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Invertebrate drift in the upper Columbia River with implications for young-of-the-year white sturgeon

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INVERTEBRATE DRIFT IN THE UPPER COLUMBIA RIVER WITH

IMPLICATIONS FOR YOUNG-OF-THE-YEAR WHITE STURGEON

A Thesis

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Master of Science in Biology

By

Ryan W. Reihart

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Master's Thesis

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Abstract

White sturgeon (*Acipenser transmontanus*) in the upper Columbia River have been in decline for over 60 years and are listed as critically endangered by the International Union for Conservation of Nature. Annual spawning produces dispersing larvae; however, recruitment into the juvenile age class has not occurred since the early 1970s. One hypothesis to explain this failure in recruitment is that larvae may not find adequate prey. Little is known about diet and prey availability for white sturgeon larvae, especially in Lake Roosevelt, a reservoir of the Columbia River. The Spokane Tribe of Indians collected littoral and mid-channel benthic drift samples annually at 9 locations in Lake Roosevelt from 2007 to 2010. Zooplankton dominated drift samples in both littoral and thalweg zones, while excluding zooplankton, *Hydra* dominated the drift. Overall, the mean density and biomass of invertebrates within the flow was highest in littoral sites and highest in years with the greatest flow. Littoral and thalweg sites had no impact on drifting benthic macroinvertebrate diversity, richness, or evenness. To determine the feeding ecology, 590 larvae were collected in July 2015. A total of 14 prey items were found in 9 stomachs, mainly Dipteran larvae and Temoridae. Our results suggest that prey scarcity could be limiting white sturgeon recruitment in Lake Roosevelt.

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Introduction

The white sturgeon (*Acipenser transmontanus*) is the largest and longest-lived freshwater fish in North America (McCabe et al. 1993; Hildebrand et al. 1999; Irvine et al. 2007; Tompsett et al. 2014). Sturgeon live in large, fast-moving, and diverse river systems (Jager et al. 2001, 2006; Hatten & Parsely 2009) found between Monterey, California and the Cook Inlet, Alaska (Wydoski & Whitney 1979; Beamesderfer & Far 1997). The Columbia River, which is located in the middle of their habitat range, supports the largest population of white sturgeon in the world (McCabe et al. 1997).

This population is currently segmented due to the construction of hydroelectric dams, which began in 1933 and has landlocked populations above the Bonneville Dam (Haynes et al. 1978; McCabe et al. 1993; North et al. 1993; Dillon & Grunder 2008). The population of white sturgeon that is unimpounded, below the Bonneville Dam, is one of the only Columbia River populations that is currently showing annual recruitment (Devore et al. 1995). On the other hand, populations above the Bonneville Dam that are landlocked are now threatened or endangered due to poor recruitment (Muir et al. 2000; Paragamian 2012).

Since the early 1970s, populations of white sturgeon in Lake Roosevelt, upstream of the Bonneville Dam, have been showing signs of recruitment failure (Howell & McLellan 2011, 2013; Irvine et al. 2007). There have been many hypothesized causes of white sturgeon recruitment failure, including, angling and commercial fishing (Hildebrand et al. 1999), lack of suitable habitat for spawning (North et al. 1993; Jager at al. 2001, 2006; Paragamian et al. 2002; McAdam et al. 2005; Kock et al. 2006; Paragamian & Wakkinen 2011), poor water quality (Feist et al. 2005; Besser et al. 2008;

Tompsett et el. 2014), predation (Gadomski & Parsley 2005; Howell & McLellan 2011), and prey availability (McCabe et al. 1993; Muir et al. 2000); however, the cause for recruitment failure is still unknown. There are two major causes of larval fish mortality, which are starvation and predation (Houde 1987; Muir et al. 2000). These two causes are likely interrelated because when larval sturgeon cannot find viable food, these fish will swim up in the water column to look for food, thus exposing them to a greater risk of predation.

Although little is known about the feeding ecology of the white sturgeon larvae in Lake Roosevelt, extensive research has been conducted throughout reservoirs in the lower Columbia River. Juvenile and larval white sturgeon below the Bonneville Dam primarily fed on *Corophium* spp., *Ramellogammarus* spp., Cyclopoida, Diptera larvae and pupae, *Neomysis mercedis*, and Oligochaeta (McCabe et al. 1993; Muir et al. 1986; 2000). *Corophium* spp. was found to be the most important prey item for white sturgeon larvae and juveniles below the Bonneville Dam and in the Dalles Reservoir (McCabe et al. 1993; Muir et al. 2000); however, this amphipod is not found in the Upper Columbia River, including Lake Roosevelt.

One study has been conducted in Lake Roosevelt to determine the diet of white sturgeon larvae. Of the 586 sturgeon larvae stomachs examined, prey items were only found in 27 of the larvae, which measured between 19.1 to 21.5 mm total length (TL). Diptera larvae were found in 62.9% of the sturgeon larvae that contained prey, indicating Diptera larvae are the most important prey items for white sturgeon larvae in Lake Roosevelt (Howell & McLellan 2013). Copepoda, Diptera pupae, and mites were also found but not as frequently as Diptera larvae (Howell & McLellan 2013). Although these

results seem promising for showing what white sturgeon larvae are consuming in Lake Roosevelt, the larval samples may have been collected before most larvae began exogenous feeding. Muir et al. (2000) found that the smallest larvae collected below Bonneville Dam that contained a prey item was 21 mm in length, whereas the largest larvae collected in Lake Roosevelt was only 22.2 mm in length (Howell & McLellan 2013). This would explain why 559 of the larval stomachs were empty or still contained part of their yolk sac. Capture of larger larvae in Lake Roosevelt could better represent their diet; however, when comparing diets of white sturgeon larvae in both Lake Roosevelt and below the Bonneville Dam, sturgeon larvae prey upon items commonly found within the drift, especially at the onset of exogenous feeding.

Invertebrate drift in streams, rivers, and reservoirs is regulated by many factors. The most influential factors are diel periodicity (drift increases during the night), depth stratum (highest near the bottom), and location in the watershed (increasing near shore; decreasing downriver; Junk et al. 1989; Winnell & Jude 1991). Furthermore, anthropogenic factors such as dams, land use, and habitat degradation are known to influence macroinvertebrate community structure and abundance (Zhang et al. 2010; Ellis & Jones 2013). Zooplankton dominate drift samples in large rivers and reservoirs; however, aquatic insects from the orders Ephemeroptera, Plecoptera, and Diptera are also commonly found with chironomids being the most abundant of the aquatic macroinvertebrates (Winnell & Jude 1991; Johansen et al. 2010). Drift densities of amphipods and chironomids were positively related to the benthic density (Kennedy et al. 2014) and are known to be preferred prey for white sturgeon larvae (McCabe et al. 1993; Muir et al. 1986; 2000).

Flow in large rivers greatly dictates community structure, density, and production of benthic macroinvertebrates and zooplankton (Bothar 1987; Brittain & Eikeland 1988; Saunders and Lewis 1988; De Ruyter Van Steveninick et al. 1990; Pace et al. 1992; Kennedy et al. 2014, 2016). Small zooplankton dominate drift in turbulent flow (Hynes 1970; Saunders and Lewis 1988; De Ruyter Van Steveninick et al. 1990; Pace et al. 1992), whereas larger zooplankton are found in greater abundances as flow decreases and the water becomes less turbulent (Saunders and Lewis 1988; De Ruyter Van Steveninick et al. 1990). Drastic increases in flow have been shown to cause large increases of drifting macroinvertebrates (Anderson and Lehmkuhl 1968; Poff and Ward 1991; Kennedy et al. 2014), while zooplankton communities are often transported downstream with no upstream net production (Saunders and Lewis 1988; Pace et al. 1992). In addition, macroinvertebrates will enter the drift with sudden decreases in flow to avoid stranding (Perry and Perry 1986; Poff and Ward 1991). These changes in flow are seen in many reservoirs and below dams, and could be negatively impacting the availability of suitable prey for drift feeding fish (Kennedy et al. 2014).

Recruitment and year-class strength are influenced by the availability and size of preferred prey as larval fish begin exogenous feeding (Muir et al. 2000). White sturgeon have a gape-limited mouth, therefore it is vital for preferred prey to be small enough for sturgeon larvae to be able to eat and mature to the juvenile stage (Hildebrand et al. 1999). Muir et al. (2000) found that young-of-the-year white sturgeon found below the Bonneville Dam consumed prey ranging from 1.0 to 5.0 mm in length. Prey availability has received little attention in reservoirs showing recruitment failure, but has been well researched in the lower Columbia River, where there is constant recruitment (McCabe et

al. 1993, 1997; Muir et al. 2000). Mean benthic macroinvertebrate (BMI) density below the Bonneville Dam ranged from 955 to 49,701 organism/m^2 at river mile 211 and 153, respectively, where the substrate has been estimated to be 99% sand (McCabe et al. 1993). Furthermore, amphipods were abundant and preferred prey for the white sturgeon larvae indicating the importance of amphipods for the white sturgeon larvae below the Bonneville Dam (Sprague et al. 1992; McCabe et al. 1997).

Because the prey availability and feeding ecology of the white sturgeon larvae is currently unknown in the Lake Roosevelt reservoir of the Columbia River, I investigated if prey availability is a significant factor in white sturgeon recruitment failure. I had three objectives for this study, which were to (1) describe the benthic macroinvertebrate (BMI) community structure (2) describe the feeding ecology of white sturgeon larvae and (3) determine how the BMI community structure and abundance is impacting white sturgeon recruitment. I hypothesized that: (1) mid-channel (thalweg) BMI samples will support a lower density and biomass of BMIs compared to littoral samples, (2) BMI density and biomass will decrease moving downstream, (3) BMI density, biomass, richness, diversity, and evenness will increase in years showing higher flow when compared to years showing low flow values, and (4) white sturgeon larvae will consume zooplankton (pelagic BMI) in higher frequencies when compared to littoral BMIs.

Methods

Study Site

Lake Roosevelt, a 243 km reservoir of the upper Columbia River, is found between the Grand Coulee Dam and the Canadian border (Fig. 1). The Grand Coulee

Dam began construction in 1933, and the resulting reservoir reached full pool in 1941. The Grand Coulee Dam is the largest hydropower producer in the United States, and is part of the Columbia Basin Project. The total active water storage of Lake Roosevelt is just less than 5.2 million acre-feet, while the total capacity is just less than 9.4 million-acre feet (USBR 2014). This dam stores and carries water for irrigation, electricity production, and flood control. To accomplish these goals, Lake Roosevelt has two main drawdown events per year (February – May and August – September), which alter the water depth, total water, elevation, and flow regime of Lake Roosevelt.

The study area includes the upper portion of Lake Roosevelt extending from river km (Rkm) 1133.6 (RM 704.39) near Marcus Flats, WA to Rkm 1168.6 (RM 726.14) near Little Dalles, WA. Mean and maximum depths in the study area are 12.6 and 63.5 m, respectively (Scofield 2007). Substrates in the study area are believed to be mainly rocky including gravels to boulders and bedrock. Sediment core sampling indicated that some finer/sandy sediments can be found in shallower water along shelves and bars running parallel to shore, while the thalweg appears to usually be swept of finer sediments (EPA 2006). Mean flow values collected from the Canadian border in Lake Roosevelt from April to June varied between 2007 – 2011, but were highest in 2007 and lowest in 2010 (Fig 2.; DART 2016).

Sampling locations within the 39 km section between Marcus Flats and the Little Dalles were randomly chosen with the use of a random number generator, Columbia River mile, and the aid of Geographical Information Systems (GIS). Each location was further stratified by depth into a deep (thalweg; >26 m) and shallow (littoral; ~10 m) site. A total of 9 locations were sampled annually, producing a total of 18 sampling sites. The

depths were chosen based on past BMI sampling efforts (Beckman et al. 1985; Griffith et al. 1995; Voeller 1996) and water elevations observed since 2001. The purpose of sampling two depth zones was to look for differences between the dewatered zone (littoral) and unexposed zone (deep), due to the annual drawdown events. Shallow sites alternated systematically between left and right banks of the river channel. If the systematically chosen site could not be sampled for logistic reasons, alternate sites were sampled by moving 0.5 km downstream from the proposed site. A GPS device was used in the field to determine exact locations.

Benthic Invertebrate Sampling

Two benthic drift nets (15x30 cm; 0.25 mm mesh) were deployed for two sets (dawn and dusk) on the bottom at each site for 15 minutes from 2007-2010. The drift nets were rectangular and attached to an anchor/buoy system. Depth, time, flowmeter counter, and GPS coordinates were noted at time of deployment. Samples were collected in mid-July to coincide with the dispersal of white sturgeon larvae that were beginning exogenous feeding. The drift nets were operated and managed by the Spokane Tribe of Indians (STOI). Drift in large rivers is often higher near the bottom (Matter & Hopwood 1980; Klemm et al. 1990) and in greater abundance and BMI size during the time immediately following dusk (Brittain & Eikeland 1988; Slack et al. 1991). Cross sectional water velocities in Lake Roosevelt from July and August (2002 to 2006) were estimated to range from 0.2 m/s to 1.2 m/s within the study area (EPA 2006), while water velocities of >0.05 m/s are required for benthic drift nets to operate properly (Klemm et al. 1990; Scofield 2007).

When the drift nets were retrieved, contents from the nets were washed into a 0.25 mm sieve and emptied into an appropriately labeled jar for preservation in 70% ethyl alcohol. Ending flowmeter count was also recorded so organisms per volume of water filtered could be calculated. An average of 54 samples per year were collected from 2007-2010. A small number of samples were not included in the study due to net clogging or the flowmeter being covered by algae.

BMI Sample Handling and Processing

For years 2007-2010 a total of 216 samples were collected. BMI were counted and identified at 10X magnification with a stereoscopic dissecting microscope and identified to the lowest practical taxon, which was at least to family. Genus and/or species were identified when possible. Organism keys included Merritt et al. (2008), Thorp and Covich (2010), and Smith (2001). In addition, the first 25 organisms of each taxon were measured to the nearest 0.1 mm using an ocular micrometer.

Most of the benthic drift samples contained a large number of organisms, so samples containing more than 500 organisms were subsampled by a variation of the quantitative fixed-count technique outlined by Moulton et al. (2000). The main objective of the fixed-count method is to estimate the abundance of each BMI taxon sorted from the sample. Instead of homogenizing the sample into one large tray with equal sized quadrants, the BMI samples were homogenized and spread evenly between 10 different square gridded petri dishes. Each tray was divided into 4 quadrants, thus giving a total of 40 quadrats used in the subsample. After the sample was evenly distributed within the gridded petri dishes, a random number generator was used to sort at least 2 different

quadrants or as many quadrats as required to get to the fixed-count number of 500

organisms. When 500 organisms were counted in the subsample, the quadrant was

completed so the total abundance could be determined from a ratio determined by how

many quadrants were sorted. The remaining quadrants were then examined to look for

rare invertebrates that were not originally counted.

White Sturgeon Larval Collection

White sturgeon were collected through double D-frame benthic plankton nets in

the summer of 2015. Net frame openings measured approximately 0.8 m wide at the base

and 0.6 m high. Two nets were placed on a rectangular base (1.6 x 0.7 m) to keep them

upright on the bottom. Each net was approximately 3.4 m long and narrowed to a

removable PVC collection bucket with ports lined in 333 or 750 um Nitex mesh screen.

Deployment of the D-frame nets coincided with the dispersal of white sturgeon larvae in

mid-July and collected by the STOI near the China Bend boat launch (Rkm 1,174).

China Bend is the last known location of white sturgeon larvae and provides the highest

catch rate in Lake Roosevelt (Andy Miller, personal communication).

A total of 10 double D-frame nets were placed at the bottom of Lake Roosevelt

and allowed 0.5-2 hours to sit on the bottom, depending on debris loading. The contents

of the nets were emptied into a white-bottomed tray and sorted for white sturgeon larvae.

Collected sturgeon larvae were fixed in 10% neutral buffered formalin.

White Sturgeon Laboratory Handling and Processing

White sturgeon larvae were processed under a stereoscopic dissecting microscope

and were measured for total length (TL) to the nearest mm. Sturgeon larvae were also

weighed to the nearest 0.1 mg (blotted weight). Fulton's condition factor was used to

compare the length to weight relationship (Parsley et al. 2010). A condition factor of 1

indicates that length and weight are added proportionally as the fish matures. The

condition factor can be calculated using the equation:

$$K_{TL} = \frac{W}{(L_3/10)^3}$$

where K_{TL} is Fulton's condition factor with total length, W is weight (g), and L is total

length (mm). The stomach contents were excised and prey items were identified to the

lowest possible taxon, generally family level. All stomach contents were stored in vials

containing 70% ethyl alcohol for preservation. Statistical analyses were not performed for

the feeding ecology study due to a low sample size.

Data Analysis

Benthic invertebrate data were analyzed by site (littoral/thalweg), river mile, and

year to determine species composition, density (by taxon and total; organisms/m^3),

community structure (diversity, richness, and evenness), and biomass. The mean and

standard error of the BMI density (organisms/m^3), biomass, diversity, richness, and

evenness were used for statistical analysis. Because two benthic drift nets were collected

at each site, these nets were averaged for one sampling occasion per site.

The Shannon-Weaver Index (H) was used to determine diversity (Krebs 1978),

which was calculated using the equation:

$$H = -\sum_{i=1}^{s}(p_i)(log_2 p_i).$$

where H is the Shannon-Weaver Index, p_i is the proportion of individuals belonging to

taxon *i*, and s is the number of taxa. To further investigate diversity and community

structure, taxa richness was estimated by counting the number of taxa observed in each

sample, site, river mile, and year. Evenness (E) was used to analyze the community

structure, as E measures the proportional abundances among different taxa in the dataset

(Krebs 1978). This measure ranges from 0.00 and 1.00 with 1.00 begin complete

evenness. Evenness was calculated using the equation:

$$E = H/log_2 s$$

where E is the evenness, H is the Shannon-Weaver Index, and s is the number of taxa.

Biomass was used to estimate the mass of invertebrates drifting throughout Lake

Roosevelt. Dry weight of individual zooplankton and macroinvertebrates was estimated

using a length-weight regression by measuring the total length of an invertebrate not

including projections or caudal setae. Biomass was calculated using the equation:

$$W = aL^b$$

where a and b are regression coefficients, L is the length of the invertebrate, and W is the

biomass (Appendix A).

The overall BMI densities, preferred white sturgeon larvae prey densities,

diversity, biomass, evenness, and richness were compared between site (littoral and

thalweg), river mile, and by year. General linear regression models were performed on all

calculated BMI population descriptors (density, biomass, diversity, richness, and

evenness). The initial overall model was based created using density as the dependent

variable and site, RM, and year as the independent variables. The best model was

determined by using a step-down method of eliminating non-significant independent variables and comparing AIC values. Density and biomass values were $\log_{10}$ transformed to meet the assumptions for general linear regression analysis.

Results

BMI Drift Composition

Overall, zooplankton in Lake Roosevelt made up 90% of the total drift density. Zooplankton from the family Cyclopidae were found to be the most abundant organisms in the drift, composing 44% of the overall drift. This result held in both the littoral and thalweg sites. In the littoral zone, zooplankton made up 90% of the overall drift, with members from the Cyclopidae composing 43% of the drift; however, in the thalweg, these densities slightly increase to 91% and 48%, respectively.

Excluding zooplankton, *Hydra* (55%), Oligochaeta (16%), and Chironomidae (13%) made up 84% of the benthic macroinvertebrate drift. Surprisingly, *Hydra* was found to be most abundant, in both the littoral (55%) and thalweg (54%), despite being a sessile organism. In the littoral zone, Oligochaeta made up 16% of the drift, whereas in the thalweg, this number declined to 13%. Chironomidae showed a reverse trend from Oligochaeta, and made up 12% in the littoral zone, while composing 17% in the thalweg.

Organisms from taxonomic groups that are traditionally known to drift, such as Ephemeroptera and Plecoptera, were only found to make up a small portion of the drift, with or without zooplankton. Overall, without zooplankton included, Ephemeroptera and Plecoptera made up 0.7% and 0.002%, respectively. In the littoral zone, Ephemeroptera and Plecoptera densities decreased and only composed 0.6% and 0.0002%; however, in

the thalweg, Ephemeroptera composed 1.23%, while Plecoptera were absent from the drift.

Other important prey items for white sturgeon larvae including Chironomidae pupae, Simuliidae, *Mysis*, *Hydracarina*, and Amphipoda were found to compose <1% of the overall and littoral drift; however, in the thalweg, these numbers slightly increased, but made up <2% of the thalweg drift.

BMI Drift Density

Overall, the average total drift density in the littoral zone ranged from 0.72 ± 0.19 organisms/m^3 in 2009 to 300.6 ± 189.88 organisms/m^3 in 2007, whereas the average total drift density in the thalweg ranged from 0.54 ± 0.18 organism/m^3 in 2009 to 10.74 ± 2.97 organisms/m^3 in 2007. One littoral sample with a high density of 1387.1 organisms/m^3 drove a majority of the variance found in 2007 and within the littoral sites. When excluding this value, the average littoral density in 2007 dropped to 119.6 ± 67.6 organisms/m^3. Drifting invertebrates were found in greater densities in years with the highest mean flow from July, with flow values calculated from near the Canadian border (DART 2016). Although 2008 and 2010 had differences in flow, these years had similar variability in flow and density (Fig. 2)

The relationships between BMI density and site, year, and river mile were analyzed in an attempt to explain variations in BMI density throughout the reservoir (Table 1). As expected, total macroinvertebrate density was significantly influenced by site (Fig. 3; $P < 0.001$) and year (Fig. 4; $P < 0.001$), with greater densities in the littoral zone. Drift density between sites and years was variable; however, the BMI density was significantly influenced by the interaction between site and year ($P < 0.001$). Even

though river mile did not show a significant difference in macroinvertebrate density ($P = 0.99$), the interaction between site, year, and river mile did have a significant influence on macroinvertebrate density ($P < 0.05$). Site, year, and river mile accounted for 60% of the variation found within the BMI drift density.

Chironomidae larvae density was significantly influenced by site (Fig 5; $P < 0.001$) and year (Table 4; $P < 0.001$), and the interaction between site and year ($P < 0.001$). Overall, Chironomidae densities were greater in the littoral zone, and remained similar downstream. The mean density of Chironomidae larvae was significantly higher in the littoral zone in 2007 than densities during the other three years, which were not significantly different than each other. Regression analysis indicated that 42% of the Chironomidae density was explained by site, year, and river mile.

Chironomidae pupae density was significantly influenced by site ($P < 0.05$) and year (Table 4; $P < 0.01$), with greater abundances found in the littoral zone. In addition, Chironomidae pupae density was significantly influenced by the interaction of site and year ($P < 0.001$), and by the interaction of site, year, and river mile ($P < 0.05$). Chironomidae pupae densities were significantly higher in the littoral zone in 2007 and decrease while moving downstream; however, every other year remained similar between site and river mile. Overall, 26% of the Chironomidae pupae density was explained by site, year, and river mile.

For Simuliidae larvae mean densities, the density was significantly influenced by year (Table 4; $P < 0.01$), site (Fig. 5; $P < 0.05$), the interaction between year and site ($P < 0.001$), and the interaction between year, site, and river mile ($P < 0.05$). Simuliidae were found in the highest densities in 2007 and within the littoral zone, but were not

significantly different by site in the remaining years or between the remaining years. River mile did not have a significant impact on Simuliidae density (P = 0.78), but a greater variation of densities were found upstream near the Little Dalles when compared to downstream near Kettle Falls.

Temoridae density was significantly influenced by site (Fig. 5; P < 0.001), year (Table 4; P < 0.01), the interaction between site and year (P < 0.001), and the interaction between site, year, and river mile (P < 0.05). Littoral zones had significantly higher densities when compared to thalweg densities. Temoridae density was greatest in 2007 and was significantly different than 2010; however 2008 and 2009 remained similar.

BMI Drift Biomass

The biomass of drifting invertebrates in the littoral zone ranged between 0.59 ± 0.35 mg/m^3 in 2007 and 0.016 ± 0.0097 mg/m^3 in 2009. In the thalweg, the mean biomass ranged between 0.028 ± 0.0075 mg/m^3 in 2007 and 0.0038 ± 0.0017 mg/m^3 in 2009. Throughout the study, zooplankton biomass dominated the drift in both littoral and thalweg zones.

The relationships between BMI biomass and site, year, and river mile were analyzed in an attempt to explain variations in BMI biomass throughout the reservoir (Table 2). BMI biomass was significantly influenced by site (Fig. 6; P < 0.01), with a greater biomass in the littoral zone. Biomass of drifting BMIs was also significantly influenced by year (Fig. 7; P < 0.001), and the interaction between site and year (P < 0.001). The littoral biomass was consistently greater than the thalweg biomass; however,

the biomass within each site remained variable when comparing these values between years.

Chironomidae biomass was significantly influenced by site (Fig. 8; P < 0.01), year (Table 5; P < 0.01), and the interaction between year and river mile (P < 0.01). River mile had a slightly influenced Chironomidae biomass (P = 0.0501), with increasing biomass downstream (Fig. 9). In general, larger Chironomids were found downstream of the Canadian border.

Chironomidae pupae biomass was significantly influenced by year (Table 5; P < 0.01), and the interaction between site and year. Similar to the interaction with density, the littoral zone in 2007 supported greater densities than any other year and site.

There were few differences in Simuliidae biomass by site (Fig 8; P = 0.146), year (Table 5; 0.0715), and river mile (P = 0.190); however, Simuliidae biomass was significantly influenced by the interaction between site and year (P < 0.05). Simuliidae biomass was similar from 2008 – 2010 in both sites, but 2007 had a greater biomass in 2007 when compared to the other years.

Biomass of Temoridae was significantly influenced by site (Fig. 8; P < 0.05), with a greater biomass found within the littoral zone. Year had a significant influence on drifting Temoridae (Table 5; P < 0.05), with the greatest biomass in 2007. There was also a significant interaction between site and year (P < 0.001), which could be explained by the increase in littoral biomass in 2007 and 2010.

BMI Community Structure

The relationship between diversity, richness, and evenness to site, year, and river mile were analyzed using the same general linear regression model as density (Table 3). BMI diversity was significantly influenced by year (Fig. 10; $P < 0.001$), with the highest diversity found in 2009. Shannon-Weaver values ranged between 0.71 in 2008 to 2.15 in 2009, with both the low and high values found within the thalweg. Years that showed lower mean flow values had a higher diversity, due to greater concentrations and more families of macroinvertebrates found within the drift. There was also a significant interaction between year and river mile ($P < 0.05$). Shannon-Weaver values remained similar by river mile within years; however, these values were significantly different between years.

BMI richness was significantly influenced by year (Fig. 11; $P < 0.001$), with 2010 containing the most taxa found. Similar to diversity, more taxa were found in 2009 and 2010, which were years showing the lowest mean flow. Even though the density and biomass of drifting invertebrates were higher in 2007 and 2008, there was a greater diversity of insects found in 2009 and 2010. Site and river mile had little impact on BMI richness ($P = 0.99$, $P = 0.22$, respectively). BMI evenness had similar results as richness and was significantly influenced by year (Fig. 12; $P < 0.001$), with 2008 having the highest evenness.

Sturgeon Stomach Contents

In total, 590 sturgeon larvae were captured during June and July 2015 near the China Bend boat launch. Sturgeon larvae ranged in size from 14.7 to 21.8 mm TL (median = 19.1, mean = 19.0, n = 590) and from 22.4 to 56.4 mg (median = 37.5, mean =

37.8, n = 590) blotted weight. The mean condition factor (Fulton's K) was 0.60 (SD = ±

0.07) for sturgeon larvae. Of the 591 larvae that were collected, 98.8% (n = 583) of the

larvae still contained portions of their yolk sac, indicating the larvae were not yet ready to

begin exogenous feeding. Typically, sturgeon that were longer in length, also weighed

more (Fig. 13). Only 1.53% (n = 9) of sturgeon, which ranged in size from 18.2 and 21.8

mm TL had prey items in their stomach contents. Of that 1.53%, six of the nine sturgeon

showed mixed feeding, meaning prey items and yolk were found in their digestive tracts.

A total of 1.18% (n = 7) of the sturgeon larvae had completely exhausted their yolk sac;

however, 58% (n = 4) of the sturgeon that exhausted their yolk had no prey items present

in their stomach contents.

A total of 14 prey items were found in the 9 stomachs, in which prey items ranged

in size from 0.4 to 3.9 mm (Fig. 14; median = 1.5, mean = 1.66, n = 9) in length. Of the

14 prey items found, 85.7% (n = 12) were greater than 1.0 mm, while 14.3% (n = 2) of

the prey items were smaller than 1.0 mm, being partially digested. Five sturgeon

contained two prey items in their stomachs, while the remaining four contained just one

prey item. Sturgeon that had portions of their yolk preyed upon copepods, isopods, and

Dipteran larvae and pupae, and portions of a mayfly (Fig. 15). Smaller prey items were

found in sturgeon that still contained portions of their yolk (mean = 1.43 mm, SD = ±

0.45) when compared to sturgeon that had completely exhausted their yolk sac (mean =

1.96 mm, SD = ± 1.23).

Discussion

Our results stress the temporal and spatial variation in benthic macroinvertebrate populations in the upper portions of Lake Roosevelt. In this study, littoral density and biomass were significantly higher than thalweg densities in the upper portions of Lake Roosevelt, which supported my first hypothesis. Even though the littoral zone supported higher densities and biomass, littoral diversity, richness, and evenness did not increase. The low densities of BMI found in the thalweg suggest that the thalweg may be a poor area for white sturgeon larvae to feed as they are drifting downstream. Despite the low densities of BMI found within the thalweg, Black et al. (2003) found that fishes in Lake Roosevelt that are usually considered obligate benthivores, consumed pelagic carbon almost exclusively. Zooplankton were found in both littoral and thalweg zones to compose >90% of the overall drift, and can explain why Lake Roosevelt fishes consume high amounts of pelagic carbon.

Despite the difficulties of comparing BMI data performed by different researchers (e.g. sieve size, experimental design, statistical analyses; McCabe et al. 1997), it is important to try to understand differences in composition and abundance between different systems. In the subarctic River Alta, Norway, overall drift densities ranged from 0.85 ± 0.13 organisms/m^3 to 52.6 ± 6.1 organisms/m^3 in April, while zooplankton composed 96 to 100% of the overall drift (Johansen et al. 2010). Summer densities in this river are known to be up to 24 times greater during the summer months (Johansen et al. 2010). Downstream of the Glen Canyon Dam, mean densities of Chironomidae and Simuliidae were 3.2 organisms/m^3 and 0.30 organisms/m^3 (Kennedy et al. 2014), which were greater than mean densities of these taxa in this study. McCabe et al. (1997) found

mean densities of BMIs below the Bonneville Dam in the main channel around 1,000

organisms/m^2, which is consistent with other studies performed in the Fraser River,

Brittish Columbia and the Hudson River, New York (Northcote et al 1976; Ristich et al.

1977). Although these studies performed different sampling techniques, these results

give us a wide range of studies to compare. In the upper portions of Lake Roosevelt,

zooplankton communities composed roughly 90% of the overall drift; however, mean

densities were similar to winter densities in Norway (Johansen et al. 2010), while

individual taxa densities were slightly lower than below the Grandby Dam on the

Colorado River (Kennedy et al. 2014). Fluctuating flow can often decrease BMI density

and biomass (Ward 1975), and could explain why low values were found in the upper

reaches of Lake Roosevelt. Although direct comparisons cannot be made, densities of

drifting invertebrates in Lake Roosevelt appear low in comparison to other systems.

Lotic ecosystems are often influenced by flow regime and can influence

environmental factors and life history traits of BMIs (Poff et al. 1997). As reservoirs

reach full pool, peak flows can impact communities of invertebrates downstream, which

often cause invertebrate drift (Perry and Perry 1986; Bruno and Siviglia 2012). Flow in

Lake Roosevelt is highest from April through July as the reservoir is reaching full pool,

and the inflow during these months accounts for 65 to 70% of the total inflow volume for

the year (USBR 2014). In the Colorado River, downstream of the hydropeaking Glen

Canyon Dam, a model by Kennedy et al. (2014) predicted that a benthic Chironomidae

density of around 100 organisms/m^3 would correlate to a drift density of 0.1

organisms/m^3, similar to Chironomidae drift densities found in Lake Roosevelt.

Contrary to our second hypothesis that BMI density and biomass would decrease moving downstream, BMI density and biomass remained similar while moving downstream (H$_2$). White sturgeon larvae drift from near the Canadian border to downstream habitats, thus indicating the importance of the spatial variation of BMIs in Lake Roosevelt. Benthic drift is positively correlated to discharge (Waringer 1992; Johansen et al. 2010; Kennedy et al. 2014); however, this is not the entire picture, as benthic densities and flow are often closely related. The high water velocities near the Canadian border could be limiting standing crops of specific BMIs by not allowing detritus and organic matter to sufficiently support benthic invertebrates (McCabe et al. 1997). Therefore, without a large standing crop of benthic macroinvertebrates, there will not be a high density of drifting invertebrates, which could likely be occurring in Lake Roosevelt.

In general, higher densities and biomasses were found in Lake Roosevelt in years with higher flow, which supported our third hypothesis. Sampling in 2009 occurred approximately one week prior to previous years, due to sturgeon spawning slightly earlier than in previous years. July 2009 had a higher mean flow than in 2010, but showed the lowest density and biomass values. The low density and biomass found in 2009 could be due to the sampling events occurring approximately one week after Lake Roosevelt reached full pool. Not only flow, but also the rate of increase or variability in flow could be impacting invertebrate communities (Ellis and Jones 2013). Due to the drawdown events in Lake Roosevelt, benthic productivity does not begin until early summer, while significant benthic production may not occur until late summer (Black et al. 2003). If significant benthic production does not begin until late summer, and sturgeon spawn in

early to mid July, there will not be great enough densities during the period where white sturgeon transition to exogenous feeding.

In 1997, historic flow levels were found in Lake Roosevelt, which was supplemented by limited natural recruitment (Howell and McLellan 2014). This event led to the hypothesis that larval transport could be a key factor in recruitment (Howell and McLellan 2013, 2014). High flow was again seen in 2011, in which limited recruitment did occur (Howell and McLellan 2014). At the U.S.-Canada border, historic records indicate flows were often greater than 4,500 cms during spring flows (Hildebrand and Parsley 2013); however, mean daily flows from 2007 – 2010 during spring peak were nearly 1,000 cms less. Although the larval transport hypothesis is viable for a main contributor for recruitment failure, flow does not describe the entire picture. Flow is a major factor that controls invertebrate drift, whether it is behavioral or accidental (Brittain and Eikeland 1988). The high flows found in 1997 and 2011 could be acting on populations of white sturgeon recruitment in two different ways related to prey availability: 1) high flow events transport larvae downstream to a suitable habitat (ie. high density of prey) before their yolk is exhausted or 2) drift found in upper Lake Roosevelt originate from benthic communities in the Canadian Reach and are transported further downstream with high flow events. Despite the low drift densities found during our study, nine of the sturgeon larvae examined consumed prey items.

The feeding ecology for white sturgeon larvae in Lake Roosevelt in our study, agreed with findings from Howell and McLellan (2013). Even though 590 white sturgeon larvae stomachs were examined, only nine sturgeon contained prey items, which is less than the 27 sturgeon that Howell and McLellan (2013) found in 2008. Although

fewer prey items were found in this study, similar results were found that dipteran larvae and pupae (Chironomidae larvae and pupae and Simuliidae) remain the most important prey item for white sturgeon larvae in Lake Roosevelt. In this study, copepods from the Temoridae family were found to also be an important prey item and were found to be the most abundant individual organism in the stomach contents. Interestingly, only sturgeon that still contained their yolk sac contained copepods in their stomach. Sturgeon that had completely exhausted their yolk were found consuming larger prey items and preferred Dipteran larvae and pupae. Overall, Chironomidae and Temoridae were found to have similar densities in both littoral and thalweg zones and were found in roughly the same amount of sturgeon larvae in this study.

Because the principal focus of the study was larval white sturgeon prey availability, our sampling was limited to July while larvae were drifting downstream. Sampling benthic drift in June (prior to spawning), July (while drifting downstream), and August (after drifting downstream) would be beneficial in the future to examine if white sturgeon larvae impact the abundance of drifting invertebrates. Furthermore, this method would also give insight as to how the drastic drawdown events impacts the invertebrate population in Lake Roosevelt that is vital for white sturgeon larval development, and could indicate how far white sturgeon are drifting past China Bend.

In addition, only 7 of our 590 white sturgeon larvae had completely exhausted their yolk. These results could indicate that the China Bend boat launch is not a viable location for feeding ecology studies. Sampling locations located further downstream could be beneficial to allow white sturgeon larvae a slightly longer time period for development. It could also be beneficial to collect white sturgeon larvae in conjunction

with BMI drift nets to fully understand how the prey availability in Lake Roosevelt is impacting the white sturgeon larvae diet. Furthermore, sampling of white sturgeon for diet analysis during the day and night could indicate if sturgeon are feeding during the night as they drift or during the day in the shelter of aquatic plants.

In conclusion, our finding suggest that prey availability could be contributing to white sturgeon recruitment failure in Lake Roosevelt. Spatial and temporal changes in BMI density and biomass suggest that reservoir operations could be limiting the distribution and abundance of BMIs when sturgeon are at their most vulnerable life stage. Increased flow consistency during the summer months in Lake Roosevelt could help support greater abundances of BMIs and available prey in upper Lake Roosevelt. Although further studies need to be performed to validate that prey availability is a cause of recruitment failure, our results strongly indicate that the concentrations of drifting invertebrates in Lake Roosevelt is not sufficient to support the development of white sturgeon larvae and is a likely cause of recruitment failure.

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Tables

Table 1. F-statistic, r^2, and level of significance for comparison of regression models of

drift density on site, RM, and year throughout upper Lake Roosevelt from 2007-2010.

Taxon	F	r^2	P-Value
Overall	12.36	0.6076	2.38e-16
Chironomidae Larvae	6.408	0.4244	3.46e-09
Chironomidae Pupae	3.64	0.2647	5.44e-05
Simuliidae	3.99	0.2896	1.49e-05
Temoridae	4.126	0.2989	9.12e-06

Table 2. F-Statistic, r^2, and level of significance for comparison of regression models for drift biomass on site, RM, and year throughout upper Lake Roosevelt from 2007-2010.

Taxon	F	r^2	P-Value
Overall	3.98	0.2892	1.537e-05
Chironomidae Larvae	3.382	0.2452	0.0001416
Chironomidae Pupae	3.632	0.2642	5.594e-05
Simuliidae	2.071	0.1274	0.01785
Temoridae	2.672	0.1857	0.001999

Table 3. F-Statistic, r^2, and level of significance for comparison of regression models for drift diversity, richness, and evenness on site, RM, and year throughout upper Lake Roosevelt from 2007 – 2010.

Taxon	F	r^2	P-Value
Shannon-Weaver	20.05	0.7184	< 2.2e-16
Richness	3.139	0.2227	0.0003389
Evenness	17.29	0.6858	< 2.2e-16

Table 4. Mean densities (#/m^3) and standard error of all important prey items for white sturgeon larvae and mean flow (cms) and standard error from April 1 – July 31 taken from near the Canadian border (DART 2016). L represents larvae and P represents pupae.

|	2007	2008	2009	2010
Chironomidae (L)	1.09 (0.68)	0.025 (0.0064)	0.15 (0.037)	0.054 (0.015)
Chironomidae (P)	0.065 (0.034)	0.0052 (0.0016)	0.0040 (0.0010)	0.0056 (0.0012)
Simuliidae	0.058 (0.033)	0.0031 (0.0011)	0.0016 (0.00051)	0.0058 (0.0022)
Mysis	0.042 (0.033)	0.0015 (0.00083)	1.91E-05 (1.33E-05)	0.00031 (0.00023)
Temoridae	0.81 (0.58)	0.070 (0.024)	0.0012 (0.00036)	0.12 (0.049)
Cyclopidae	39.18 (25.09)	5.25 (1.00)	0.059 (0.011)	1.80 (0.45)
Diaptomidae	3.27 (1.73)	1.02 (0.17)	0.013 (0.0041)	0.30 (0.068)
Daphnia	2.46 (1.06)	0.041 (0.009)	0.021 (0.0091)	0.16 (0.067)
Hydracarina	0.0049 (0.0019)	0.0019 (0.00077)	0.0020 (0.00073)	0.0099 (0.0040)
Flow	3,815 (28.37)	3,742 (168.42)	3,019 (75.94)	2,674 (118.77)

Table 5. Mean biomass (mg/m^3) and standard error of all important prey items for white sturgeon larvae and mean flow (cms) and standard error from April 1 – July 31 taken from near the Canadian border (DART 2016). L represents larvae and P represents pupae.

|	2007	2008	2009	2010
Chironomidae (L)	0.0017 (0.00054)	9.47E-05 (3.25E-05)	0.0015 (0.00059)	0.00036 (0.00020)
Chironomidae (P)	0.0018 (0.00076)	0.00026 (0.00010)	0.00012 (3.10E-05)	0.00033 (6.79E-05)
Simuliidae	0.021 (0.0016)	0.00013 (4.78E-05)	1.12E-05 (5.14E006)	0.00011 (3.78E-05)
Mysis	4.44E-05 (3.42E-05)	4.56E-06 (3.05E-06)	1.47E-08 (1.05E-08)	1.32E-05 (1.19E-05)
Temoridae	0.013 (0.0084)	0.0013 (0.00051)	1.29E-05 (3.72E-06)	0.0020 (0.00087)
Cyclopidae	0.063 (0.045)	0.0069 (0.0015)	6.73E-05 (1.35E-05)	0.0022 (0.00077)
Diaptomidae	0.013 (0.0056)	0.0026 (0.00041)	3.11E-05 (4.81E-06)	0.0012 (0.00040)
Daphnia	0.012 (0.0056)	0.00012 (3.07E-05)	0.00012 (6.52E-05)	0.00058 (0.00020)
Hydracarina	0.00011 (9.04E-05)	5.57E-05 (3.42E-05)	2.91E-05 (1.83E-05)	6.88E-05 (3.10E-05)
Flow	3,815 (28.37)	3,742 (168.42)	3,019 (75.94)	2,674 (118.77)

Figures
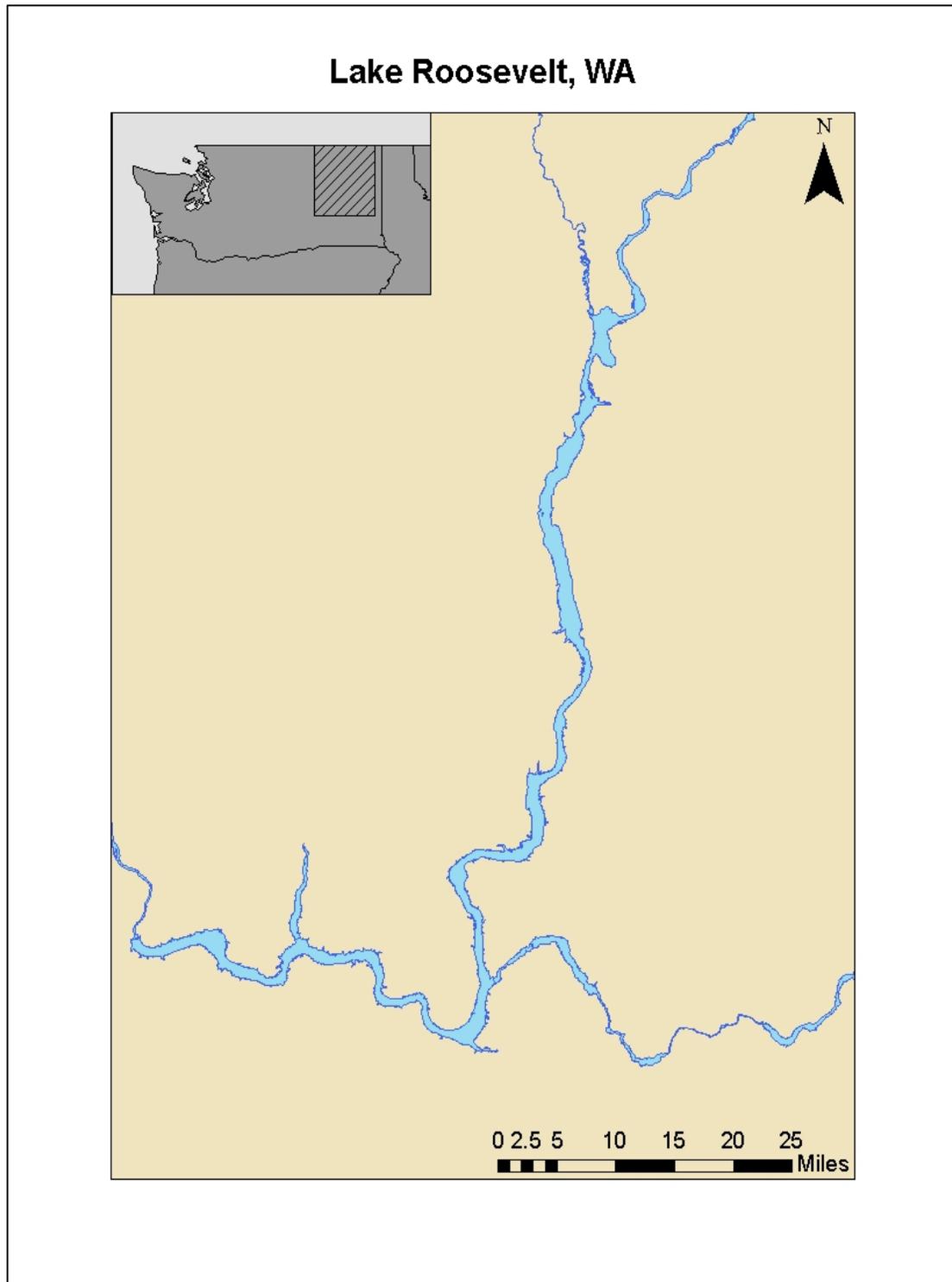
**Figure 1.** Map of Lake Roosevelt, WA, a reservoir of the Columbia River.
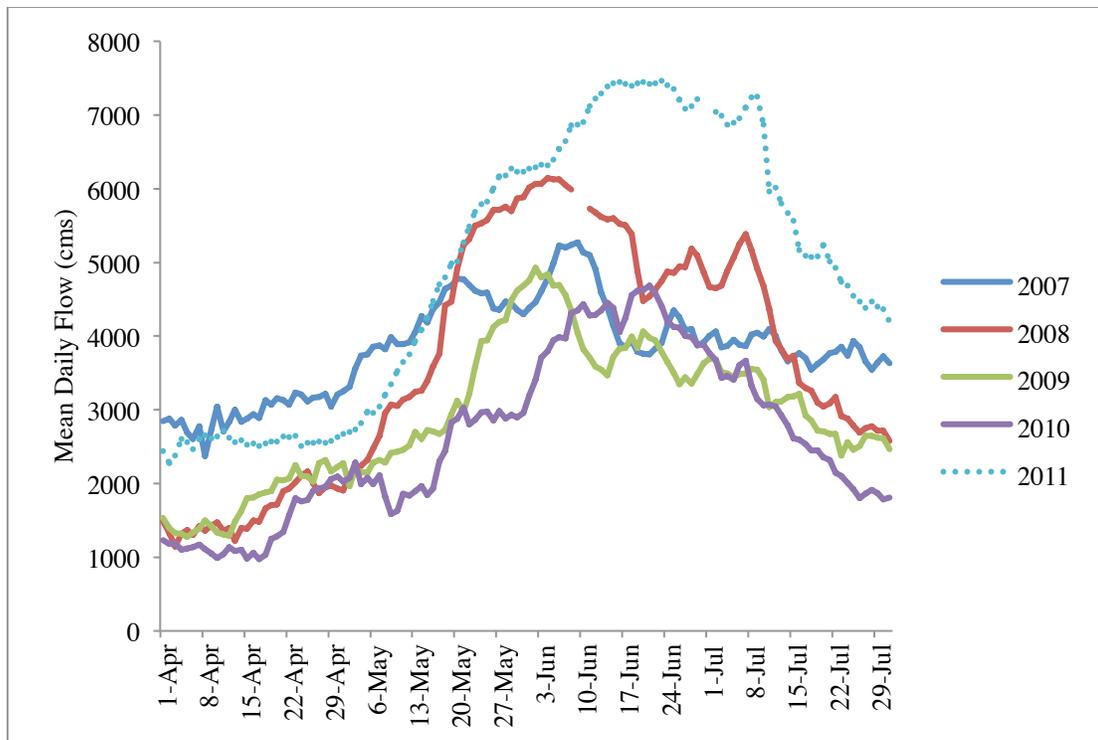
Figure 2. Daily flow (cms) collected from the Canadian border in Lake Roosevelt from April 1 – July 31, 2007-2010.
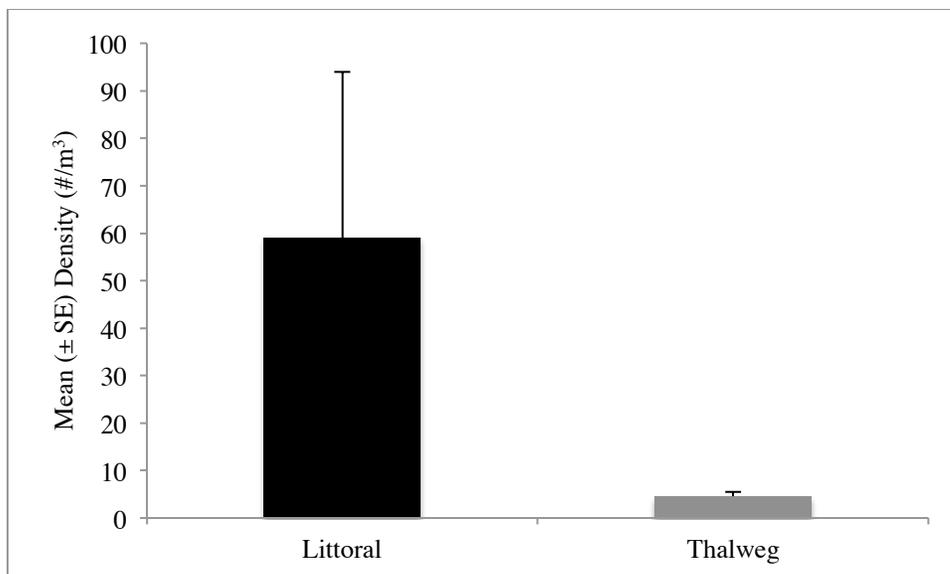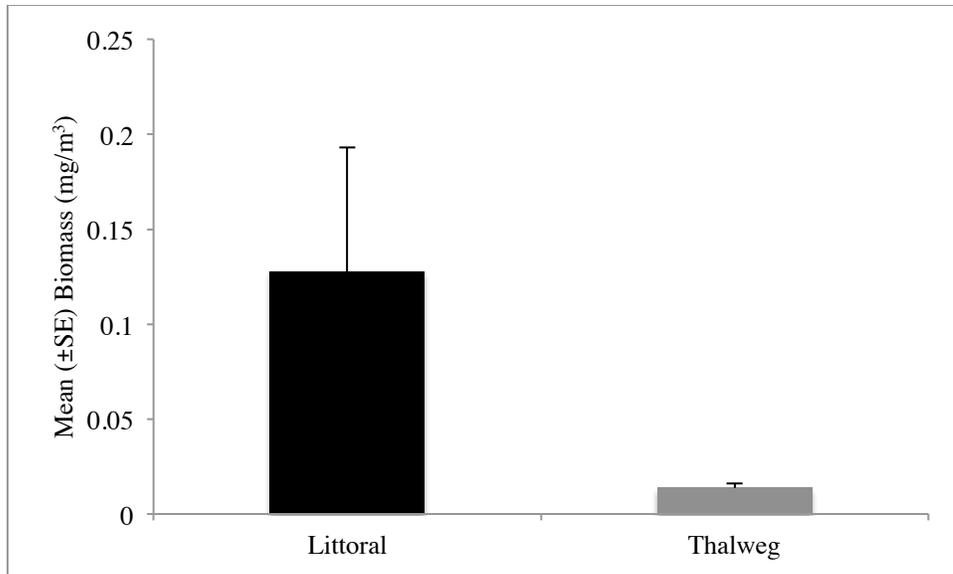
Figure 3. Mean (±SE) BMI density found within the littoral and thalweg sites from years 2007-2010. Significant differences were determined by general linear model (P < 0.001).
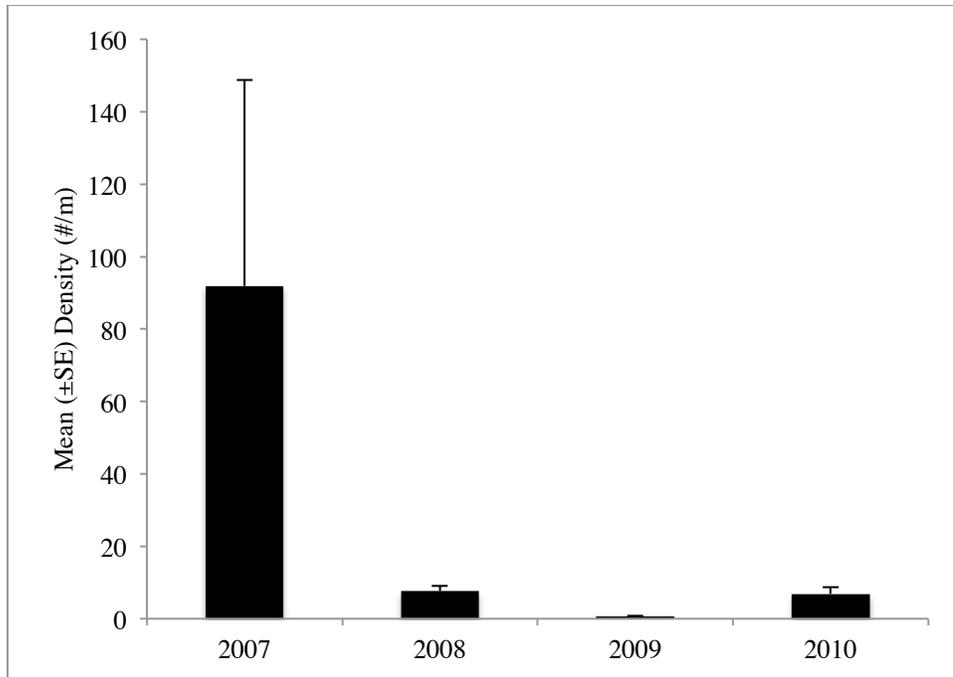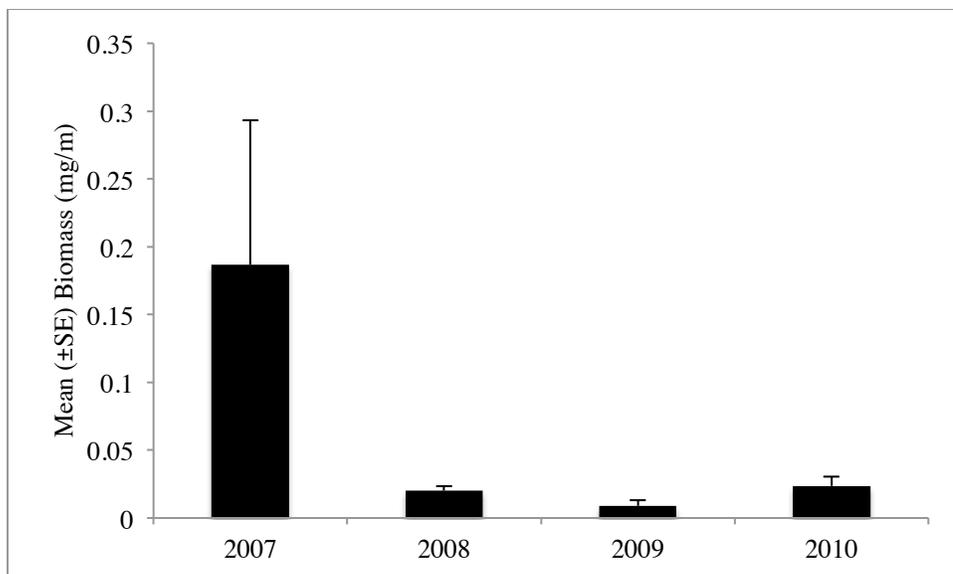
Figure 4. Mean (±SE) BMI density between years found within drift in Lake Roosevelt.

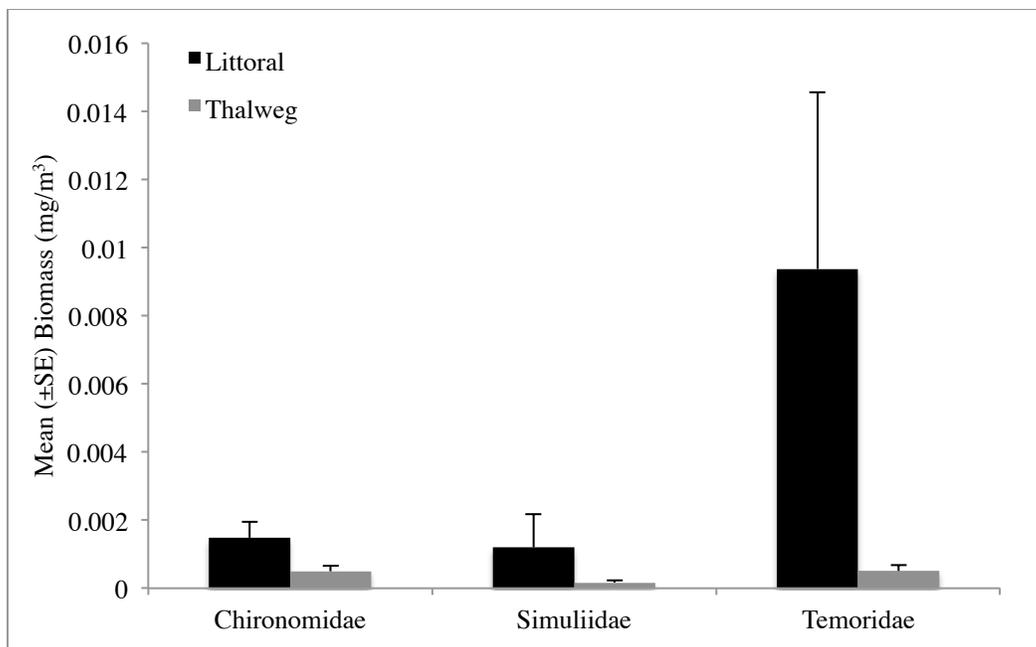Significant differences were determined by general linear regression model (P < 0.001).
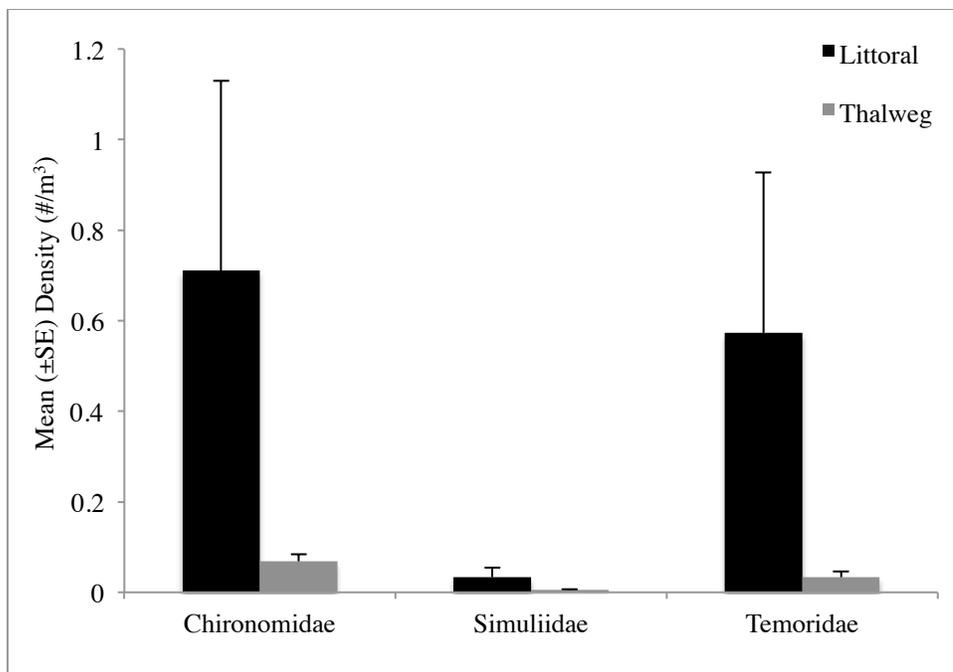
Figure 5. Mean (±SE) BMI density for prey most important to white sturgeon larvae found within littoral and thalweg sites. Significant differences were determined by general linear regressions for Chironomidae (P < 0.001), Simuliidae (P < 0.05), and Temoridae (P < 0.001).

Figure 6. Mean (±SE) BMI biomass (mg/m^3) within the drift in littoral and thalweg sites from 2007 – 2010 in Lake Roosevelt. Significant differences were determined by general linear regression (P < 0.01).

Figure 7. Mean (±SE) BMI biomass (mg/m^3) within the drift in Lake Roosevelt from

2007 – 2010. Significant differences between years were determined by general linear

regression (P < 0.001).

Figure 8. Figure 4. Mean (±SE) BMI biomass (mg/m^3) for prey most important to white sturgeon larvae found within littoral and thalweg sites. Significant differences were determined by general linear regressions for Chironomidae (P < 0.01), Simuliidae (P = 0.14598), and Temoridae (P < 0.05).
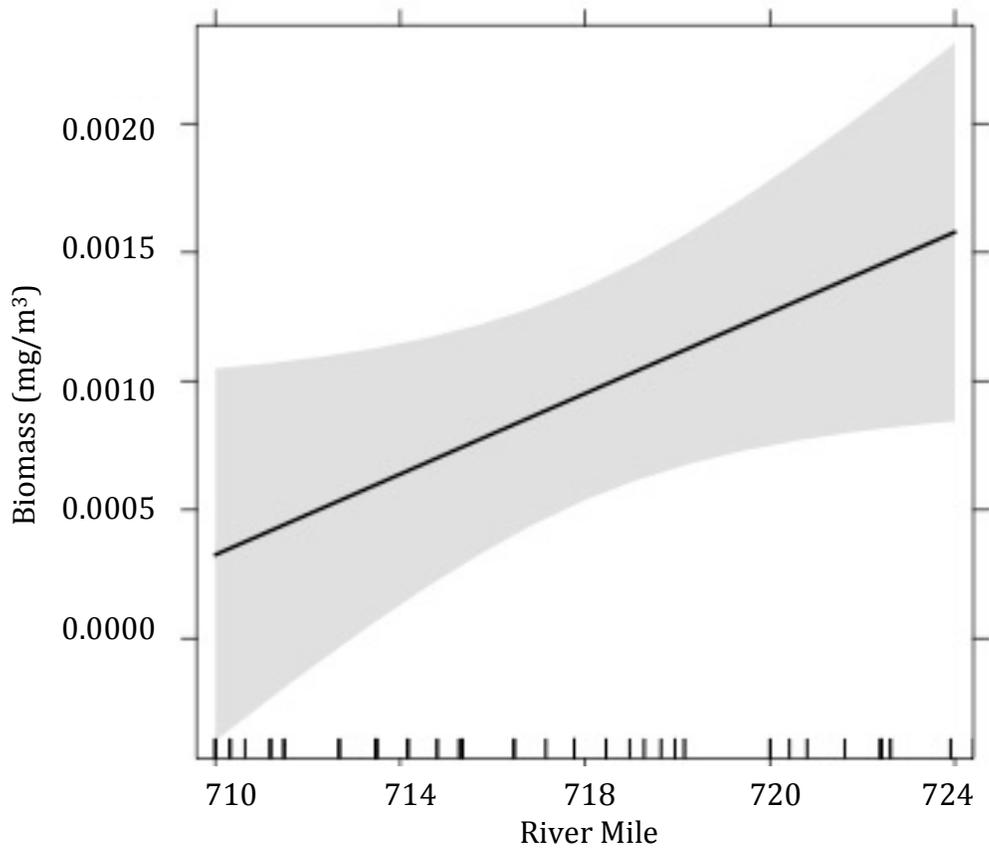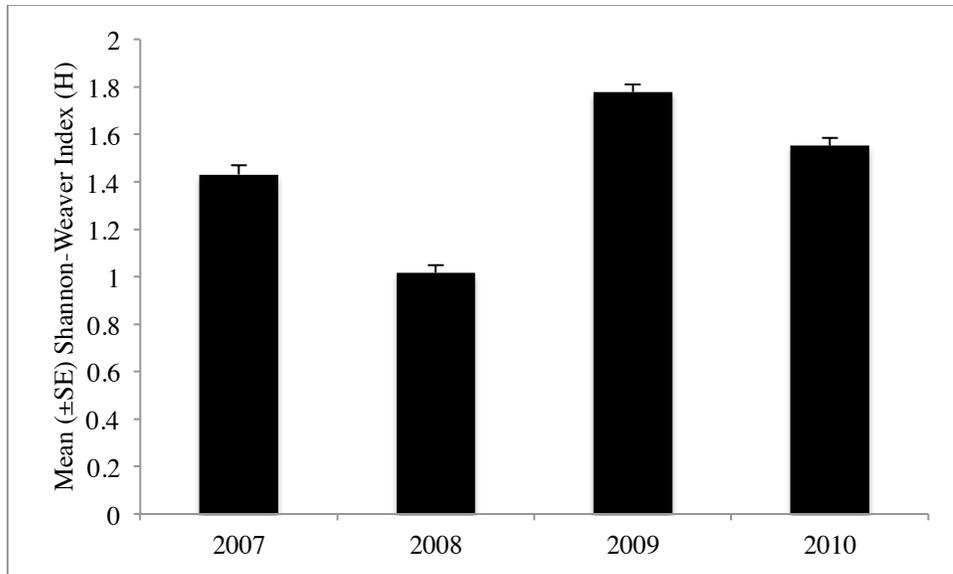
Figure 9. Regression of Chironomidae drift biomass on river mile in Lake Roosevelt from 2007 – 2010. Significant differences were determined by general linear regression (P = 0.0501).

Figure 10. Mean (±SE) Shannon-Weaver diversity index (H) values from 2007-2010 in upper Lake Roosevelt. Significant differences between years were determined by general linear regression (P < 0.001).

Figure 11. Mean (±SE) number of taxa found from 2007-2010 in Lake Roosevelt. Significant differences between years were determined by general linear regression ($P < 0.001$).

Figure 12. Mean (±SE) evenness value of drifting BMIs in Lake Roosevelt from 2007 –

2010. Significant differences between years were determined by general linear

regression (P < 0.001).

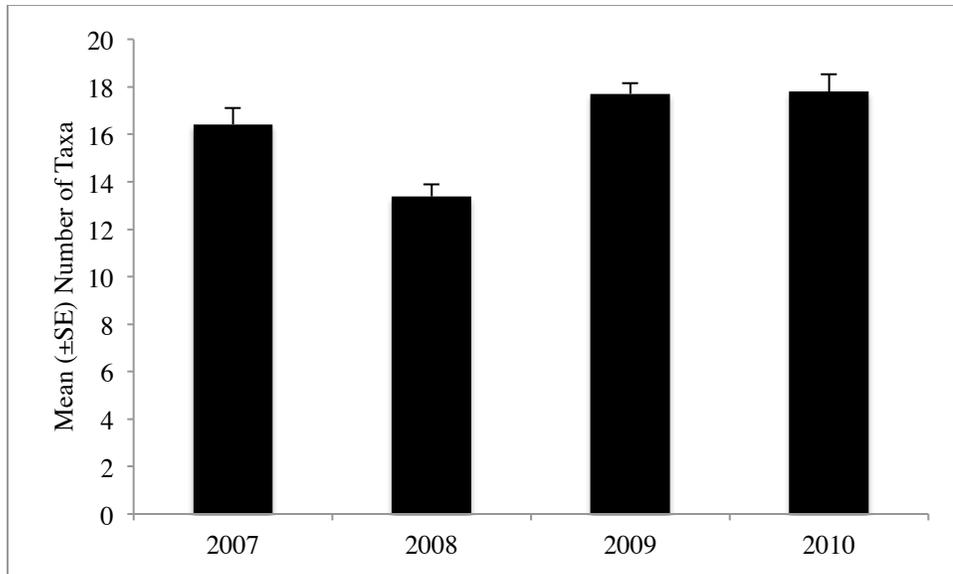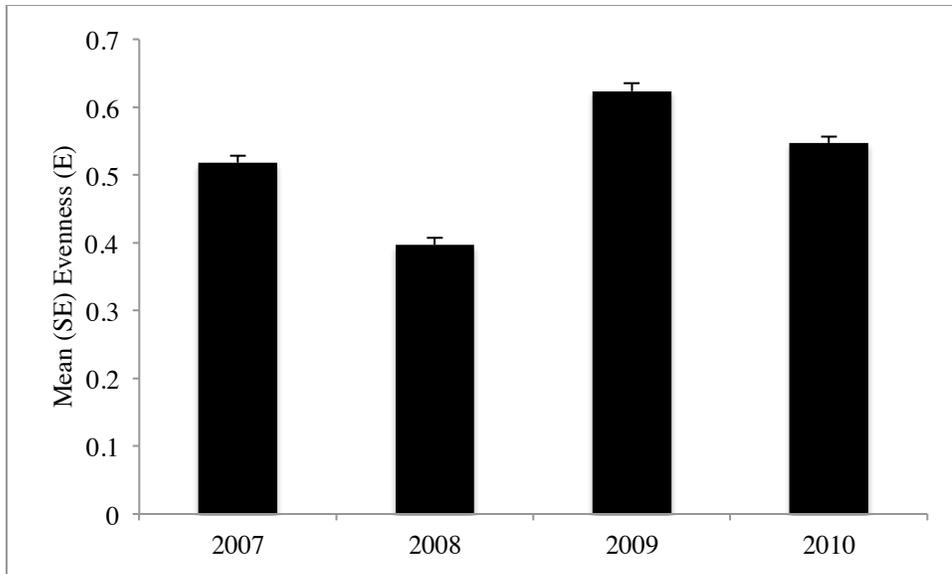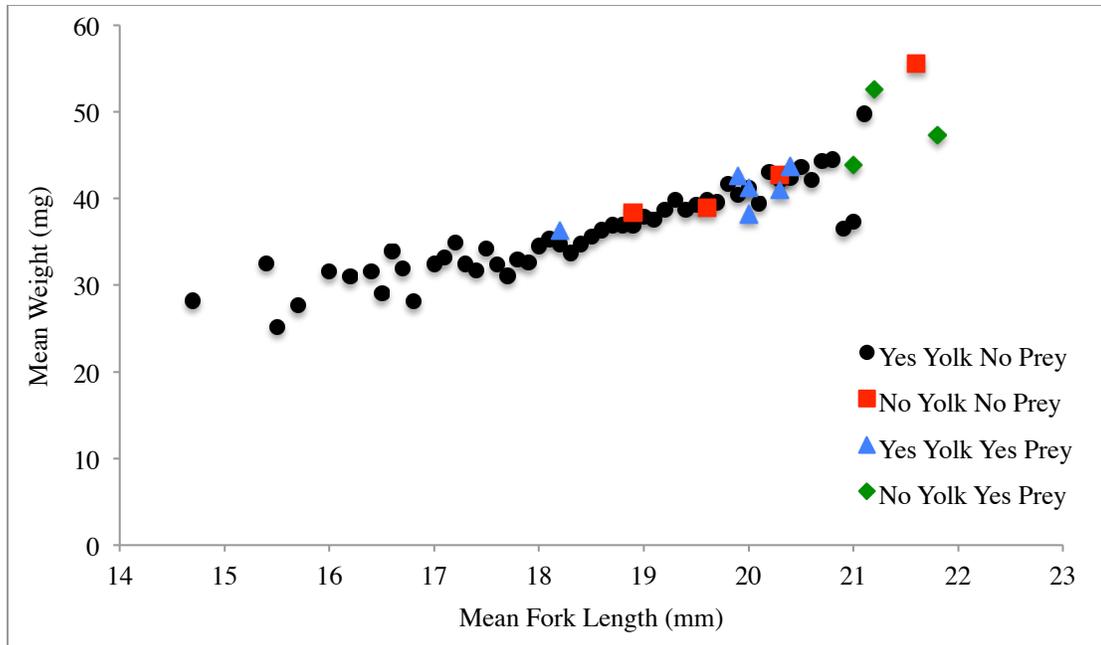**Figure 13**. Mean length versus mean weight of white sturgeon larvae in Lake Roosevelt.
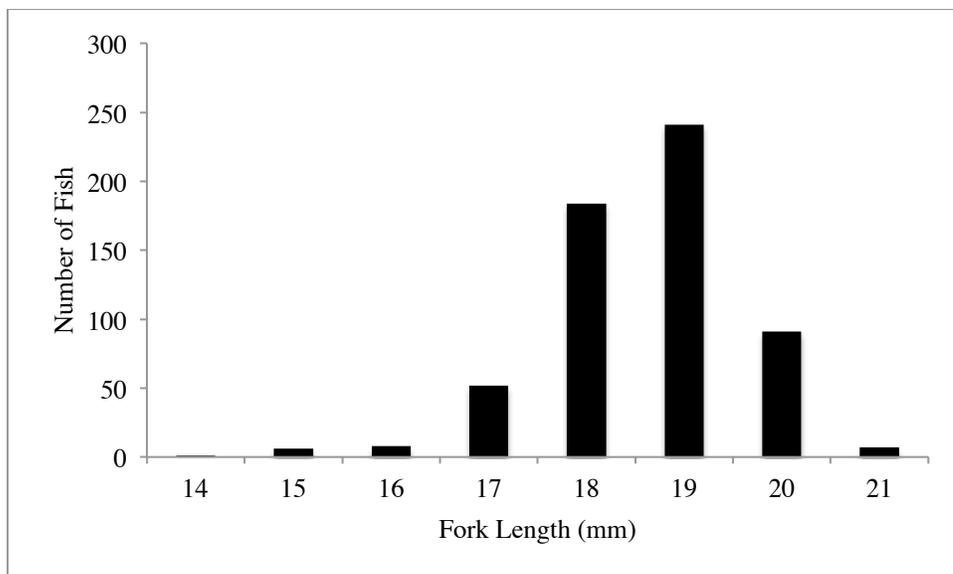
Figure 14. Distribution of white sturgeon larvae size (mm) and number of fish examined in each size class.
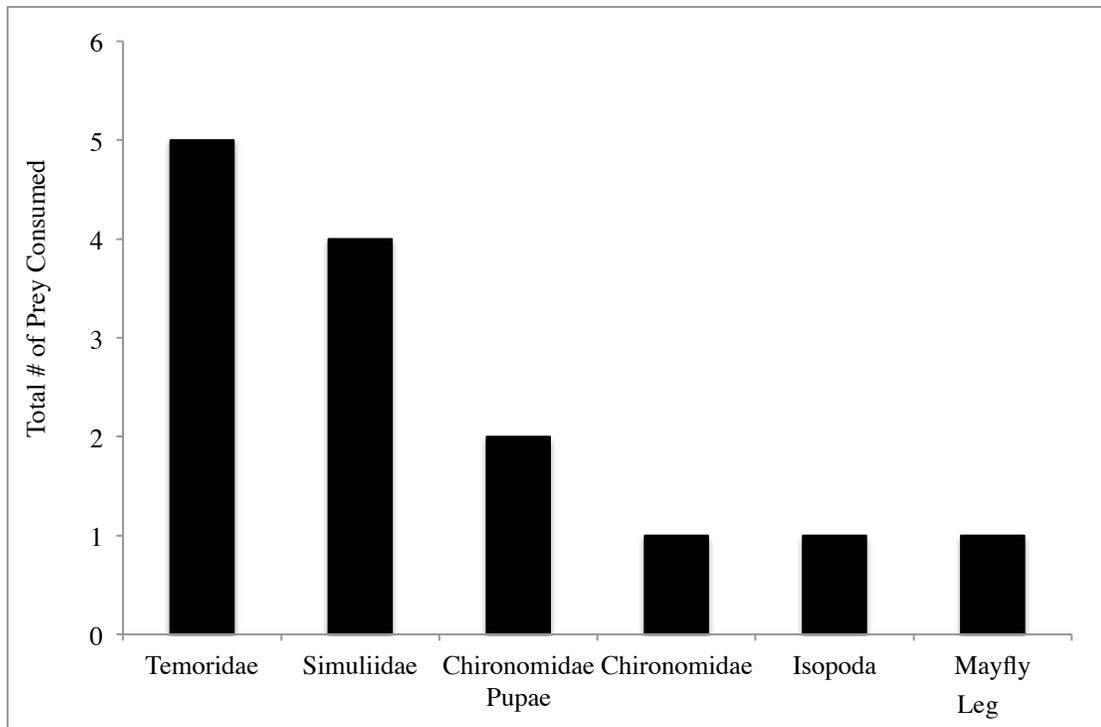
Figure 15. Total number of prey items found in white sturgeon larvae stomach samples.

APPENDIX A

Regression coefficients used in the estimation of biomass (W) from length measurements

of invertebrate and zooplankton taxa using the formula $W = Ax^b$.

Order	Taxa	Coefficient a	Coefficient b	Source
Amphipoda	Gammarus	0.0058	3.015	Benke et al. 1999
	Hyalella	0.0058	3.015	Benke et al. 1999
Isopoda	Asellus	0.0054	2.948	Benke et al. 1999
Coleoptera	Dytiscidae	0.0077	2.91	Benke et al. 1999
	Gyrinidae	0.0077	2.91	Benke et al. 1999
Diptera	Adult	0.00465	2.4222	Matousek 2007
	Ceratopogonidae	0.0025	2.469	Benke et al. 1999
	Chaoboridae	0.0018	2.617	Benke et al. 1999
	Chironomidae (L)	0.0018	2.617	Benke et al. 1999
	Chironomidae (P)	0.004651	2.4222	Matousek 2007
	Dixidae (P)	0.004651	2.4222	Matousek 2007
	Simuliidae	0.002	3.011	Benke et al. 1999
	Syrphidae	0.0025	2.692	Benke et al. 1999
	Tipulidae (P)	0.0029	2.681	Benke et al. 1999
	Ephemeroptera	0.0071	2.832	Benke et al. 1999
Ephemeroptera	Heptageniidae	0.0108	2.754	Benke et al. 1999
	Ephemerellidae	0.0103	2.676	Benke et al. 1999
	Baetidae	0.0053	2.875	Benke et al. 1999
	Baetis	0.007432	2.5798	Benke et al. 1999
	Labiobaetis	0.0053	2.875	Benke et al. 1999
	Procloeon	0.0053	2.875	Benke et al. 1999
	Ephemerella	0.01125	2.5535	Benke et al. 1999
	Caudatella	0.0103	2.676	Benke et al. 1999
	Drunella	0.01221	2.904	Benke et al. 1999
	Cinygmula	0.0108	2.754	Benke et al. 1999
	Maccaffertium	0.0103	2.676	Benke et al. 1999
	Hemiptera	0.0108	2.734	Benke et al. 1999
Hemiptera	Callicorixa	0.0031	2.904	Benke et al. 1999
	Plecoptera	0.0094	2.754	Benke et al. 1999
Plecoptera	Nemouridae	0.0056	2.762	Benke et al. 1999
	Perlodidae	0.0196	2.742	Benke et al. 1999
	Isoperla	0.0094	2.754	Benke et al. 1999
	Trichoptera	0.0056	2.839	Benke et al. 1999
Trichoptera	Brachycenridae	0.0083	2.818	Benke et al. 1999
	Brachycentrus	0.0025	2.6	Benke et al. 1999
	Hydropsychidae	0.0046	2.926	Benke et al. 1999
	Hydropsyche	0.0041	2.97	Benke et al. 1999
	Ceraclea	0.0034	3.212	Benke et al. 1999
	Cyclopidae	0.00000011	2.59	Dumont et al. 1975
Cyclopoida	Diaptomidae	0.0077	2.33	Dumont et al. 1975

Calanoida	Temoridae	6.5	2.63	Johannsson et al. 2000
	Bosminidae	0.0266	3.13	Dumont et al. 1975
Cladocera	Acroperus	0.00905	0.85	Dumont et al. 1975
	Alona	0.01592	3.84	Dumont et al. 1975
	Alonella	0.00017	1.39	Dumont et al. 1975
	Chydorus	0.08943	3.93	Dumont et al. 1975
	Eurycercus	0.01592	3.84	Dumont et al. 1975
	Leydigia	0.01592	3.84	Dumont et al. 1975
	Daphnia	0.000000024	2.77	Dumont et al. 1975
	Ceriodaphnia	0.0000017	2.26	Dumont et al. 1975
	Scapholeberis	0.000000089	2.7	Dumont et al. 1975
	Simocephalus	0.00743	3.28	Dumont et al. 1975
	Leptodoriidae	1.11E-06	3.678	Chipps and Bennett 2000
	Macrothricidae	0.000000024	2.77	Dumont et al. 1975
	Diaphanosoma	0.00000176	2.11	Dumont et al. 1975
	Latona	0.00000176	2.11	Dumont et al. 1975
	Sida	0.00000176	2.11	Dumont et al. 1975
	Harpacticidae	0.01251	4.4	Dumont et al. 1975
Harpacticoida	Hydridae	N/A	N/A	N/A
Anthomedusae	Hirudinea	0.287200	1.000	Matousek 2007
Clitllata	Hydrobiidae	0.257	0.217	Eckblad 1970
Gastropoda	Cincinnatia	0.257	0.217	Eckblad 1970
	Physidae	0.257	0.217	Eckblad 1970
	Planorbidae	0.257	0.217	Eckblad 1970
	Hydracarina	0.053	2.494	Benke et al. 1999
Trombidiformes	Mysis	1.11E-06	3.678	Chipps and Bennett 2000
Mysida	Oligochaeta	0.287200	1.000	Matousek 2007
Oligochaeta	Nematomorpha	0.287200	1.000	Matousek 2007
Nematomorpha	Ostracoda	0.053	2.494	Benke et al. 1999
Ostracoda	Sphaeriidae	0.053	2.494	Benke et al. 1999

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