-1}$, growth rate 0.016 ± 0.006). *Hydropsyche sp.* densities were minimal from May – July at DS2; however, a sudden increase was observed in August and September. Additionally, benthic chlorophyll concentrations were only elevated at DS2 in September and were similar to other sites at DS3 (Table 2.1). The substrate observed at these locations was mostly coarse sand with no visible macrophyte growth. Considering these factors, we suspect that the sudden surge of *Hydropsyche sp.* observed at DS2 was a product of drift triggered by the changes imposed by hydropeaking. BMI drift caused by hydropeaking operations has been documented by several researchers, including Tonkin et al. (2009), Kennedy et al. (2014), Schülting et al. (2016), and Holzapfel et al. (2017). In particular, Tonkin et al. (2009) observed significant increases in both invertebrate drift and periphyton biomass in the Tongariro River downstream of a hydropeaking facility.

Most studies of secondary production are performed over the course of a year or several years, resulting in an annual production value and growth rate (Butler 1982, Berg and Hellenthal 1991, Benke and Jacobi 1994, Benke 1998, Benke and Huryn 2006,

Scholl et al. 2016). In regions of the world where streams and rivers remain ice-free and flowing year-round, BMIs are often readily accessible and can be sampled any day of the year. Temperate regions differ in that streams and rivers are covered in ice for a period of the year, presenting additional difficulties in collecting BMIs. In Canada, the Saskatchewan River freezes as early as November, remaining frozen until April or May; smaller streams often freeze earlier and thaw later than large rivers. Winter sampling in temperate regions therefore requires drilling through the ice to access the water and sediment to collect BMIs, thereby increasing time and costs, with potential safety risks. Researchers studying temperate systems during the winter months have utilized alternative methods to observe production and growth rates of BMIs. Bouchard and Ferrington (2009) examined the production and growth rates of the chironomid *Diamesa mendotae* during the winter by collecting eggs from adults and rearing larvae in growth chambers placed in Minnesota streams for a full year. However, the streams studied by Bouchard and Ferrington (2009) do not typically freeze over in winter, and therefore puncturing ice cover to sample BMIs was not required. While our study encompassed the majority of the open water growing season in Saskatchewan, time and budget constraints prevented winter sampling of BMIs. However, as production and growth rates have been shown to be highly temperature-dependent (Hauer and Benke 1991, Reynolds and Benke 2005, Benke and Huryn 2006), most BMIs in the Saskatchewan River are likely to grow fastest from July to October when temperatures are the highest. It is likely that there are exceptions to this, such as winter stoneflies that prefer cooler water temperatures for growth (Dosedall 1976).

Our study has described the secondary production and growth rates of four taxa in a large Great Plains river and has illustrated that the hydropeaking E.B. Campbell Dam may support a highly productive filter-feeding community at locations immediately downstream. This suggests that the transport of zooplankton from the reservoir may contribute to the change in benthic community structure at downstream locations, which was illustrated in the previous chapter. Pairing our estimates of secondary production and growth rates with our evaluation of the BMI community lends insights to how hydropeaking may affect different taxa. Hydropeaking-tolerant taxa such as Chironomidae are able to proliferate without competition while intolerant taxa may drift further downstream to escape the disturbance. Other taxa such as *Hydropsyche spp.* benefit from the increased flows caused by hydropeaking as well as reservoir-derived plankton. We recommend exploring these patterns further by monitoring BMI production below hydropeaking dams for longer periods of time to track long-term changes to the BMI community.

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CHAPTER 4: SUMMARY AND CONCLUSIONS

4.1 Key Findings

Our analysis of the effects of E.B. Campbell Dam on the Saskatchewan River found altered BMI assemblages below the dam, resulting in degraded river health according to the BI and EPT/C metrics we applied. At DS2 and DS3, where the effects of hydropeaking were most severe in August and September, the BMI community was reduced to only a few disturbance-tolerant taxa, suggesting that it may contribute to the alteration of the BMI community. Of those taxa found below the dam, *Hydropsyche sp.* was highly productive, suggesting greater availability of zooplankton as food for this

taxon. Conversely, *S. lineata* was largely absent at downstream locations except for those observed at DS4. This population had drastically reduced mean daily secondary production compared to the upstream population. Considering that *S. lineata* was present upstream and only at locations further than 50 km downstream from the dam, it is unlikely that this would happen under natural hydrologic conditions.

4.2 Practical Relevance

4.2.1 Relevance to river monitoring

The findings of this study suggest that monitoring BMI communities below E. B. Campbell Dam should be carried out frequently, perhaps on an annual or biannual basis. This information could be used to track changes over multiple years and better understand the impact of differing levels of discharge released by the facility. An adaptive management approach in which the results of changes to the operation of the dam are monitored would be beneficial. Monitoring data could then be used to inform future changes in flow release schedules.

4.2.2 Relevance to stakeholders

Among those that could benefit from this research are the operators of the facility (SaskPower), researchers studying the aquatic biota in the region, and residents who are affected by changes to the operation of the dam. While the dam is essential for providing a clean form of electricity for many in the region, the impact of its operation cannot be ignored. Changes to the BMI community could propagate to the fish community, which could then affect those that rely on fish for sustenance. Striking a balance between societal and environmental needs will be a difficult process, but a better mimicry of the

natural flow regime of the Saskatchewan River could potentially improve the downstream environment.

4.3 Future Research and Recommendations

I recommend that future researchers monitor the BMI communities above and below the dam and observe any changes over multiple years. This could include winter sampling to assess how hydropeaking might affect BMI communities under ice cover and the calculation of annual production values for several taxa. The development of a taxonomic key to identify immature Corixidae in Saskatchewan would be highly beneficial for future river monitoring research. While our furthest sampling location was 52 km downstream of the dam, the BMI community was only beginning to resemble upstream locations. Therefore, I suggest that future research include locations that extend further downstream into the Saskatchewan River Delta. Sampling additional upstream locations would also be beneficial to develop a true Reference Condition Approach (Bailey et al. 2014).

The present study has illustrated how seasonality plays an important role in the determination of river health and the possible effects of hydropeaking on downstream biota. Therefore, I suggest that future studies incorporate seasonality into their sampling designs when possible. This can be achieved through sampling year-round or during the entire ice-free season on a regular basis. Unlike most sampling designs that focus on the late summer months, incorporating seasonality will allow researchers to examine how BMI communities change throughout the year as well as how those changes affect the communities' response to hydropeaking.

To better separate the effects of hydropeaking from those imposed by the dam as a structure, I recommend that future researchers arrange to sample the river under normal hydropeaking operations and again during a period where hydropeaking is absent. Achieving this would require participation of the dam operators to suspend hydropeaking activities for a short period of time. How taxa respond to the daily fluctuations in discharge and water depth could be determined by sampling BMIs during the daily peak and again during the daily low.

Successful mitigation of hydropeaking impacts will likely involve a combination of structural and operational alterations to existing dams (Bruder et al. 2016, Hauer et al. 2017, Premstaller et al. 2017). Recent studies have suggested several options, including the incorporation of basins and caverns to slow the ramping effect caused by hydropeaking (Tonolla et al. 2017), increasing minimum flow requirements, and creating a low-volume 'pre-surge' before the daily peak (Bruder et al. 2016). Identifying the optimal mitigation measures will require participation from key stakeholders, including operational staff, scientists, and members of the public. Proposed mitigation efforts should reduce negative impacts to downstream biota while considering socioeconomic implications, and monitoring programs should be established to compare the outcome of the chosen mitigation strategies to initial expectations.

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