# Effect of Dam Removal on Aquatic Communities in the Salmon River, New York

## Interim Report for 2010

Project #2005-0129-013

Photo provided by Jackie Harvey

Submitted to the National Fish and Wildlife Foundation

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Cooper Environmental Research

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By

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## Executive Summary

Dam removal has become a major part of restoring aquatic communities to an approximation of the pre-disturbance condition. Phase 1 of this study (2002–2004) was designed to collect data on the aquatic community (defined here as sediment, macroinvertebrates, fish, unionid mussels, and aquatic plants) of the Salmon and Little Salmon rivers to evaluate the effects of removing the Fort Covington Dam. Additional fish data was obtained in 2008 and mussel data from 2005–2009.

The dam was removed in 2009, nearly 100 years after construction, and opened about 22 km of river to migrating fishes. The opening of the dam lowered the reservoir water level by 47 cm over a 25-hr period and increased the water velocity from three to five times the previous rate, particularly between transects 3 and 5. This resulted in scouring of sediment, primarily sand, from just downstream of transect 7 through the former reservoir. The sediment was deposited from transect 3 downstream to transect 1 reaching a depth of 3 m at transects 1 and 2. The amount of redistributed sand was estimated to be 42,480 $m^3$.

Alteration of the habitats did not cause major changes in any of the macroinvertebrate indices although scouring of sand away from rocks on the east side of transect 2 and west side of transect 4 in 2010 resulted in an increase in mayfly and caddisfly families, most likely due to an increase in habitat complexity. Transect 4 also showed an increase in Total Families and decreased Percent Dominants and Percent Chironomids, all of which suggest an increase in diversity. The macroinvertebrate assemblage within the former reservoir remains much as it was prior to dam removal. Glide transects were dominated by midges (Chironomidae) while caddisflies, mayflies, riffle beetles, and midges were more abundant in riffle transects.

The lowering of the water level in the reservoir stranded, and subsequently killed, approximately 77% of the reservoir population of mussels in 2009, and this was reflected in a decline in mussel density at transect 5 in 2010. Deposition of sand in downstream areas buried an unknown number of mussels including two species, *Lampsilis ovata* and *Lampsilis cariosa*, considered to be in greatest conservation need by the New York Heritage Program. The abundance of these two species was estimated from shell middens and perhaps represented a dense mussel bed that is now covered by 3 m of sand.

Total scores for the fish IBI declined in the Salmon River from 48 in 2002–2004 to 38 in 2010. The lower scores resulted from higher percentages of dominant species, omnivores, and insectivores. The total score for the Little Salmon River has varied between 40 and 44 for the three sampling periods of 2002–2004, 2008, and 2010. The IBI score was 'good' for the Salmon River and 'very good' for the Little Salmon River.

Eastern sand darter (a threatened species in New York) was sixth in relative abundance and was collected in the main channel of the Salmon River and in the Little Salmon River. Eastern sand darter was not collected at Lewis Marina where it was abundant in 2004, perhaps due to increased silt and algae covering the former sand habitat.

The abundance and distribution of aquatic plants did not change substantially since the 1930s in the Little Salmon River but nearly all submersed plants were either removed or buried in the Salmon River. The more abundant plant genera in the Little Salmon River were *Potamogeton*, *Elodea*, and *Vallisneria*. These plants provided the major nursery habitat for fish larvae and juveniles. The distribution of the exotic flowering rush (*Butomus umbellatus*) was similar to that in pre-dam removal years but the European frogbit *Hydrocharis morsus-ranae*, found in 2004, was no longer present.

## Introduction

Dams have altered the natural cycle of water flow, sediment transport, and water temperature regimes in many streams in the United States (Ligon et al. 1995). The recent emphasis on river restoration is a response to correct the effects of dams on aquatic communities and is a practical approach to dam management (Orr et al. 2006). Changes in land use and instream morphometry due to dams were frequently deleterious to the stream as well as costly in lost property (Schroeder and Savonen 1997). Dams have slowed the range expansion of introduced species, such as carp and round goby, and the introduction of disease vectors, such as viral hemorrhagic septicemia (Cooper 2006). Reproductive success of sea lamprey in Lake Ontario tributaries has been reduced by the presence of dams (Christie 1974).

More than 450 dams have been removed in the US but less than 5% of these have included published ecological studies (Hart et al. 2002). The paucity of dam removal studies was due to lack of funding and a perceived feeling of urgency (Bednarek 2001) perhaps intensified by the potential catastrophic failure of dams (Evans et al. 2000). The pace of dam removal is increasing and has included assessments of the aquatic community such as fish (Kanehl et al. 1997; Catalano and Bozek 2007), macroinvertebrates (Stanley et al. 2002; Thomson et al. 2005), and water chemistry (Velinsky et al. 2006).

There are nearly 3000 dams in New York State, primarily in the Susquehanna and Hudson River drainages. The majority of these dams are small, run-of-river dams that do not affect moderate or high flow in downstream reaches (Heinz Center 2002). These run-of-river dams may have less effect since the reservoir area is limited, and the alteration of the flow regime is restricted to low-flow periods which affect only the pool area upstream of the dam. The pool areas formed behind the dams are generally lower in species diversity (Stanley et al. 2002) since there are fewer habitat types.

The Salmon River drainage basin extends from the northwestern part of the Adirondack Park to the international border with Quebec, Canada (Figure 1), and covers 2838 $km^2$ with 1476 km of stream (NYSDEC 1999). There are five dams remaining on the Salmon River and two dams on the Little Salmon River; these are a mixture of recreational, hydropower, and abandoned mill dams.

The Salmon River headwaters emerge near Elbow Ponds (north of Loon Lake) at an elevation of 548 m. The Little Salmon River headwaters arise near Twin Ponds at an elevation of 427 m. Both rivers have a steep gradient (approximately 11 m/rkm) until they reach the study area where the gradient ranges between 0.6 to 1.0 m/rkm. The rivers are 4th-order in the study area.

The objectives of this study were to determine the effect of removing the Fort Covington Dam by comparing the pre-dam removal characteristics of the aquatic community (2002–2004) to those in the post-dam removal period in 2010.

## Study area

The Fort Covington dam was located on the first riffle of the Salmon River, approximately 8 km from the St. Lawrence River. The original dam was built in the late 1800s as a wood crib structure and was damaged in a freshet in 1912. It was rebuilt in 1913 as a concrete run-of-river dam that was used for hydroelectricity and as a grist mill. The dam was removed in 2009. Characteristics of the dam are described in the baseline data report (Cooper et al. 2004), as well as a discussion of the geology, land use, and elevation characteristics of the study area.

*Flow characteristics*. Water discharge in the Salmon and Little Salmon rivers is variable and responds rapidly to inputs of precipitation. Flooding has occurred upstream of the study area and within the study area due to ice floes collecting in more narrow parts of the river. Mean daily stream flow can reach 3280 cubic feet per second (cfs; 93 $m^3$/sec) in the Salmon River (Figure 3) and 2620 cfs (74.2 $m^3$/sec) in the Little Salmon River.

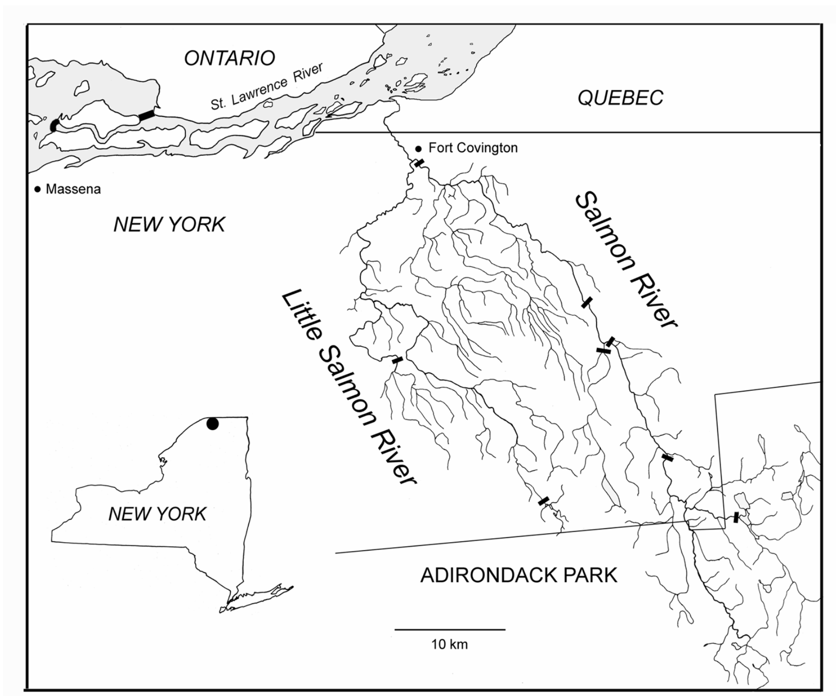



Figure 1. Location map of the Salmon and Little Salmon rivers, Franklin County, New York. The black boxes in the St. Lawrence River are the Robert Moses-Saunders Power Dam (right) and Long Sault Dam (left). The Fort Covington Dam was the most downstream dam in the Salmon River; the other boxes represent dams upstream from the study area.

## Methods

*Sampling design*. Fifteen transects were established, nine in the Salmon River and six in the Little Salmon River, divided between riffles (transects 3, 7, 9, 13, 15) and glides (transects 1, 2, 4-6, 8, 10-12, 14). One additional riffle, designated as 67, was sampled in 2010 after it was exposed following the draining of the reservoir (Figure 3).

Transects were paired across rivers (riffle to riffle, glide to glide) with the exception of transects 1 through 3 — these three transects did not have analogous reaches in the Little Salmon River — and the riffle transect 67. Each transect was subdivided into east, center, and west areas (when facing north). The sampling period was from April to November, 2010.

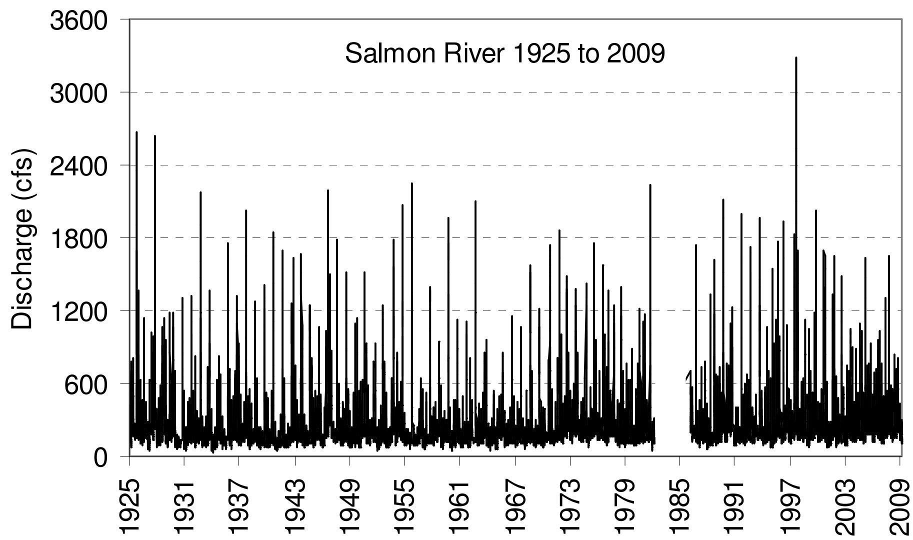



Figure 2. Mean daily flow of the Salmon River from 1925 to 2009 as measured by the USGS gage at Chasm Falls (gage 04270000). There is a break in the record from 1982 to 1986.

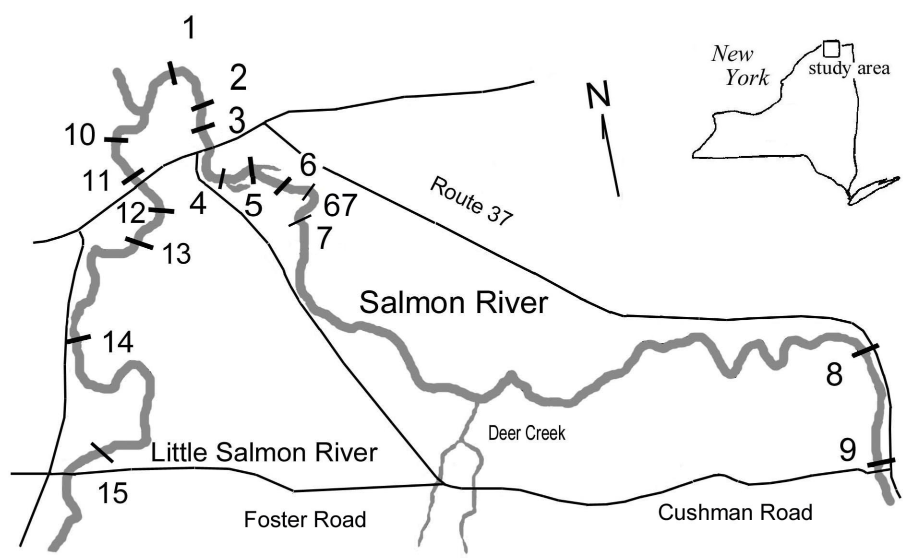



Figure 3. Location of transects sampled for sediments, macroinvertebrates, fish, and aquatic plants. Water level loggers were located near transects 3, 6, and 10. Water temperature recorders were located near transects 3, 6, 7, 9, 10, 13, and 15. The barometric pressure logger was located near transect 10. Transect 67 was a riffle that was exposed with the drawdown of the reservoir.

Table 1. Maximum width and depth of transects: width and depth in the Salmon River has two values corresponding to pre- and post-dam removal conditions. Transect 67 was a glide prior to dam removal and a riffle after dam removal.

| Salmon River | | | | | Little Salmon River | | |
|---|---|---|---|---|---|---|---|
| | Width (m) | | Depth (m) | | | | |
| Transect | Pre- | Post- | Pre- | Post- | Transect | Width (m) | Depth (m) |
| 1, glide | 25 | 25 | 3 | 0.5 | 10, glide | 17 | 3 |
| 2, glide | 50 | 50 | 2 | 0.5 | 11, glide | 34 | 1 |
| 3, riffle | 34 | 32 | 0.5 | 0.5 | 12, glide | 21 | 2 |
| 4, glide | 25 | 22 | 3 | 0.5 | 13, riffle | 13 | 0.5 |
| 5, glide | 30 | 27 | 2 | 0.2 | 14, glide | 38 | 1 |
| 6, glide | 50 | 48 | 1 | 0.5 | 15, riffle | 30 | 0.5 |
| 67, glide, riffle | 25 | 25 | 2 | 0.5 | | | |
| 7, riffle | 70 | 70 | 0.5 | 0.5 | | | |
| 8, glide | 34 | 34 | 1 | 1 | | | |
| 9, riffle | 40 | 40 | 0.5 | 0.5 | | | |

*Sediment*. Three grabs taken with a 6 X 6 ponar dredge (0.02 $m^2$) were composited to make one sample from each subdivision (east, center, and west) in each glide transect (N = 30). Grain size was not determined in the riffle areas but was assumed to approximate that found in the glide transects with the exception of cobbles, boulders, and bedrock. Samples were stored at 3.8º C until analyzed. Grain size was characterized into only three size categories in 2002 (sand, silt, and clay; Cooper et al. 2004) so archived samples were re-screened in the same manner as in the present study: dry material was screened through five mesh sizes (6 mm, 1 mm, 0.5 mm, 0.125 mm, and 0.062 mm). Silt (0.0039 to 0.031 mm) and clay (0.002 mm) proportions were determined by the dispersal method (Folk 1980) and all fractions reported as the percent dry weight of the original sample. Sorting was determined by the inclusive graphic standard deviation method of Folk (1980), which relates the cumulative percent by weight of sediment fractions to phi values at four percentages (84%, 16%, 95%, and 5%):

$$\frac{\Phi 84- \Phi 16}{4} + \frac{\Phi 95- \Phi 5}{6.6}$$

The solution of this equation results in an estimate of the average particle size encompassing 95% of the size distribution. Porosity was determined by dividing bulk density (sediment dry weight divided by volume) by 2.65, the density of quartz, which is the predominant mineral. Estimates of embeddedness (Barbour et al. 1999) were made at five locations in the center of each riffle transect.

*Water chemistry*. Water temperature was recorded at 1 hr intervals using Onset thermographs or level loggers at transects 3, 6, 7, 9, 10, 13, and 15. Monthly grab samples were used to estimate water temperature and dissolved oxygen (Hach sension6), pH (ecotestr ph2), total dissolved solids (TDSTestr3), alkalinity (Lamotte titrator), nitrogen ammonia, chloride, nitrate, sulfate, and turbidity (Hach colorimeter) at transects 2, 7, 9, 10, 13, and 15. Onset level loggers recorded water level changes at 1 hr intervals at transects 3, 6, and 10 (Figure 3). A separate level logger was used to determine barometric pressure to correct the measured values of water level and to record air temperature at transect 10.

*Water velocity*. Estimates of water velocity were made monthly from April through October using a Price-type "mini" current meter at the center of transects 3, 7, 13, and 15. The bucket wheel was set at 40% of the water depth and recorded for 30 seconds. Water velocity was also recorded at glide transects but was generally too slow to be measured with the meter. Estimates at glide transects were made by timing a neutrally buoyant ball over a specific distance.

*Macroinvertebrates*. Macroinvertebrates were collected with a rectangular kick net (0.26 $m^2$; 500µ mesh) in the riffles and ponar dredge (0.023 $m^2$) in the glides. Four kick net samples were composited from the east, center, and west areas of each riffle transect and four ponar dredge samples were composited from the east, center, and west areas of each glide transect resulting in three samples from each transect. Each sample was washed through a 500μ mesh screen before compositing. All samples were preserved with 10% buffered formalin and returned to the laboratory for sorting and counting. No sub-sampling was used. Organisms were identified to the family level except for oligochaetes, nematodes, leeches, diptera pupae, diptera adults, and water mites. Mussels were identified to species, measured in the field, and returned to the water.

Six indices were calculated for macroinvertebrates using 65 families out of the 76 taxa identified: *EPT* – the number of families in Ephemeroptera, Plecoptera, and Trichoptera; *richness* – the total number of families; *dominance* – sum of percentages of the five more abundant families out of the total number of individuals; *percent Chironomidae* – the percentage of chironomids out of the total number of individuals; *Family Biotic Index* (formerly Hilsenhoff Biotic Index) – family tolerance value (Barbour et al. 1999) multiplied by abundance and divided by total number collected; and the *Percent Model Affinity* – a comparison of the percent similarity between seven taxonomic groups in the samples to the percent of the same taxonomic groups in a 'model' community (Novak and Bode 1992). The methods for Percent Model Affinity differed from that specified by Novak and Bode (1992) in that all organisms were used in the calculations rather than a 100-organism subsample. The Biological Assessment Profile (Bode et al. 1996) was not used in the present study as the macroinvertebrates were not identified to species.

*Unionid mussels*. Systematic sampling (Strayer and Smith 2003) with three random starts was used at transects 3, 5, 7, 8, 9, 12, 13, 15, Lewis Marina, and Deer Creek. Double sampling (visual/tactile plus excavation) was used at transects 5, 8, 12, and Lewis Marina. Each of three reaches across the river had 10 1-m$^2$ quadrats with each quadrat subdivided into four 0.25 m$^2$ areas. Reaches in Deer Creek were directed upstream due to the narrow channel, and reaches at Lewis Marina were located only on the west side as the east side was too deep. Each sub-quadrat was searched with visual and tactile methods, and 20% of the sub-quadrats were excavated. The excavated material was sifted through a 6-mm screen. An underwater viewing scope was used to facilitate finding mussels. Shorelines were searched for muskrat middens and empty shells were identified and measured. Population estimates were made for each transect.

*Fish*. Collections of fish were made using hoop nets (1.2 m hoop, 6 m wings, 12 mm bar mesh) in May and September, and a 3 m X 1 m bag seine (3 mm mesh) in June and September, in various locations representing the various habitat types. Hoop nets were set overnight and fished in the same order as deployed. The total fishing time for each net was recorded. Seining was done in representative habitats with each haul distance recorded. All fish were identified to species, and the majority was measured for total length (mm) and wet weight (g) in the field and returned alive to the collection area. Some minnows were preserved in 10% buffered formalin to verify their identification.

An index of biotic integrity (IBI) was constructed for fish for each river based on 12 metrics following Daniels et al. (2002). These metrics were 1) total number of fish species (excluding carp, American eel, and stocked trout); 2) number of benthic insectivores; 3) number of water column species (excluding smallmouth and largemouth bass); 4) number of terete minnow species; 5) dominant species – 3 more abundant species as a percentage of the total number of species; 6) percentage of total individuals that were white sucker; 7) percentage of total individuals that were omnivores; 8) percentage of total individuals that were insectivores; 9) percentage of total individuals that were top carnivores – largemouth bass, smallmouth bass, northern pike, longnose gar, and walleye; 10) density as number/m$^2$ per river (these values were determined only from seining data since trap net data does not account for an area that is fished); 11) percentage of species that had two age classes (estimated from length frequency plots); and 12) the percentage of individuals that had tumors, lesions, or parasites.

Each metric was then scored from 1 to 5 with 5 representing the least effect. The index was the sum of scores for the metrics.

*Statistical methods*. All statistical comparisons were made on untransformed variables using the General Linear Model in SAS (version 8.2, SAS 2001). Regression was used to explore the length–weight relationship of fish and the relationship of surface counts to total counts of mussels. Bonferroni t-tests were used to examine sorting and porosity for 2010 data as well as comparing pre- and post-dam removal data. Correlation was used to compare water level to barometric pressure. Mussel distribution between riffles and glides was compared using the non-parametric Kolmogorov-Smirnov statistic. Macroinvertebrate density in three areas (east center, and west) in riffles and glides was compared using least-squares means as were comparisons of macroinvertebrate density between transects in the former reservoir and other transects, and mussel density by transect. Least-squares means used transect or sampling period as a covariate.

*Aquatic plants*. A qualitative survey was made of aquatic plants within the study area in August. Plants were identified to genus, ranked by abundance, and locations noted. The primary objective was to locate areas that could function as spawning areas for fish.

## Results

*Sediment*. Sand comprised the greatest percentage of any fraction at all glide transects (Figure 4). The center of each river was composed primarily of fine to medium sand (0.062 to 0.5 mm) with fine sand and silt along the river banks. Clay was the least abundant fraction in both rivers averaging less than 1% with the exception of the west side of transect 10 where it was 7%. The west side of the Salmon River had a greater percentage of silt than the east side, with the exception of transect 1. This was true of the Little Salmon River at transects 10 and 14 but not at 11 and 12, although the differences at the east and west sides were small. Several large sediment deposits were exposed in the center of the former reservoir, which had greater percentages of clay (7%) and silt (11%), but less sand (81%), than adjacent areas of the reservoir; the average fractions for transects 5 and 6 were 0.3% (clay), 4.3% (silt), and 95% (sand).

Comparison of the archived sediment samples from 2002 to the sediment samples from 2010 revealed that fine to medium sand had increased in the slower velocity areas of the glide transects in the former reservoir as well as downstream to transect 1 (Appendix Figure 1). These sand deposits have buried much of the silt sediment along the river banks.

*Sorting and Porosity*. Sediments were moderately to poorly-sorted and had relatively high porosity in 2010. The east bank of the Salmon River was less sorted than the center or west bank but there was no significant difference by location (east, center, west: $F = 3.13$, $P = 0.057$). Porosity in the Salmon River was significantly greater in the center than in the east or west sides ($F = 9.21$, $P < 0.007$) but there was no significant difference in the Little Salmon River ($F = 2.60$, $P = 0.098$).

The mean sorting value increased in the east and west banks of the Salmon River from 2002 to 2010 (Table 2) but not significantly so ($F = 1.52$, $P = 0.23$). There was no significant difference in mean sorting values between the three areas in the Little Salmon River ($F = 0.30$, $P = 0.74$) and there was little change in sorting by year ($F = 0.05$, $P = 0.83$). There were no significant differences in porosity by year in either river (Salmon River, $F = 0.08$, $P = 0.78$; Little Salmon River, $F = 1.14$, $P = 0.296$).

Embeddedness was 20% at transects 13, 15, and 67, 30% at transect 7, 40% at transect 9, and 60% at transect 3 in July, 2010. There was no change in embeddedness at

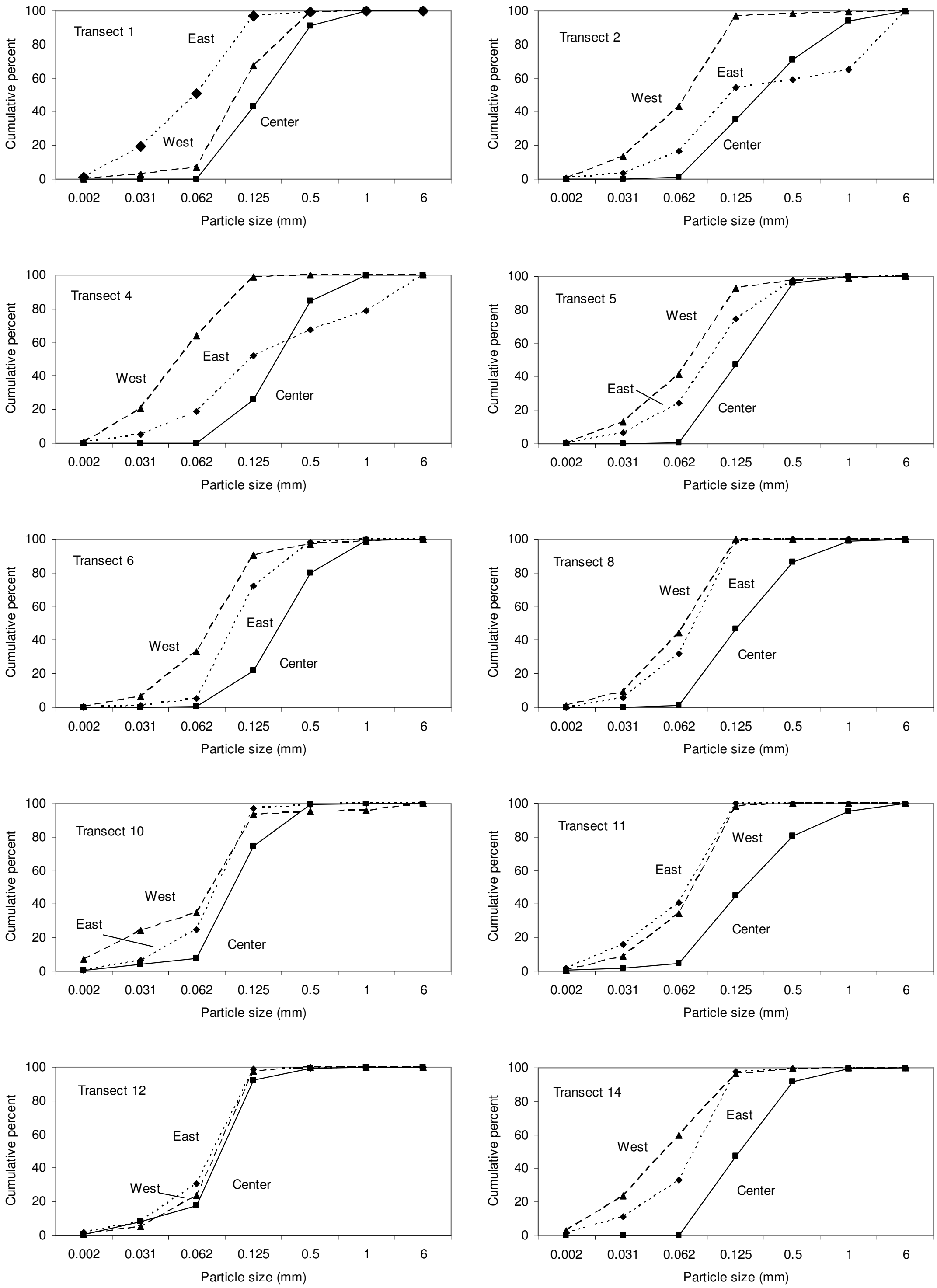



Figure 4. Cumulative percent of sediment fractions from the glide transects. Particle sizes were: 0.002 = clay, 0.031 = silt, 0.062 to 0.125 = very fine sand, >0.125 to 0.5 = fine and medium sand, >0.5 to 1.0 = coarse sand, and 6 = pebble.

Table 2. Values for sorting and porosity (mean ± one SE) in three areas of the glide transects. East, center, and west refer to the locations in the river when facing north. N = 18 for the Salmon River; N = 12 for the Little Salmon River for each year.

| | Salmon River | | | Little Salmon River | | |
|---|---|---|---|---|---|---|
| | East | Center | West | East | Center | West |
| Sorting 2002 | 0.99 ± 0.19 | 0.84 ± 0.08 | 0.68 ± 0.09 | 1.09 ± 0.08 | 0.93 ± 0.13 | 0.87 ± 0.08 |
| Sorting 2010 | 1.11 ± 0.10 | 0.80 ± 0.05 | 0.95 ± 0.06 | 0.91 ± 0.02 | 0.95 ± 0.08 | 0.99 ± 0.09 |
| Porosity 2002 | 0.49 ± 0.03 | 0.54 ± 0.02 | 0.48 ± 0.04 | 0.52 ± 0.04 | 0.52 ± 0.04 | 0.50 ± 0.03 |
| Porosity 2010 | 0.45 ± 0.03 | 0.61 ± 0.01 | 0.42 ± 0.03 | 0.42 ± 0.03 | 0.50 ± 0.02 | 0.43 ± 0.04 |

transects 9, 13, and 15 from 2002. Embeddedness at transect 7 in 2010 declined 10% from 2002 but embeddedness had increased at transect 3 from 20% in 2002 to 60% in July, 2010, due to movement of sediment following the opening of the dam, but after flooding in October, 2010, embeddedness was reduced at transect 3 to 40%. The substrate at each riffle transect was cobble (64 to 256 mm), boulder (>256 mm) and bedrock mixed with sand.

*Discharge and water level*. Discharge values were not available from the Salmon River for 2010 but comparisons of historical discharge records from the USGS gage at Chasm Falls (Salmon River; gage 04270000), and the USGS gage at Bombay (Little Salmon River; gage 04270200), showed that the response to precipitation was similar in the two rivers, although the Salmon River discharged at a greater rate. Recorded discharge of the Little Salmon River in 2010 (Figure 5) was similar to the records at the two monitoring locations (Figure 6) and showed rapid, although brief, water level changes. Three rain events in September (20 cm) and early October (10 cm; NOAA National Climate Data Center) resulted in high water in both rivers. The third high water event removed the water level logger at transect 3, and might have covered the water temperature logger with sand at transect 9 (neither logger has been located). Changes in water level were correlated weakly with changes in barometric pressure (Pearson coefficient = -0.10, $P$ <0.0001).

*Water velocity*. Measurements were taken at discharge rates between 1.4 and 5.2 $m^3/s$ but the measured velocity was not sensitive to discharge. A regression of discharge on measured velocity resulted in an $r^2 = 0.32$. Mean velocity was greatest at transect 7 and least in Deer Creek (Table 3). Water velocity at the center of the reservoir (transect 5) increased from 0.08 m/s in 2008 to 0.34 m/s after the dam was removed in 2009 (both measurements made at a discharge of 0.8 $m^3/s$). Water velocity at transects 3 and 4 would be greater than at transect 5 with increased gradient.

Table 3. Mean measured water velocity at selected transects in 2010. Means are ranked from fastest to slowest from left. N is the number of estimations.

| | Transect | | | | | | |
|---|---|---|---|---|---|---|---|
| | 7 | 9 | 3 | 15 | 13 | 2 | Deer Creek |
| Mean velocity (m/s ± 1 SE) | 0.93 ± 0.05 | 0.79 ± 0.11 | 0.78 ± 0.05 | 0.69 ± 0.04 | 0.54 ± 0.06 | 0.5 ± 0.009 | 0.44 ± 0.10 |
| N | 16 | 13 | 17 | 19 | 16 | 4 | 4 |

*Water temperature*. Ice cover formed in early December, 2009, and broke up in early March, 2010, in the Little Salmon River (Figure 5), and then reformed in early December. Water temperature increased at about 5 ºC per month in both rivers reaching a maximum of 32 ºC in early July in the Little Salmon River, and 29 ºC in the Salmon River. The seasonal temperature profile (Figure 6) was similar in both rivers although the Salmon River was about 1 ºC colder than the Little Salmon River. Water temperature at Lewis Marina averaged 0.5 ºC colder than at transects 12 and 15.

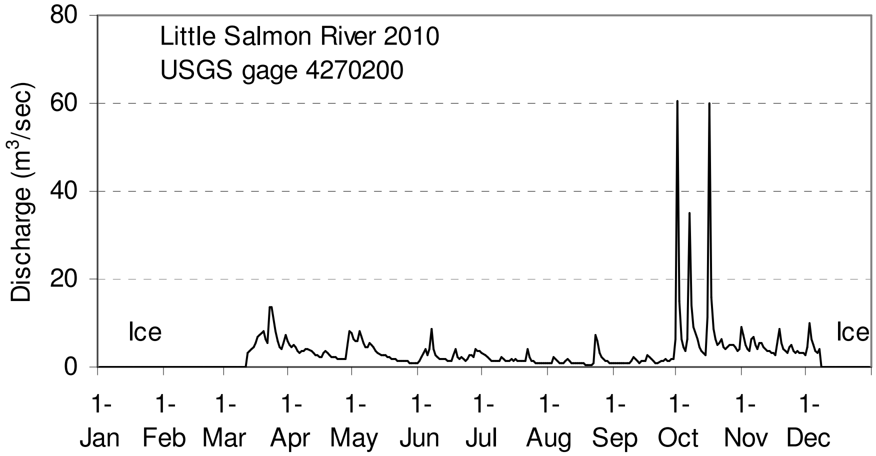
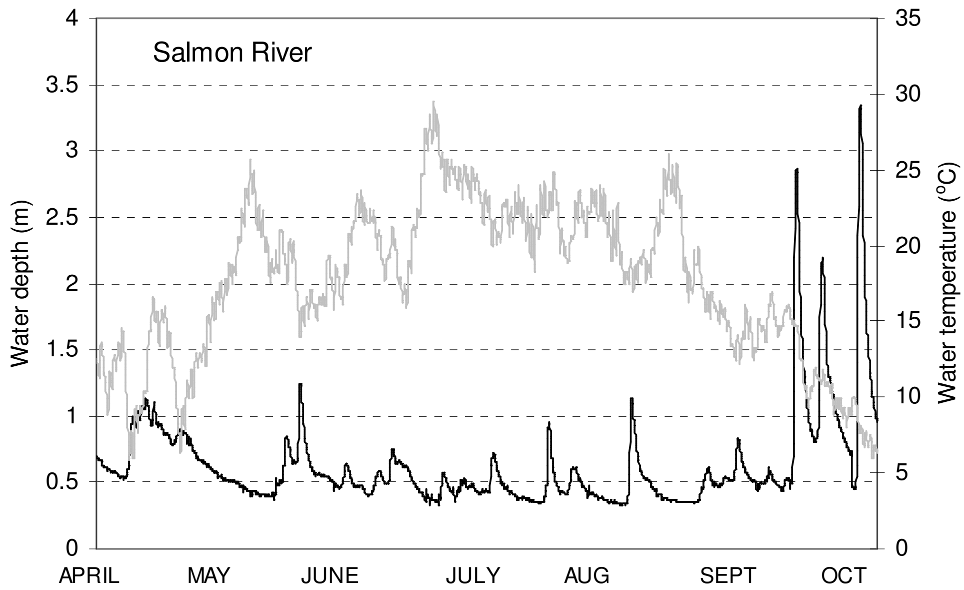



Figure 5. Mean daily discharge of the Little Salmon River during 2010 measured at Bombay (USGS gage 04270200). The gaging station is approximately 7 river km upstream of the study area.

Dissolved oxygen ranged from 6.5 to 12.3 mg/L (68% to 135% saturation) from May through October 2010, and measured saturation was 83% or greater except for one sample at transect 7 (68%, 15 June) in the Salmon River. Dissolved oxygen and percent saturation were similar between the two rivers. pH ranged from 7.3 to 8.6 during the study period and mean pH was greatest in August in both rivers at 8.3.

Total dissolved solids (TDS) ranged from 90 to 220 µS and were generally greater in the Little Salmon River. TDS values increased by month and were greater in August and September. Alkalinity ranged from 62 to 104 mg/L and was greater in the Little Salmon River. Alkalinity did not show any seasonal trends. Total alkalinity was from bicarbonates as phenophthalein titrations were always zero. Chloride and sulfate concentrations were similar in both rivers although chloride was greater in May and June at transect 2. Ammonia concentrations were greater near pastures (transects 7, 9, and 15) and nitrate was greatest at transect 7. Mean turbidity by transect ranged from 4.2 to 8.3 (Table 4) with the single greatest reading of 18 in July as a result of precipitation.



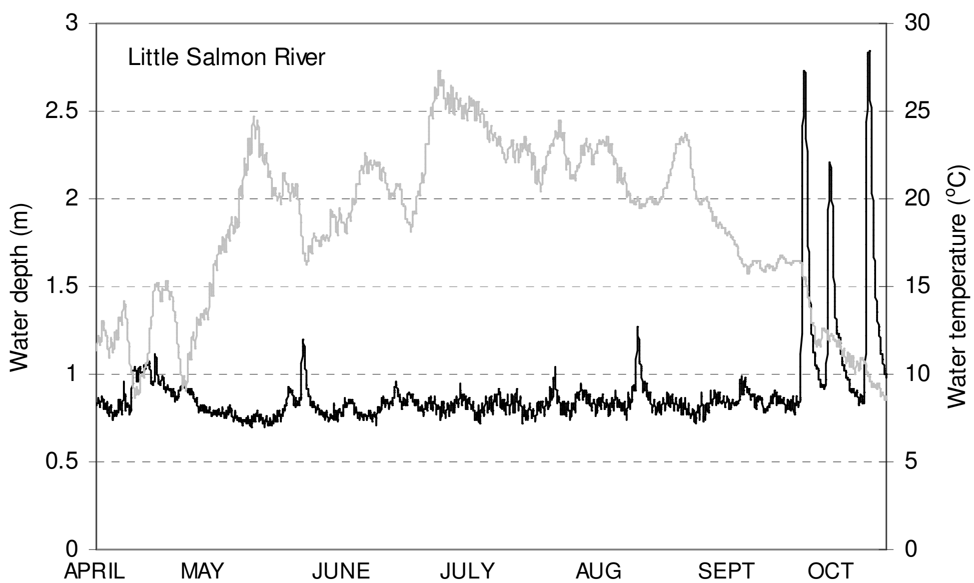




Figure 6. Water level changes recorded in 2010 in the Salmon River (transect 6) and in the Little Salmon River (Lewis Marina near transect 10). Water level values are compensated for changes in barometric pressure.

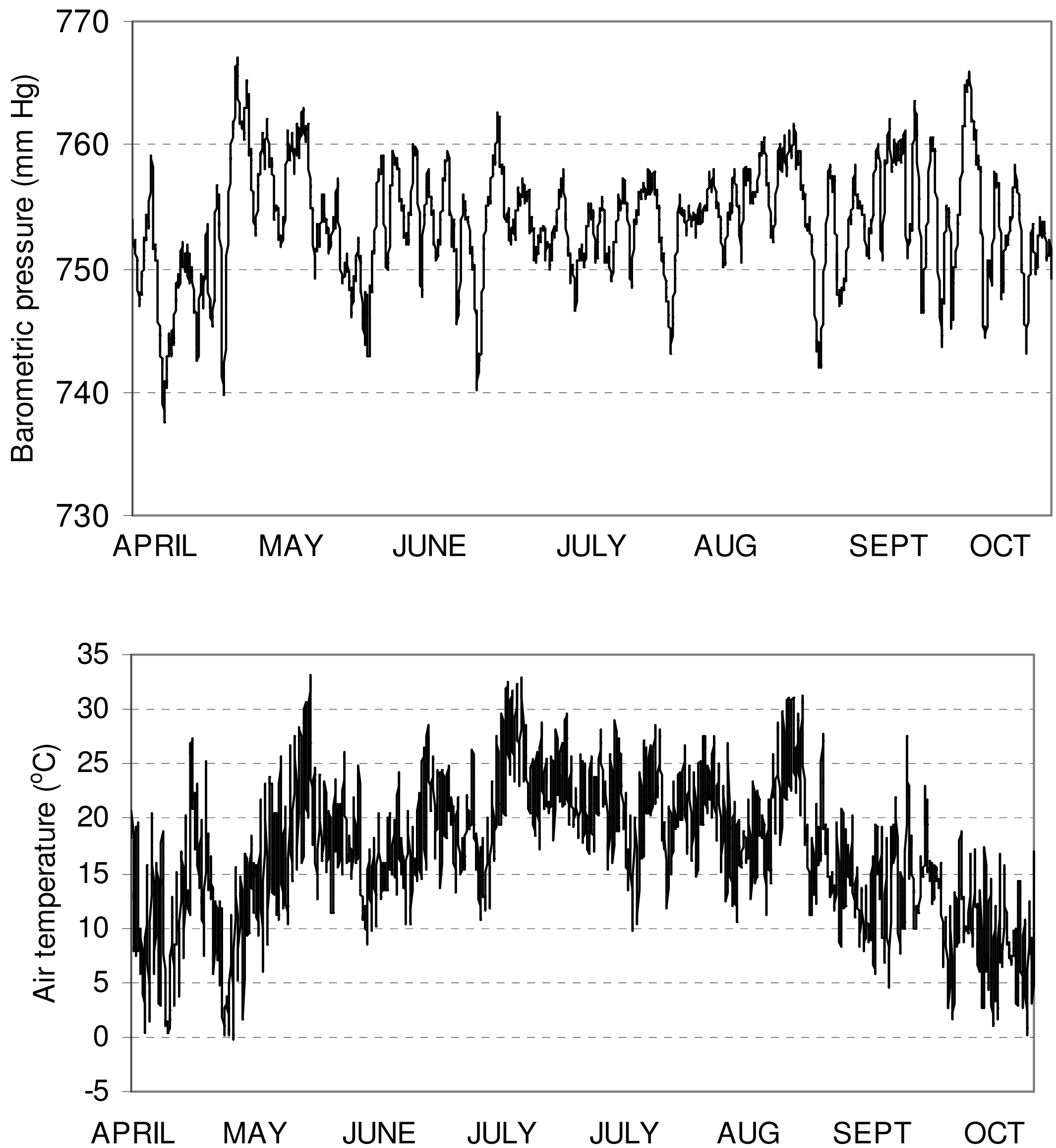



Figure 7. Barometric pressure and air temperature recorded in 2010 at Lewis Marina, near transect 10.

*Macroinvertebrates*. A total of 76 taxa was identified (not including unionid mussels) but only 65 families were used in the construction of indices. The indices used various combinations of the collected families. Those taxa that were not identified to family level or did not have known pollution tolerance values were excluded: water mite, nematode, diptera pupa and adult, leech, planaria, spider, and sponges.

Table 4. Mean concentration (± 1 SE) of water chemistry parameters taken as grab samples (N = 6) from May through October, 2010.

| | Salmon River | | | Little Salmon River | | |
|---|---|---|---|---|---|---|
| Transect | 2 | 7 | 9 | 10 | 13 | 15 |
| Dissolved oxygen (mg/L) | 10.3<br>(0.7) | 8.9<br>(0.6) | 10.5<br>(0.5) | 9.3<br>(0.9) | 9.5<br>(0.7) | 9.4<br>(0.7) |
| Percent saturation | 104.5<br>(6.7) | 92.2<br>(5.8) | 102.5<br>(4.2) | 96.1<br>(8.2) | 100.9<br>(5.5) | 94.3<br>(2.3) |
| pH | 7.8<br>(0.1) | 7.9<br>(0.1) | 7.8<br>(0.2) | 7.8<br>(0.1) | 8.0<br>(0.1) | 8.0<br>(0.1) |
| Total dissolved solids (µS) | 148.3<br>(11.9) | 141.7<br>(11.4) | 133.3<br>(9.5) | 171.7<br>(12.7) | 170.0<br>(15.3) | 165.0<br>(13.4) |
| Alkalinity (mg/L $CaCO_3$) | 80.7<br>(2.8) | 76.3<br>(3.7) | 70.0<br>(4.7) | 90.3<br>(3.7) | 88.7<br>(2.8) | 89.3<br>(3.0) |
| Chloride (mg/L, $Cl^-$) | 0.24<br>(0.07) | 0.12<br>(0.02) | 0.10<br>(0.03) | 0.11<br>(0.02) | 0.11<br>(0.02) | 0.08<br>(0.01) |
| Nitrate (mg/L, $NO_3$-N) | 0.63<br>(0.09) | 0.72<br>(0.05) | 0.67<br>(0.12) | 0.64<br>(0.11) | 0.56<br>(0.09) | 0.56<br>(0.11) |
| Sulfate (mg/L, $SO_4$) | 1.7<br>(0.2) | 1.3<br>(0.6) | 1.0<br>(0.4) | 1.8<br>(1.0) | 1.3<br>(0.8) | 1.3<br>(0.3) |
| Turbidity (FAU) | 8.3<br>(0.5) | 5.7<br>(0.4) | 7.3<br>(2.2) | 5.0<br>(0.6) | 4.2<br>(1.0) | 5.2<br>(0.8) |
| Nitrogen, Ammonia (mg/L, $NH_3$-N) | 0.01<br>(0.003) | 0.04<br>(0.02) | 0.06<br>(0.01) | 0.01<br>(0.004) | 0.02<br>(0.009) | 0.04<br>(0.02) |
| Water temperature ($^oC$) | 16.7<br>(2.6) | 17.0<br>(2.4) | 15.2<br>(2.4) | 18.5<br>(2.9) | 18.7<br>(3.1) | 17.2<br>(2.9) |

A total of 37,298 macroinvertebrate organisms were collected in 2010 for a density of 2,743 organisms/$m^2$. Chironomidae dominated the glide transects in abundance (mean = 57% of organisms collected) and abundance ranged from 36% at transect 10 to 96.5% at transect 5. Five of six riffle transects were dominated by Hydropsychidae (mean = 37%) but Baetidae were more abundant at transect 13 (39.1%). Hydropsychidae abundance ranged from 15% at transect 13 to 54% at transect 67. Chironomidae dominated all transects in mean density (537/$m^2$) followed by Hydropsychidae (250/$m^2$), oligochaetes (173/$m^2$), Baetidae (164/$m^2$), and Elmidae (116/$m^2$). Mean density of allorganisms was similar between the east and west sides of the glide transects but 72% less in the center. There was considerable variation in density in the center of the glide transects ranging from 97 organisms/$m^2$ at transects 1 and 5 to 1,247 organisms/$m^2$ at transect 11 (coefficient of variation = 103%). Mean density of all organisms was significantly less in the center of the glides (lsmeans = 0.017) than in the east areas of the glides but not different from that in the west side. Mean density in the former reservoir (transects 4, 5, and 6) was not significantly different from glide transects 1, 2, and 8 in the Salmon River (lsmeans = 0.93). A comparison of the percent of density accounted for by the 10 most abundant macroinvertebrates showed that diversity increased at transects 2 and 4 (less percent attributed to 10 more abundant families) after the dam was removed. There was no significant difference in areas of the riffles (lsmeans >0.17) and no significant difference in mean density between riffles and glides ($F$ = 1.87, $P$ = 0.18).

The number of EPT families was significantly greater ($F$ = 14.25, $P$ = 0.0005) in the riffles (mean = 6.2) than in the glides (mean = 2.4) in 2010. Transects 5 and 6 in the former reservoir, transect 8 upstream of the reservoir, and transect 11 in the Little Salmon River contained fewer EPT families than the other transects (Figure 8). The number of EPT families collected at transect 67 was similar to that collected at other riffle transects.

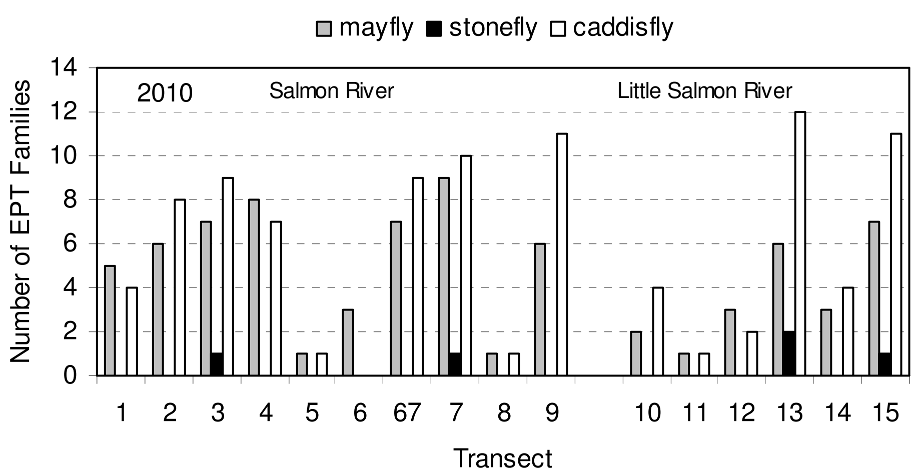



Figure 8. Number of EPT families collected in 2010. Transects 4 through 67 are in the former reservoir.

Values for Total Families were similar to the EPT values with lower scores in the former reservoir and transects 8 and 11. Riffles supported more families than glides did but transects 2, 4, and 10 were similar to riffle transect 3. The Family Biotic Index (FBI) was close to a mirror image of the Total Families index where a greater number of families with lower pollution tolerance values resulted in lower FBI values (Figure 9).

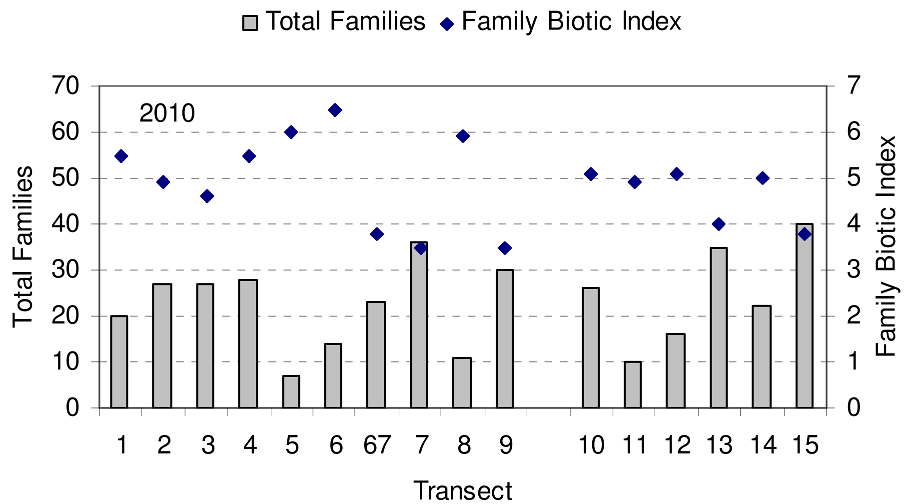



Figure 9. Total Families Index and Family Biotic Index for the Salmon (transects 1–9, 67) and Little Salmon (10–15) rivers in 2010.

Lower values occurred at transects in the former reservoir and at transects 8 and 11 while riffles had higher values.

Percent Dominants had an index value of 80% or greater in 11 of the 16 transects and two of the three transects in the former reservoir (5 and 6) had values of over 99%. Transect 4, a former reservoir transect, had a value similar to the riffle transects (Figure 10).

Percent Chironomidae was greater at transects 5, 8, and 1 (mean = 82%) and least in the riffle transects (mean = 7.9%). Transect 67 was similar to transects 13 and 5.

Percent Model Affinity was developed from riffle samples (Novak and Bode 1992) but has been applied to glide samples here as well as riffle samples. Glide transects

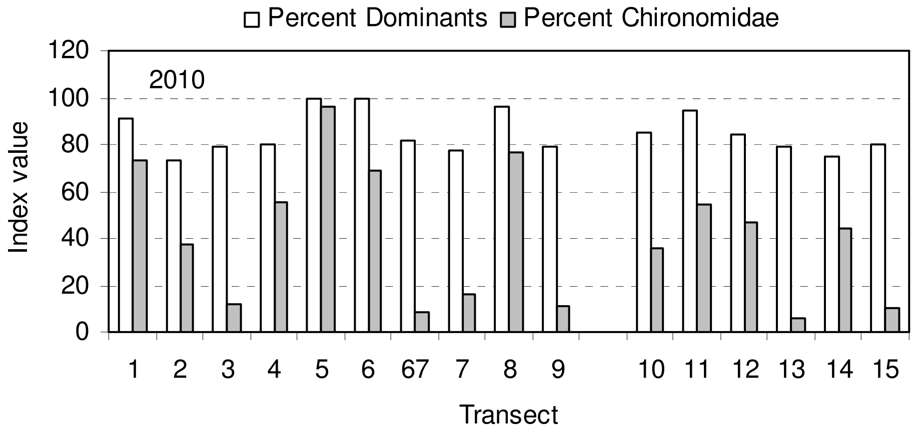



Figure 10. Percent Dominants index and Percent Chironomidae Index for the Salmon (transects 1–9, 67) and Little Salmon (transects 10–15) rivers in 2010.

1, 2, 4, 10 and 14 were similar to riffle transects 3, 67, and 9. Glide transects 5, 6, and 8 had lower model affinity than all other transects (Figure 11).

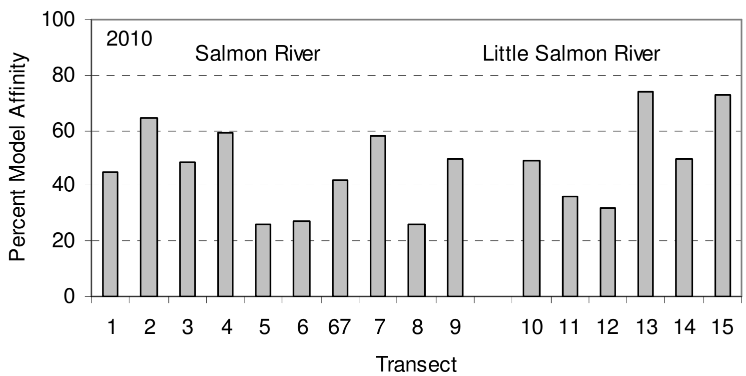



Figure 11. Percent Model Affinity for the Salmon (transects 1–9, and 67) and Little Salmon (10–15) rivers in 2010.

Mayfly families increased at transects 2, 4, and 7, decreased at transects 5, 8, 10, and 11, and were similar in abundance at all other transects (Figure 12) compared to the pre-dam removal collections. Stonefly families generally decreased except for transect 13 but few stonefly families were collected at any transect. Caddisfly families showed a similar response as mayfly families, increasing at 10 of the 15 transects, and particularly at transects 2 and 4.

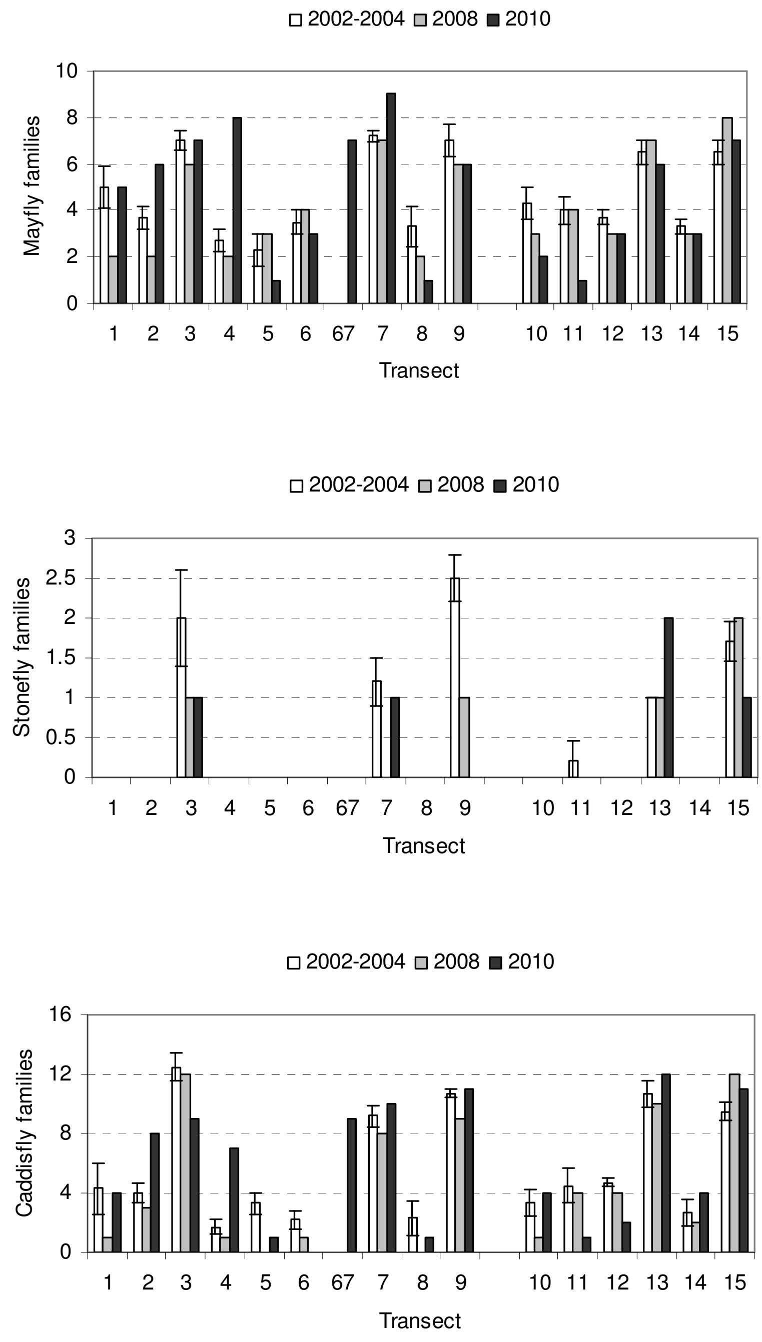



Figure 12 . Number of mayfly, stonefly, and caddisfly families by transect. Four sampling months occurred from 2002 to 2004 thus these values are means (± 1 SE); 2008 and 2010 are family number for one sampling month in each year. Transects 1–9 are in the Salmon River and 10–15 are in the Little Salmon River.

The six sampling periods (2002 through 2010) included a cumulative 113 identified taxa (range 68–88, mean = 78.5, 1 SE = 2.8) but only 65 families were used in the construction of indices. The number of EPT families increased at transects 1, 2, 4, 7, 13 and 14, were similar at transects 10 and 15, and decreased at transects 3, 5, 6, 8, 11, and 12 from the pre-dam removal period to the post-dam removal collection in 2010 (Figure 13), however, only the increase at transect 4 was significant ($F$ = 9.15, $P$ = 0.008).

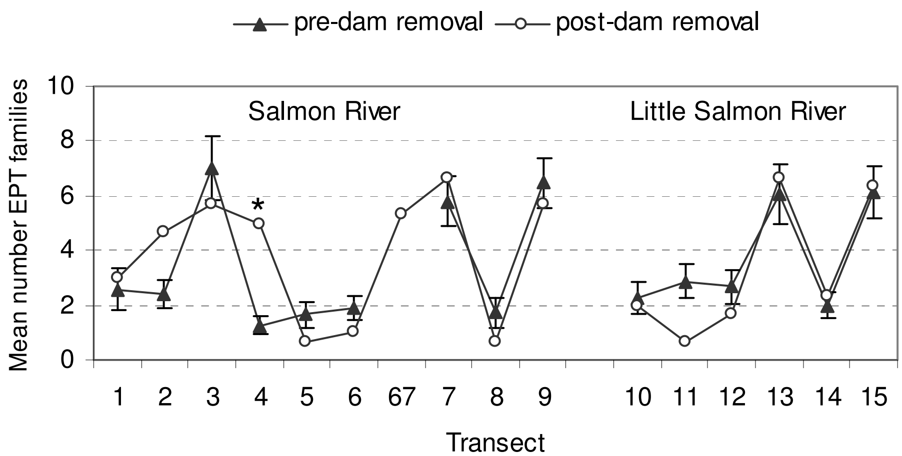



Figure 13. Number of EPT families from the pre-dam removal period (mean ± 1 SE) and the number collected in post-dam removal (2010). An asterisk indicates the only statistical difference by period (lsmeans = 0.0008).

The four indices of Total Families, Family Biotic Index, Percent Dominants, and Percent Chironomidae showed varied responses in 2010 compared to previous collections. Total Families was similar at transects 6, 10, and 14, increased at transects 2, 4, 7, and 15, and decreased at all remaining transects. Family Biotic Index values were similar at transects 5, 8, 9, and 13, increased at transects 3 and 6, and decreased at all remaining transects. Percent Dominants values increased at transect 3, decreased at transects 2, 4, 14, and 15, and were similar at the remaining transects. Percent Chironomidae was similar at 11 of the 15 transects and decreased at the remaining transects.

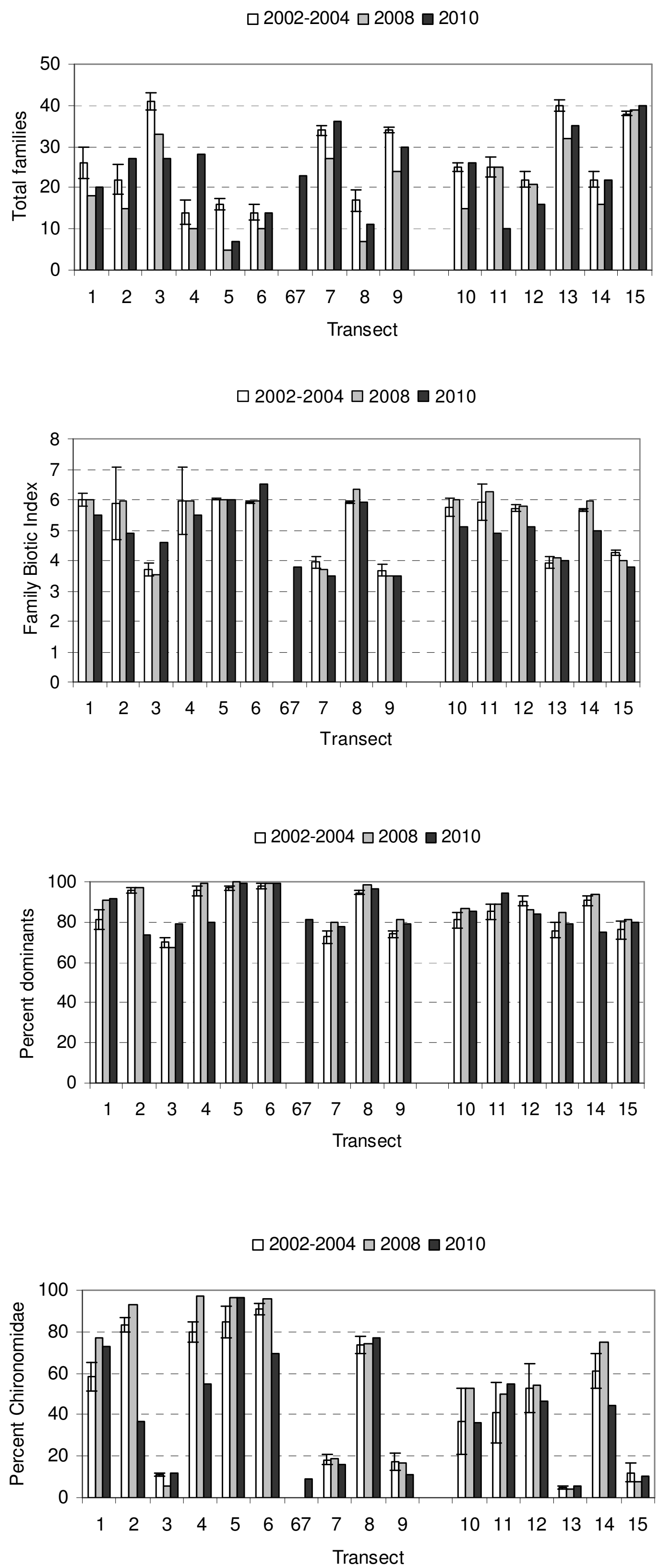



Figure 14. Macroinvertebrate indices for each transect and river calculated from 65 families. Four sampling months occurred from 2002 to 2004 thus these values are means (± 1 SE); the other years are represented by total number in one sampling month.

Changes in the indices for pre- and post-dam removal for each transect were examined with least-squares means using the six sampling periods (from 2002 through 2010) as a covariate. Transects generally clustered into three groups: riffle transects formed one group, transects in the former reservoir (and transect 8) formed a second group, and the remainder of transects formed a third group.

The number of EPT families by transect were not significantly different among sampling periods ($F = 0.06$, $P = 0.76$) but the riffle transects had significantly more EPT families than the glide transects ($P$ <0.003 with Bonferroni correction; Figure 15).

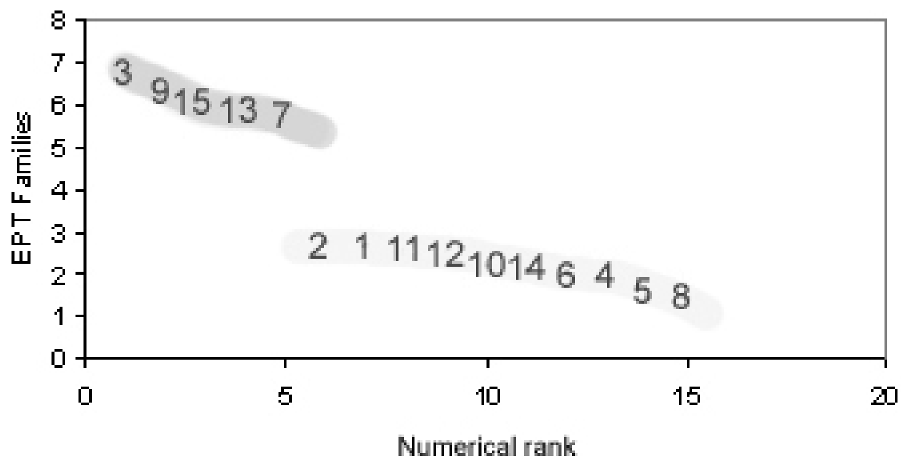



Figure 15. Numerically ranked value of EPT Families at transects for six sampling periods from 2002 to 2010. Numbers refer to transects. Transects that do not share a common color shade are statistically different.

Transects segregated into three groups for Total Families (Figure 16): riffle transects had statistically greater number of Total Families ($P$ <0.003) than did Salmon River glide transects 1 and 2, glide transects in Little Salmon River (10–14), and transects of the former reservoir (including transect 8). There was no significant difference in pre- and post-dam removal in Total Families ($F = 1.73$, $P = 0.14$). Transects segregated into three groups for Percent Chironomidae. Riffle transects had significantly lower percent chironomids than did glide transects in Little Salmon River (10–14) or glide transects in the former reservoir and lower Salmon River (*P* <0.003). There was no significant difference in pre- and post-dam removal in Percent Chironomidae ($F = 0.59$, $P = 0.44$).

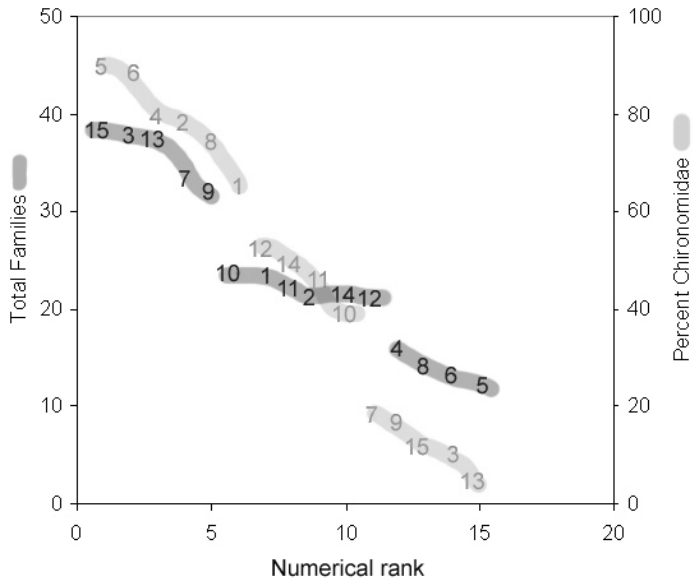



Figure 16. Numerically ranked value of Total Families and Percent Chironomidae at transects for six sampling periods from 2002 to 2010. Numbers refer to transects. Transects that are not connected with a common color shade are statistically different.

Transect value relationships for Percent Dominants were more complex than for the other indices. Riffle transects, and transect 10, formed a group that shared some similarity with transects of the Little Salmon River, which shared similarity with two other groups, the lower Salmon River and transects in the former reservoir with transect 8 (Figure 17). There was no significant difference in pre- and post-dam removal in Percent Dominants ($F = 3.69$, $P = 0.06$).

Values for the Family Biotic Index differed only between the riffle transects and the glides. There was no significant difference in pre- and post-dam removal in Family Biotic Index ($F = 3.37$, $P = 0.07$). There were no statistical differences in the values for riffle transects in any of the indices.

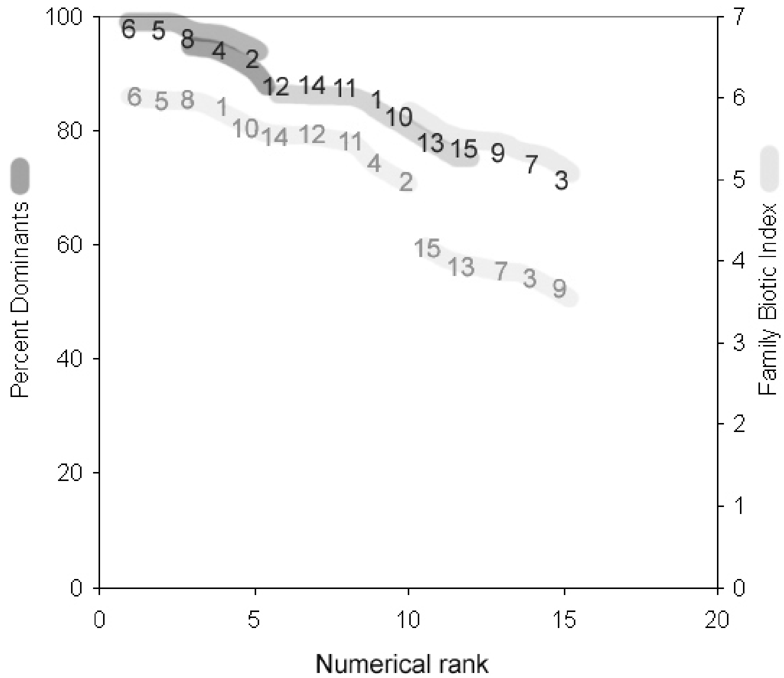



Figure 17. Statistical relationship of transects within Percent Dominants, and Family Biotic Index against the numerically ranked index value of each transect. Numbers refer to transect. Those transects that share a color shade are not significantly different (α = 0.003).

*Unionid mussels*. Nine species of living mussels were collected from 1000 quadrats of 0.25 $m^2$ area, of which 87 were excavated. The use of double sampling can provide a calibration for surface counts if the relationship between surface and total counts is linear (Smith et al. 2001). Only the collection at transect 12 provided sufficient mussels to regress the surface counts against total counts and this was found to be linear but rather weak ($r^2 = 0.62$), therefore a ratio estimator (Strayer and Smith 2003) was used to estimate the mussel population.

*Elliptio complanata*, *Lampsilis radiata*, and *Strophitus undulatus* were the more abundant species accounting for 97% of the total collected. *Elliptio complanata* was collected at all transects except transects 3 and 9, and had the greatest density, more than 17 times greater than *L. radiata*. *Lampsilis cariosa* was collected only in Deer Creek and *L. ovata* only at Lewis Marina and at relatively low density (Table 5). One living *Alasmidonta undulata* was collected in a benthic sample at transect 8.

Table 5. Density (all transects combined) and population estimate of living adult mussels collected in the mussel survey in 2010. Relative abundance is based on mussels collected. Population estimate and density are based on the total area of transects, not extrapolated to the entire study area.

| Species | Number collected | Population estimate | Population 95% CI | Relative abundance (%) | Density (no./m$^2$) | Density SE |
|---|---|---|---|---|---|---|
| | 199 | 597 | 595.0–599.0 | | 0.155 | 0.0326 |
| *Anodontoides ferussacianus* | 1 | 3 | 1.6–5.6 | 0.50 | 0.001 | 0.0010 |
| *Elliptio complanata* | 179 | 537 | 409.0–705.1 | 89.95 | 0.139 | 0.1469 |
| *Lampsilis cariosa* | 1 | 3 | 1.6–5.6 | 0.50 | 0.001 | 0.0021 |
| *Lasmigona costata* | 1 | 3 | 1.6–5.6 | 0.50 | 0.001 | 0.0007 |
| *Lampsilis ovata* | 1 | 3 | 1.6–5.6 | 0.50 | 0.001 | 0.0071 |
| *Lampsilis radiata* | 10 | 30 | 20.7–43.4 | 5.03 | 0.008 | 0.0145 |
| *Pyganodon cataracta* | 1 | 3 | 1.6–5.6 | 0.50 | 0.001 | 0.0010 |
| *Pyganodon grandis* | 1 | 3 | 1.6–5.6 | 0.50 | 0.001 | 0.0010 |
| *Strophitus undulatus* | 4 | 12 | 8.5–16.9 | 2.01 | 0.003 | 0.0026 |

Transects 8 and 12 had the greatest density of mussels (2.37/m$^2$) followed by Lewis Marina at (2.3/m$^2$) in 2010. The remaining transects had mussel densities of less than 0.7/m$^2$. Mean density of all mussels was greater in the Little Salmon River (1.5/m$^2$) than in the Salmon River (0.02/m$^2$).

The cumulative distribution of mussels was significantly different between glide transects and riffle transects (Kolmogorov-Smirnov D = 33.1, maximum difference = 45.2, $\alpha = 0.01$). Mussels in glides were collected more frequently on the east side of both rivers but mussels in riffles were more abundant on the west side (Figure 18).

Overall mussel density was not significantly different from 2005 through 2010 ($F$= 0.78, $P$ = 0.57) but Lewis Marina had a greater density of mussels (1.02/m$^2$) than other transects (lsmeans <0.0001). Mussel density decreased at transect 5 (within the former reservoir) by 83% compared to pre-dam removal years. Density increased by 2.5 times at Lewis Marina and 5.5 times at transect 12, due to collecting more juveniles from increased excavation: a similar number of adults were collected. Density also increased at transects 8 and 15 but the increase at transect 12 was the only one that was significant (lsmeans = 0.004; Figure 19).

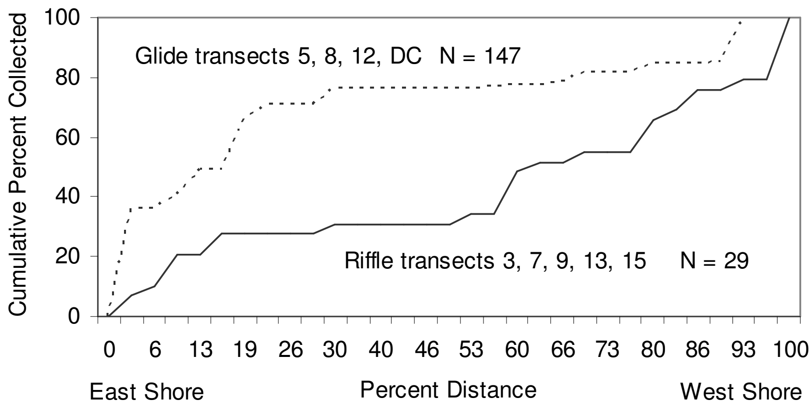



Figure 18. Cumulative percent distribution of adult mussels in glide and riffle transects, excluding Lewis Marina. Percent distance refers to the distance across the river starting from the east shore.

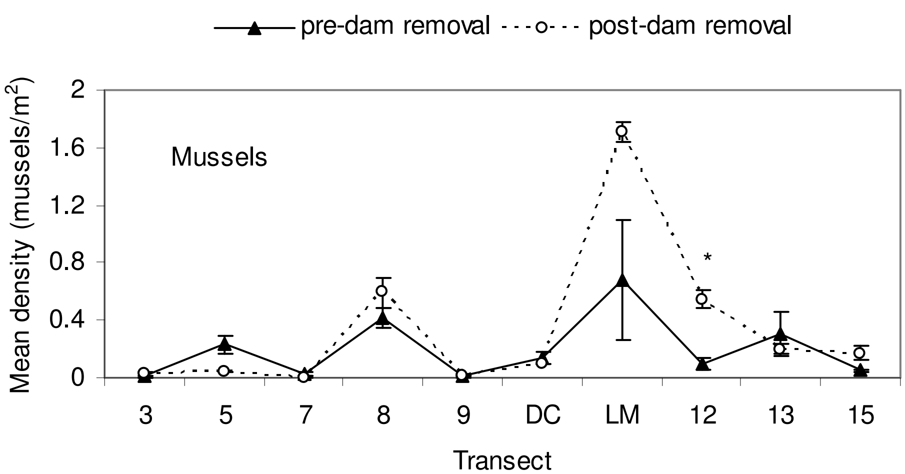



Figure 19. Mean mussel density (± 1 SE) by transect in pre-dam removal years (2005–2008) and post-dam removal years (2009–2010). An asterisk refers to a significant difference. DC = Deer Creek and LM = Lewis Marina.

*Juvenile mussels*. Quadrat excavation produced 61 juvenile mussels, all at transects 12 and Lewis Marina (Table 6). Benthic collections resulted in three additional *Elliptio complanata* (SL range 7.5–38 mm) and one *Pyganodon* sp. (SL 6.5 mm), all from transect 12. *Elliptio complanata* had the greatest density at Lewis Marina, nearly three times greater than at transect 12. No juvenile mussels were collected at other transects.

Table 6. Number and shell length (mm) of juvenile mussels collected in the mussel survey and benthic samples in 2010. Density was based on transect area: transect 12 = 105 $m^2$ and Lewis Marina = 14 $m^2$.

| | Transect 12 | | Lewis Marina | | All mussels combined | | |
|---|---|---|---|---|---|---|---|
| Species | Number collected | Density (no./$m^2$) | Number collected | Density (no./$m^2$) | Mean SL | Range | % |
| *Elliptio complanata* | 39 | 0.37 | 16 | 1.1 | 21.7 | 7.5–35.6 | 90.2 |
| *Lampsilis radiata* | 3 | 0.03 | 1 | 0.07 | 32.7 | 25.7–38.9 | 6.5 |
| *Pyganodon* sp. | | | 1 | 0.07 | | 6.5 | 1.6 |
| *Lampsilis ovata* | | | 1 | 0.07 | | 20.3 | 1.6 |
| Total | 42 | | 19 | | | | |
| % | 68.8 | | 31.1 | | | | |

*Middens*. Five middens were located in 2010 that were not seen in previous years, all were in the Little Salmon River, extending from transect 11 (midden 1) to upstream of the Foster Road Bridge near transect 15 (long midden). These new middens yielded 316 mussels of 9 species (Table 7) in roughly the same relative proportion as living mussels

Table 7. Number and mean shell length (SL) of those mussel species collected in five middens in 2010.

| | Location designation | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Species | Bridge midden 1 | Midden 1 | Bridge midden 2 | Old midden 5 | Long midden | Total | Mean SL | Percent of total |
| *Alasmidonta undulata* | | | | | 1 | 1 | 59 | 0.3 |
| *Alasmidonta marginata* | 1 | | | | 3 | 4 | 68.7 | 1.3 |
| *Elliptio complanata* | 102 | 3 | 13 | 11 | 85 | 214 | 72.5 | 67.3 |
| *Lampsilis cariosa* | 1 | | | | 3 | 4 | 86.5 | 1.3 |
| *Lasmigona costata* | 1 | | | | 10 | 11 | 73.5 | 3.5 |
| *Lampsilis ovata* | | | | | 2 | 2 | 88.7 | 0.6 |
| *Lampsilis radiata* | | 3 | | 2 | | 5 | 60.7 | 2.2 |
| *Pyganodon cataracta* | | | | | 3 | 3 | 61.6 | 0.9 |
| *Strophitus undulatus* | 17 | | 2 | | 53 | 72 | 70.2 | 22.6 |
| | 122 | 6 | 15 | 13 | 160 | 316 | | |
| Percent of total | 38.6 | 1.9 | 4.7 | 4.1 | 50.6 | | | |

collected at transects. A greater proportion of *Strophitus undulatus* was collected in 'long midden' than were present in the areas searched for living mussels in the study area. The area around 'long midden' was not searched for living mussels. Two middens accounted for more than 89% of the mussels collected. Predation damage was evident in 81% of the mussels.

*Fish*. Hoop nets were fished for 445 hr at eight locations in 2010: six in the Salmon River and two in the Little Salmon River. Twelve species were collected with hoop nets (N = 217). Longnose gar was the most abundant in CPUE (Table 8) with rock bass second; these two species accounted for 68% of the total catch in hoop nets.

Twenty-seven seine hauls were made at 14 locations covering 131 m. Seine haul distance ranged from 9 m to 13 m. Tessellated darter and rosyface shiner were the more abundant species out of the 16 species collected (Table 8). Eastern sand darter was 3$^{rd}$ in CPUE with the largest catch (N = 9) being made on a new sand bar in the Salmon River at its confluence with Little Salmon River. Eastern sand darter was also collected at transects 1, 2, 3, 5, 6, 11, and 67.

Table 8. Catch-per-unit-effort of 10 more abundant fish species collected in hoop nets and seine in 2010. CPUE for hoop net is based on total hours fished, and for total distance for seine.

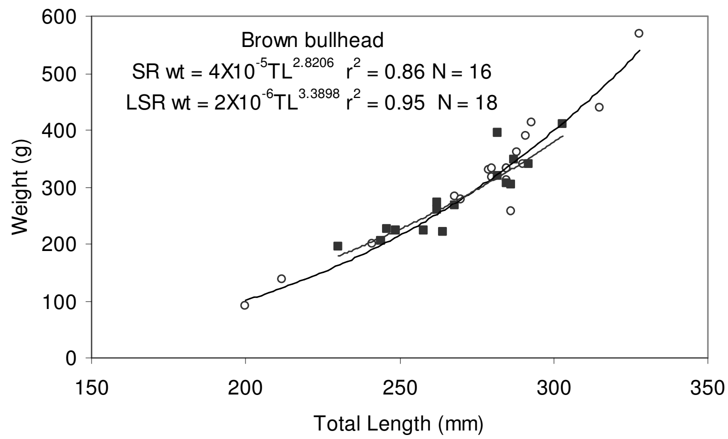
| | Hoop net (445 hours) | | Seine (131 m) |
|---|---|---|---|
| Species | CPUE | Species | CPUE |
| Longnose gar | 0.245 | Tessellated darter | 0.77 |
| Rock bass | 0.085 | Rosyface shiner | 0.55 |
| Brown bullhead | 0.076 | Eastern sand darter | 0.53 |
| Pumpkinseed | 0.052 | White sucker | 0.48 |
| Greater redhorse | 0.004 | Mimic shiner | 0.47 |
| Yellow perch | 0.004 | Pumpkinseed | 0.38 |
| Smallmouth bass | 0.004 | Smallmouth bass | 0.38 |
| Shorthead redhorse | 0.002 | Rock bass | 0.34 |
| Northern pike | 0.002 | Logperch | 0.28 |
| White sucker | 0.002 | Spottail shiner | 0.18 |

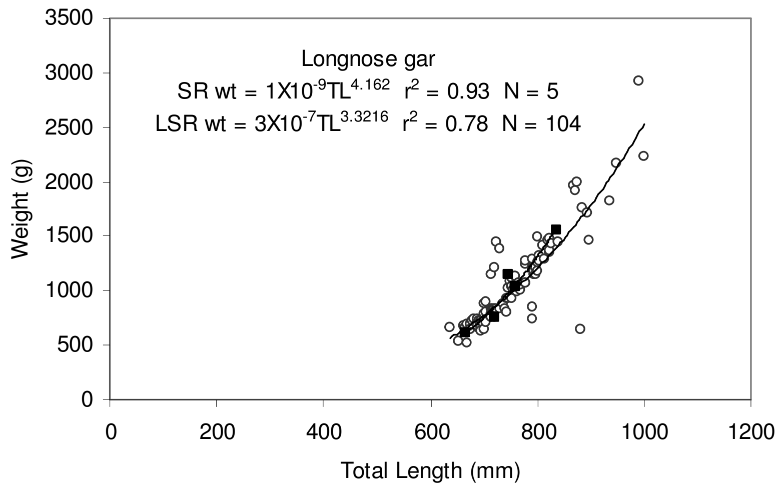
Twenty-three species of fishes (N = 814) were collected in the study area (all gear combined). A greater percentage of fish was caught in the Little Salmon River (61%) than in the Salmon River (39%).

Longnose gar and smallmouth bass were the more abundant predators: longnose gar was ranked first, smallmouth bass ranked 8th, and northern pike ranked 20th. No sea lamprey was collected in the study. Silver lamprey was the only parasitic lamprey collected (15th in relative abundance) and was collected only in the Little Salmon River. The round goby (*Neogobius melanostomus*) was collected in two locations in the Salmon River for the first time.

Logarithmic regressions of length and weight of three species resulted in relationships with good predictive characteristics with the exception of longnose gar (Figure 20). Longnose gar was caught only during the spring spawning period (although they were observed at other times) and the catch most likely included post-spawning adults, which would affect the length-weight relationship. Regressions showed that the length-weight relationship of brown bullhead, longnose gar, and rock bass were similar between the Salmon and Little Salmon rivers. There were not enough fish collected of other species at each site to make this comparison.

Length frequency plots of the more abundant fish species collected by seine showed a wide range of sizes (Figure 21) but the majority were young-of-year. More than 80% of white sucker and mimic shiner were collected in June, and 67%–88% of rock bass, rosyface shiner, tessellated darter, and spottail shiner were collected in September. Eastern sand darter and pumpkinseed were collected evenly in both months.



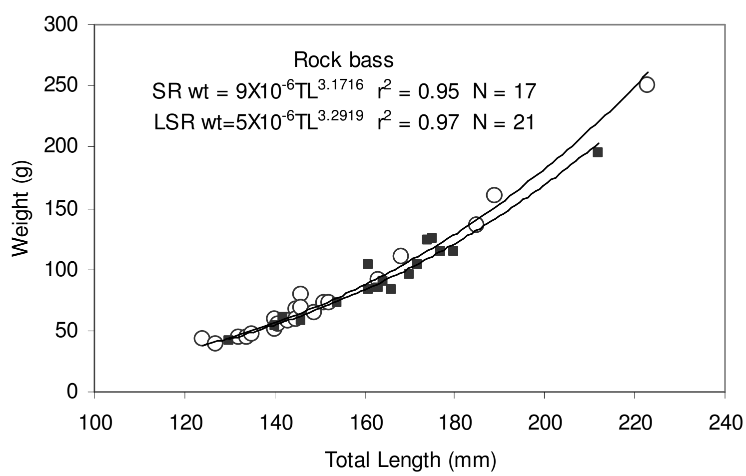




Figure 20. Length-weight relationships for three more abundant fishes collected in trap nets in the Salmon and Little Salmon rivers. Solid squares represent fish from the Salmon River (SR) and open circles represent fish from the Little Salmon River (LSR).

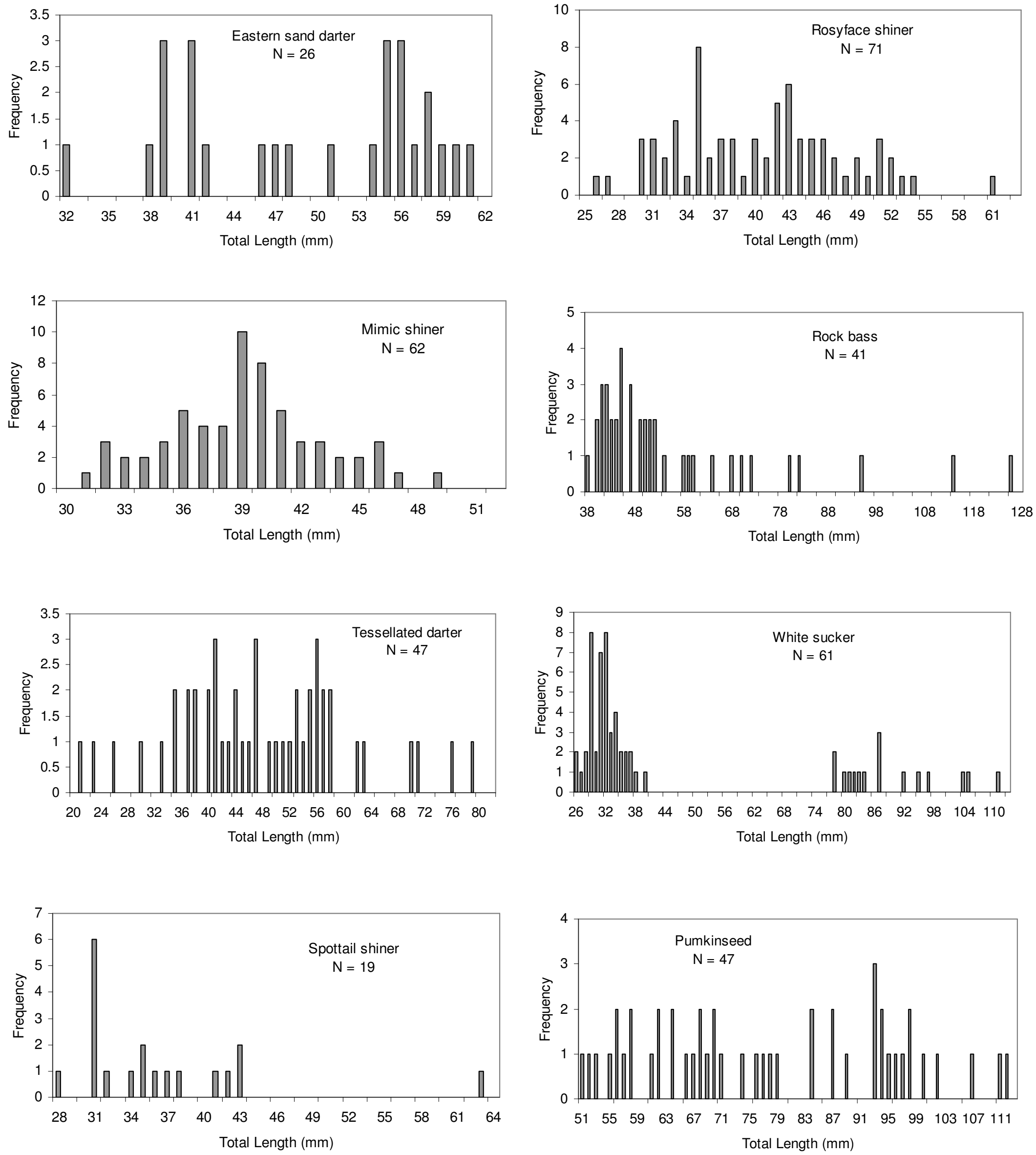



Figure 21. Length frequency plots of eight fish species collected by seining in the Salmon and Little Salmon rivers. Not all fish collected were measured.

There were 16 species not collected in 2010 that were collected in previous years, several of these species were collected infrequently in the past. The number of species collected in any year never exceeded 23 and was not lower than 13. Carp was the only species collected upstream of the former dam that had not been collected upstream previously. Yellow perch was not collected upstream of the dam from 2002–2004 but was collected upstream of the dam in 2008. Other species that were not collected upstream of the dam from 2002–2004 were walleye, longnose gar, and American eel. One walleye was collected in the Little Salmon River in 2008 and no American eel have been collected since 2003. Longnose gar was abundant in 2010 but was not collected upstream of the former dam site.

CPUE was correlated to water temperature in trap net and seine collections but the relationship was stronger in the Salmon River than in the Little Salmon River (Table 9). CPUE in trap net samples ranged from 0.06 fish/hr in the Salmon River in 2002 to 2.94 fish/hr in the Little Salmon River in 2008. CPUE in seine collections ranged from 0.35 fish/m seined in 2008 to 5.1 fish/m seined in 2010.

Table 9. Pearson correlation coefficients between catch-per-unit-effort (CPUE) and water temperature for all sampling periods from 2002 through 2010. N = number of sampling periods.

| | Salmon River | Little Salmon River |
|---|---|---|
| Trap net (N = 8) | 0.83 | 0.56 |
| Seine (N = 5) | 0.94 | 0.55 |

*Index of biotic integrity*. The Salmon River had lower scores in richness and composition metrics in 2010 than did the Little Salmon River, especially in percentage of dominant species. Number of insectivore species and water column species were also lower in the Salmon River (Table 10). Scores for the Salmon River were lower in two of the trophic composition metrics (percent omnivores and top carnivores) but at the maximum for percent insectivores. The IBI score was 'good' for the Salmon River and 'very good' for the Little Salmon River.

Total scores for the fish IBI declined in the Salmon River from 48 in 2002–2004 to 38 in 2010. The lower scores resulted from higher percentages of dominant species, omnivores, and insectivores. The total score for the Little Salmon River has varied between 40 and 44 for the three sampling periods of 2002–2004, 2008, and 2010.

Table 10. Metric scores for 2010 for fish Index of Biotic Integrity (IBI) following Daniels et al. (2002) based on a watershed of 2838 km$^2$.

| | | Scoring | | | Salmon River | | Little Salmon River | |
|---|---|---|---|---|---|---|---|---|
| Metric | Description | 5 | 3 | 1 | value | score | value | score |
| | | | | Resident fish species richness and composition | | | | |
| 1 | Total number of species | >13 | 6 - 12 | <6 | 15 | 5 | 15 | 5 |
| 2 | Number of benthic insectivores | >4 | 2 - 4 | <2 | 3 | 3 | 7 | 5 |
| 3 | Water column species | >5 | 2.5 - 5 | <2.5 | 3 | 3 | 6 | 5 |
| 4 | Number of terete minnow species | >4.5 | 2 - 4.5 | <2 | 5 | 3 | 4 | 3 |
| 5 | % dominant species | <40% | 40-55% | >55% | 59.9 | 1 | 55.0 | 3 |
| 6 | % total white sucker | <3% | 3-15% | >15% | 3.5 | 3 | 10.5 | 3 |
| | | | | Trophic composition | | | | |
| 7 | % total omnivores | <20% | 20-45% | >45% | 45.5 | 1 | 43.7 | 3 |
| 8 | % total insectivores | >50% | 25-50% | <25% | 50.9 | 5 | 24.4 | 1 |
| 9 | % carnivores | >5% | 1-5% | <1% | 2.9 | 3 | 21.4 | 5 |
| | | | | Fish abundance and condition | | | | |
| 10 | fish abundance (no./100m$^2$) | >10 | 5-10 | <5 | 206 | 5 | 283 | 5 |
| 11 | % with 2 age groups | >40% | 15-40% | <15% | 7.1 | 1 | 26.3 | 3 |
| 12 | % with tumors, lesions, parasites | 0% | >0<1% | >1% | 0 | 5 | 0.6 | 3 |
| | Total score | | | | | 38 | | 44 |

*Aquatic plants*. Nine genera of aquatic plants were identified but only three genera occurred in the Salmon River: *Vallisneria* at transects 1, 67, and 8, *Elodea* at transect 8, and rice cut-grass (*Leersia oryzoides*) at transect 9 (Figure 22). Two areas of channel-wide plant cover were found upstream of transects 10 and 12. *Potamogeton* and *Elodea* were the dominant genera in these areas. *Elodea* dominated the plant cover at transect 8 (not shown) but the area covered was restricted to the east side. Northern wild rice (*Zizania palustris*) was found in one small area at Lewis Marina as was the exotic flowering rush (*Butomus umbellatus*). The former reservoir was devoid of any submerged aquatic plants with the exception of *Vallisneria* in a small clump at transect 67. The few colonies of the exotic European frogbit (*Hydrocharis morsus-ranae*) that had been present near transect 5 in 2004 were absent. Scattered colonies of SAV were no longer present downstream of the former dam site, including a stand of wild rice. There was no apparent change in the aquatic plant coverage in the Little Salmon River. The only vegetated areas that would serve as fish larvae nurseries were in the Little Salmon River.

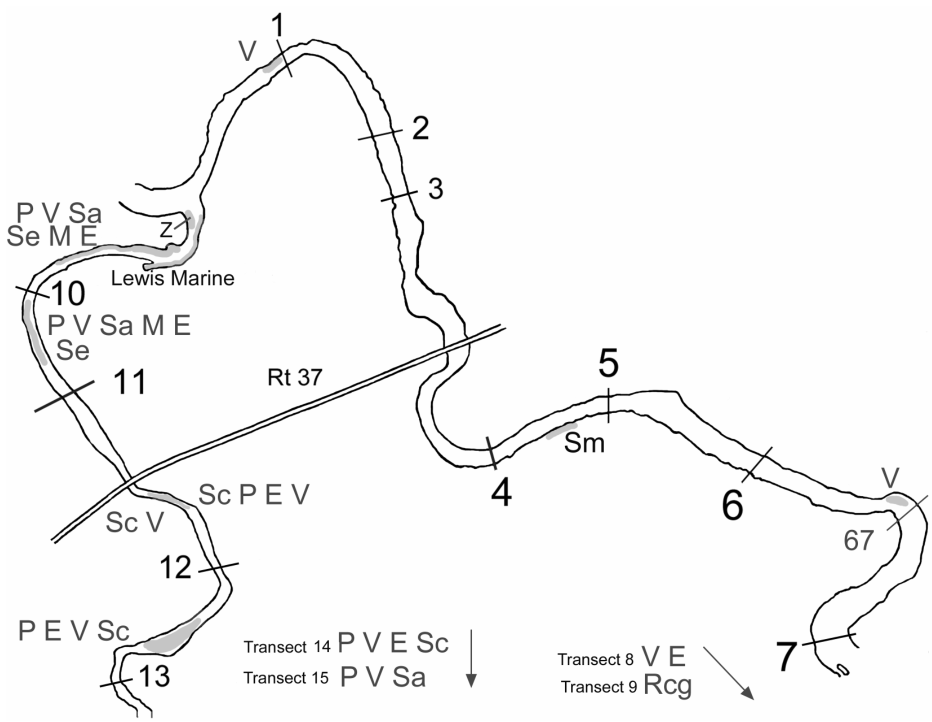



Figure 22. Aquatic plant genera identified in 2010. The letter abbreviations for plants are: V =*Vallisneria*, P = *Potamogeton*, M = *Myriophyllum*, Sc = *Scirpus*, Z = *Zizania*, E = *Elodea*, Sa = *Sagitarria*, SM = *Bidens coronata*, and Se = sedges. Flowering rush *Butomus umbellatus* was present along the north side of transect 10, and Rice cut-grass (Rcg) *Leersia oryzoides* was present at transect 9. Gray areas show the extent of the vegetation. Transects 8, 9, 14, and 15 are not shown on the figure.

## Discussion

The Salmon and Little Salmon rivers cut through a glacial moraine deposit of fine to coarse sand on the north side of Malone, New York. This sand has been deposited in the rivers and overlies the glacial lake bed silt and clay of the St. Lawrence River valley.

The Salmon and Little Salmon rivers can be classified as soft water with moderate buffering capacity. Chloride, nitrate, sulfate, and ammonia levels were lower than would be expected in natural freshwater and had similar levels in pre- and post-dam removal years, and to those determined in previous studies over the past 50 years (USGS). Nitrate and ammonia were elevated in some samples but these were localized. Dissolved oxygen levels did not change appreciably after dam removal. Velinsky et al. (2006) did not find any change in levels of pH, alkalinity, or conductivity in a Pennsylvania stream after dam removal.

The low gradient and relatively slow water velocity in the former reservoir, and large particle size of the sediment, allowed sand to accumulate. The volume of sediment behind the dam was estimated to be about 5% of the annual sediment production in the Salmon River watershed (Milone and MacBroom 2004). The numerous sand deposits in the former reservoir might not have been included in the estimate; these were located at the base of transect 7, the west side of transect 67, along shorelines at transects 5 and 6, and on the east side downstream of transect 4. The downstream movement of sediment would be of critical importance in assessing the risks of dam removal (Shuman 1995). Three conceptual models have been proposed for the transport of sediments (reviewed in Lisle et al. 1997) where the sediment can 1) move as a discrete mass with little change in shape, 2) move as a diffuse stream of particles over time, and 3) remain in place with only a small proportion moving downstream. The sand that passed through the former reservoir was most similar to conceptual model 1; the leading edge of the sand was apparent from July to November, 2009. The movement of the sand was similar to that described by Simons and Simons (1991; cited in Doyle et al. 2000) after the removal of the Newaygo Dam on the Muskegon River, Michigan, where sediment moved as a wave at about 1.6 km/year. The average rate observed in the Salmon River was equivalent to 2.3 km/year. There was speculation that much of the observed redistributed sand in the

Salmon River came from a slumping event upstream of the reservoir near Westville Center but this seems unlikely. At the maximum rate of sand movement observed in the Salmon River (8.7 m/day; Cooper, *in press*), that sand would require 9 years to reach the former dam site.

The opening of the dam increased the water velocity from three to five times the previous rate, particularly between transects 3 and 5, and resulted in scouring of sediment, primarily sand, from just downstream of transect 7 through the former reservoir. The sediment was deposited from transect 3 downstream to transect 1 reaching a depth of 3 m at transects 1 and 2. The amount of redistributed sand was estimated to be 42,480 $m^3$ (Cooper, *in press*). Erosion of sediments was facilitated by rainfall in July (12.2 cm), August (9.1 cm), and September (6.7 cm; NOAA Climate Data).

There was no change in mean sorting or porosity with removal of the dam. This was not surprising since the substrate in the Salmon River was primarily sand in all areas and redistributing similar-sized particles would not alter the values. Mean particle size in a Pennsylvania stream was reduced after dam removal (Thomson et al. 2005) but that substrate was primarily pebble with an increase in sand following dam removal.

The redistributed sand covered much of the silt habitat along the river banks and buried a good portion of the riffle habitat at transect 3. Subsequent scouring during October and November of 2009 expanded the riffles at transects 3 and 7 and this sand was deposited in the lower river. The deposition of sand downstream reached beyond the confluence of the Salmon and Little Salmon rivers by May, 2010, and formed a bar across the mouth of the Little Salmon River. This bar was partially removed by flooding in October, 2010, but had reformed by November, 2010.

Alteration of the habitats did not cause major changes in any of the macroinvertebrate indices although scouring of sand away from rocks on the east side of transect 2 in 2010 resulted in an increase in mayfly and caddisfly families, most likely due to an increase in habitat complexity. This was in contrast to macroinvertebrate density downstream of a dam in Pennsylvania where density remained lower for 1 year after dam removal (Thomson et al. 2005). Transect 4 showed an increase in mayfly and caddisfly families, Total Families, and decreased Percent Dominants and Percent Chironomids, all of which suggest an increase in diversity. The macroinvertebrate assemblage within the former reservoir remains much as it was prior to dam removal. Glide transects were dominated by midges (Chironomidae) while caddisflies, mayflies, riffle beetles, and midges were more abundant in riffle transects. This was similar to that described by Stanley et al. (2002) in the Baraboo River. The former reservoir transects, and transect 8, shared similar ranks derived from the macroinvertebrate indices, which indicated that these areas support less diverse communities. These transects have a nearly uniform sand substrate that would reduce habitat diversity (Hill et al. 1993).

Dam removal was more disruptive for mussels. The lowering of the water level in the reservoir stranded, and subsequently killed, approximately 77% of the reservoir population of mussels (N = 2,954; Cooper, *in press*) in 2009, and this was reflected in a decline in mussel density at transect 5 in 2010. A similar result was described by Sethi et al. (2004) for a dam removal in Wisconsin, although mortality was not as great. Deposition of sand in downstream areas would have buried mussels but there are no estimates of the population. The potential for burial of downstream mussels can be drawn from the contents of two mussel middens near transect 2. These two middens contained five mussel species (246 mussels) of which two species, *Lampsilis ovata* and *Lampsilis cariosa*, accounted for 34%. These two species are listed as those in greatest conservation need by the New York Heritage Program. It is possible that these shells represented a dense mussel bed that is now covered by 3 m of sand. Dissolution rates of shell material could be fairly high in the low-calcium water of the Salmon River (Strayer and Malcolm 2007), thus midden contents would not represent a long-term accumulation.

The fish IBI declined in the Salmon River from previous years but this might be due to decreased fishing efficiency from lowered water depth. The redistribution of sand rendered many areas of the river too shallow for hoop nets and those areas that had sufficient water depth also had greater water velocity, which prevented the anchored hoop nets from remaining in place. The formation of a sand bar across the mouth of the Little Salmon River might have reduced water flow from that river allowing silt and algae to accumulate at Lewis Marina and transect 11, which reduced the presence of fish. Eastern sand darter (a threatened species in New York) was sixth in relative abundance but was not collected at Lewis Marina in 2010 where the largest collection was made during pre-dam removal seining (N = 101). The percent of sand in the substrate was determined by Daniels (1993) to be the best predictor of sand darter abundance. Sand darters were collected only where the substrate was clean sand bottom with moderate current. Brook silverside, bluntnose minnow, common shiner, and spotfin shiner were not collected at transect 11 where they had been common in previous years. Downstream changes in the fish assemblage observed in the Baraboo River, Wisconsin, were attributed, in part, to sediment deposition (Catalano and Bozek 2007).

The fish collections revealed an assemblage that was similar to that collected in 1930 (NY Cons. Dept. 1931). The 1930 survey covered a wider area, including the headwaters of the Salmon River and collected 12 species that were not collected in either the pre-dam removal study or post-removal study. Nine of the 12 species were considered to be headwater species and would not be found in the lower Salmon River. Three fish species collected in the 1930 survey were not collected from 2002 to 2010: blacknose shiner, channel darter, and Johnny darter. Blacknose shiner was collected in the St. Regis River in 2004 (Dawn Dittman, USGS, pers. comm). Five fish species not collected in the 1930 study were collected from 2002–2008: longnose gar, carp, central mudminnow, American eel, and brook silverside, but of these five species, only longnose gar and carp were collected in 2010. Largemouth bass was collected in the Salmon River in 1998 and 2001 (Morrill and Tyson 2001), 2002 (Cooper et al. 2004), and in the St. Regis River in 2004 (Dawn Dittman, USGS, pers. comm). The exotic round goby was collected for the first time in 2010 downstream of the former reservoir.

*Predicted changes*. The baseline report (Cooper et al. 2004) predicted changes to the river following dam removal. Many of these were proved to be accurate but others were not. It was correctly predicted that there would not be any effect upstream from transect 7 (upstream limit of the reservoir), but predicted incorrectly that there would be no substantial effect downstream of transect 2. Although the amount of sediment directly behind the dam was of minor concern, the prediction did not anticipate the scouring of the sand deposits within the reservoir, which proved to be extensive.

The riffle at transect 7 did expand downstream as predicted but to a much greater extent than expected. The exposure of the additional riffle at transect 67 was not foreseen.

Transects 5 and 6 remained as glide-type habitats, similar to transect 8, and it is still possible that increased flow velocity could incise a deeper channel in the sand substrate. It was incorrect to predict that the resulting lower water level would allow emergent and submergent plant colonization along the river banks which could serve as spawning and nursery areas for fish larvae and habitat for macroinvertebrates. Ice-scouring and increased flow velocity has not allowed submergent plants to colonize the river banks.

It was predicted that the ponds would become narrower and that fish might gain access to them from the river if a channel was eroded through the sill at the downstream end. Both ponds have dried completely and there is little reason to expect fish gaining access to either pond from the river: the erosion of the sill has occurred but remains much higher in elevation than the river.

Transect 4 was predicted correctly to experience the most change in flow velocity and the macroinvertebrate community has increased in EPT families but remains dominated by Chironomidae. The sand bar at the upstream end of this transect was not colonized by aquatic vegetation; it was completely removed by water flow.

The riffle at transect 3 has expanded upstream as predicted, but has not reached transect 4 as yet. Continued scouring of the sand might fulfill this prediction.

Transects 1 and 2 were predicted to receive additional sediment after dam removal and that any deposition would be for a short time: sand deposition has reduced the water depth at these transects by about 80% and will require many years to move the sand downstream.

The mussel population was not expected to be at risk with the exception of stranding in the former reservoir as the water level was lowered. Stranding was more severe than expected as was sedimentation downstream. The drawdown of the reservoir was more rapid than expected and might have contributed to the extent of stranding.

Fish migration upstream is now possible for American eel, walleye, longnose gar, and carp for the first time in 91 years, although the benefit to American eel and walleye remains limited due to their low population level in the river. Predation on forage fish by longnose gar and smallmouth bass was predicted to increase, particularly if access to the in the ponds was possible. This was expected to increase the growth rate of smallmouth bass as was seen in the Milwaukee River after dam removal (Kanehl et al. 1997). None of these predictions were borne out: pond access was not possible and no longnose gar or smallmouth bass were collected upstream of the former dam site. Carp were of concern upstream of the former dam as they might reduce spawning success of some forage fish by disturbing sediments and vegetation (Roberts et al. 1995). Carp have moved upstream but there was no evidence of any disruption.

The habitat for the eastern sand darter was predicted to not change appreciably and it would appear that was accurate in the Salmon River but not in the Little Salmon River where the best habitat (Lewis Marina) was covered by silt and algae. No sand darter was caught there. New migrants, such as sea lamprey and lake sturgeon, are possible although sea lamprey have not been collected in the Salmon River. Lake sturgeon restoration through stocking appears to be successful in the St. Regis River (a tributary of the St. Lawrence River west of the Salmon River) and might lead to colonization of the Salmon River, however, no lake sturgeon have been collected.

The data presented in this report covers only the first year of habitat recovery. The flooding that occurred in October, 2010, ranked among the higher flows since 1925 and the effects on the aquatic community might mask any long-term effects from dam removal. In any case, the deposition of sand in much of the lower Salmon River will be a controlling factor in determining the structure of the aquatic community in the future.

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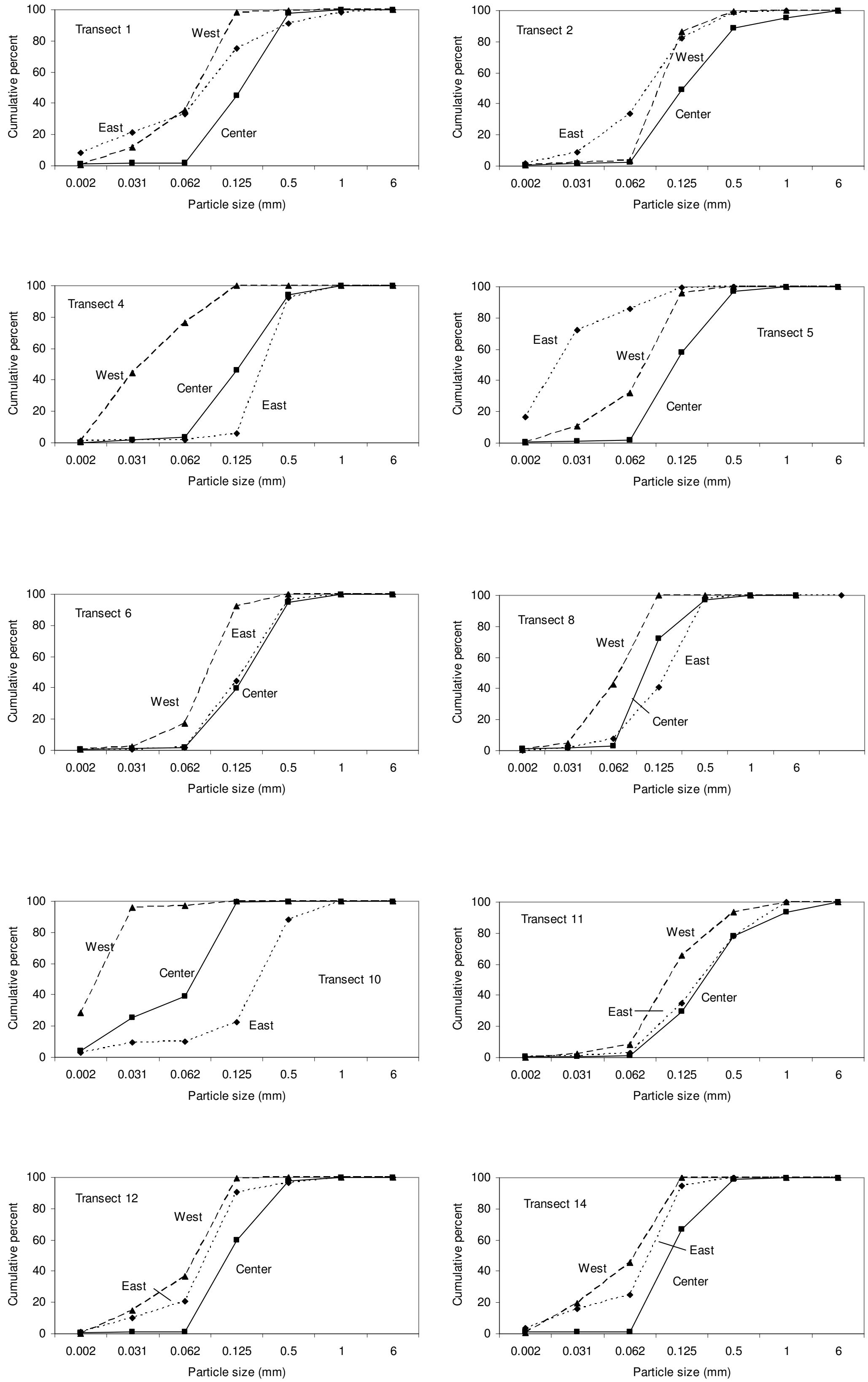


Appendix Figure 1. Cumulative percent of particle size from re-screened sediment samples from glide transects taken in 2002.

Appendix table 1. Total number of macroinvertebrates by transect for 2010. These families were used to construct the macroinvertebrate indices.

| | Transect | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **12** | **13** | **14** | **15** | **67** | **TOTAL** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Family | | | | | | | | | | | | | | | | | |
| caddisfly | Hydroptilidae | 2 | 69 | 90 | 37 | 0 | 0 | 116 | 0 | 6 | 3 | 0 | 0 | 40 | 2 | 30 | 17 | 412 |
| | Philopotamidae | 0 | 29 | 7 | 2 | 0 | 0 | 152 | 0 | 45 | 0 | 0 | 0 | 152 | 0 | 273 | 56 | 716 |
| | Hydropsychidae | 0 | 19 | 1135 | 21 | 1 | 0 | 1254 | 0 | 1153 | 2 | 0 | 0 | 1049 | 0 | 2312 | 1235 | 8181 |
| | Molannidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Limnephilidae | 0 | 3 | 0 | 4 | 0 | 0 | 8 | 0 | 22 | 0 | 0 | 0 | 12 | 0 | 13 | 1 | 63 |
| | Phyrganeidae | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 2 |
| | Helicopsychidae | 0 | 0 | 4 | 0 | 0 | 0 | 74 | 0 | 1 | 0 | 0 | 0 | 185 | 1 | 188 | 0 | 453 |
| | Brachycentridae | 0 | 0 | 88 | 1 | 0 | 0 | 629 | 0 | 677 | 0 | 0 | 0 | 39 | 0 | 5 | 212 | 1651 |
| | Polycentropodidae | 1 | 1 | 12 | 8 | 0 | 0 | 2 | 0 | 2 | 10 | 7 | 6 | 1 | 20 | 7 | 6 | 83 |
| | Leptoceridae | 1 | 1 | 0 | 0 | 0 | 0 | 1 | 1 | 1 | 1 | 0 | 1 | 11 | 2 | 122 | 1 | 143 |
| | Glossosomatidae | 0 | 0 | 1 | 0 | 0 | 0 | 4 | 0 | 15 | 0 | 0 | 0 | 4 | 0 | 6 | 0 | 30 |
| | Rhyacophilidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 1 | 8 |
| | Psychomyiidae | 0 | 7 | 36 | 4 | 0 | 0 | 36 | 0 | 15 | 0 | 0 | 0 | 0 | 0 | 0 | 15 | 113 |
| | Odontoceridae | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 8 | 0 | 3 | 0 | 18 |
| | Lepidostomatidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 |
| mayfly | Heptageniidae | 27 | 67 | 99 | 103 | 0 | 0 | 82 | 0 | 103 | 0 | 0 | 1 | 336 | 0 | 451 | 65 | 1334 |
| | Baetidae | 1 | 18 | 131 | 13 | 1 | 0 | 115 | 0 | 76 | 2 | 0 | 0 | 2740 | 2 | 2138 | 77 | 5314 |
| | Isonychiidae | 0 | 0 | 36 | 0 | 0 | 0 | 125 | 0 | 173 | 0 | 0 | 0 | 118 | 0 | 146 | 90 | 688 |
| | Ephemeridae | 0 | 1 | 4 | 1 | 0 | 0 | 1 | 0 | 0 | 5 | 3 | 1 | 0 | 8 | 0 | 0 | 24 |
| | Polymitarcyidae | 0 | 0 | 5 | 3 | 0 | 1 | 7 | 0 | 3 | 0 | 0 | 0 | 1 | 0 | 2 | 15 | 37 |
| | Ephemerellidae | 0 | 1 | 40 | 1 | 0 | 0 | 508 | 0 | 129 | 0 | 0 | 0 | 503 | 0 | 1267 | 109 | 2558 |
| | Caenidae | 8 | 7 | 0 | 17 | 0 | 1 | 5 | 0 | 0 | 0 | 0 | 5 | 0 | 5 | 98 | 0 | 146 |
| | Baetiscidae | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| | Leptophlebiidae | 1 | 0 | 0 | 1 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 9 |
| | Tricorythidae | 2 | 5 | 6 | 11 | 0 | 0 | 2 | 1 | 13 | 0 | 0 | 0 | 35 | 0 | 354 | 3 | 432 |
| stonefly | Perlidae | 0 | 0 | 4 | 0 | 0 | 0 | 3 | 0 | 3 | 0 | 0 | 0 | 56 | 0 | 12 | 0 | 78 |
| | Taeniopterygidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Nemouridae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 3 |
| beetles | Elmidae | 2 | 20 | 18 | 1 | 1 | 0 | 329 | 1 | 274 | 43 | 20 | 16 | 768 | 21 | 1715 | 46 | 3275 |
| | Psephenidae | 28 | 2 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 29 | 0 | 39 | 0 | 100 |

Appendix table 1. Total number of macroinvertebrates by transect for 2010, continued.

| | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Hydrophilidae | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 13 | 0 | 0 | 0 | 7 | 0 | 0 | 21 |
| | Dryopidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Gyrinidae | 3 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 2 | 0 | 1 | 3 | 0 | 13 |
| | Dytiscidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Noteridae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Staphylinidae | 0 | 0 | 0 | | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 |
| | Haliplidae | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| | Chrysomelidae | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 3 |
| | Curculionidae | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| amphipods | Gammaridae | 8 | 7 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 5 | 1 | 1 | 4 | 25 | 6 | 0 | 58 |
| | Hyalellidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 155 | 0 | 0 | 0 | 4 | 1 | 0 | 160 |
| snails | Viviparidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 2 |
| | Pleuroceridae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 7 |
| | Valvatidae | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 3 | 0 | 6 |
| | Physidae | 1 | 3 | 0 | 25 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 4 | 13 | 14 | 0 | 62 |
| | Hydrobiidae | 1 | 0 | 2 | 0 | 0 | 0 | 0 | 3 | 0 | 114 | 10 | 13 | 1 | 24 | 5 | 0 | 173 |
| | Ancylidae | 0 | 0 | 32 | 0 | 0 | 0 | 7 | 1 | 32 | 0 | 0 | 0 | 0 | 0 | 17 | 11 | 100 |
| | Planorbidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 1 | 0 | 1 | 0 | 7 |
| | Lymnaeidae | 0 | 0 | 0 | 29 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 31 |
| | Bithynidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 1 |
| clams | Sphaeriidae | 1 | 1 | 1 | 0 | 0 | 1 | 14 | 6 | 3 | 68 | 24 | 19 | 11 | 10 | 145 | 0 | 304 |
| diptera | Simuliidae | 0 | 0 | 147 | 0 | 0 | 0 | 94 | 0 | 74 | 1 | 0 | 0 | 475 | 0 | 158 | 47 | 996 |
| | Tipulidae | 0 | 16 | 254 | 15 | 1 | 0 | 88 | 0 | 118 | 1 | 0 | 0 | 3 | 0 | 12 | 41 | 549 |
| | Tabanidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 3 | 0 | 2 | 0 | 8 | 0 | 0 | 16 |
| | Ceratopogonidae | 10 | 20 | 0 | 27 | 5 | 53 | 1 | 21 | 0 | 3 | 0 | 2 | 1 | 3 | 0 | 0 | 146 |
| | Empididae | 0 | 6 | 33 | 4 | 0 | 0 | 14 | 0 | 17 | 0 | 0 | 0 | 4 | 0 | 14 | 13 | 105 |
| | Stratiomyidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Athericidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Dolichopodidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Ephydridae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Culicidae | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| | Muscidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Appendix table 1. Total number of macroinvertebrates by transect for 2010, continued.

| | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| chironomids | Chironomidae | 294 | 190 | 296 | 443 | 273 | 128 | 721 | 127 | 372 | 297 | 83 | 68 | 396 | 172 | 1073 | 201 | 5134 |
| spongillafly | Sisyridae | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| damselfly | Protoneuridae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Coenagrionidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 0 | 0 | 1 | 3 | 3 | 0 | 16 |
| | Lestidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Calopterygidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| dragonfly | Libellulidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Gomphidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 |
| | Aeshnidae | 0 | 0 | 0 | 1 | 0 | 0 | 2 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 4 |
| | Cordulidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Macromiidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| moths | Pyralidae | 0 | 1 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 |
| | Nepticulidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Noctuidae | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| isopod | Asellidae | 10 | 12 | 0 | 25 | 0 | 0 | 23 | 0 | 20 | 70 | 2 | 0 | 0 | 49 | 1 | 1 | 213 |
| megaloptera | Sialidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 1 | 1 | 0 | 5 | 1 | 0 | 18 |
| | Corydalidae | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 4 |
| bugs | Corixidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Belastomatidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Gerridae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 2 |
| | Aphididae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Notonectidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Nepidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Saldidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Hebridae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Veliidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 |
| wasp | Mymaridae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Braconidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| TOTAL | | 404 | 512 | 2484 | 802 | 283 | 185 | 4433 | 165 | 3356 | 825 | 152 | 146 | 7001 | 387 | 10642 | 2266 | 34043 |

Appendix table 2. Total catch of fishes by sampling gear in 2010. These totals were used to construct the fish index of biotic integrity

**Salmon River**

| species | traps | seine | benthos | total |
|---|---|---|---|---|
| brown bullhead | 16 | | | 16 |
| eastern sand darter | | 69 | | 69 |
| fallfish | | 16 | | 16 |
| longnose gar | 5 | | | 5 |
| mimic shiner | | 53 | | 53 |
| rock bass | 17 | 4 | | 21 |
| rosyface shiner | | 69 | | 69 |
| round goby | | 2 | | 2 |
| silver lamprey | 1 | | 1 | 2 |
| silver redhorse | 1 | | | 1 |
| smallmouth bass | 2 | 1 | | 3 |
| spottail shiner | | 22 | | 22 |
| tessellated darter | | 25 | | 25 |
| white sucker | 1 | 11 | | 12 |
| yellow perch | 2 | | | 2 |
| TOTAL | 45 | 272 | 1 | 318 |

**Little Salmon River**

| species | traps | seine | benthos | total |
|---|---|---|---|---|
| Am brook lamprey | | | 1 | 1 |
| brown bullhead | 17 | | | 17 |
| carp | 2 | | | 2 |
| eastern sand darter | | 1 | | 1 |
| fantail darter | | | 1 | 1 |
| grass pickerel | | 1 | | 1 |
| greater redhorse | 2 | | | 2 |
| logperch | | 37 | | 37 |
| longnose gar | 104 | | | 104 |
| mimic shiner | | 9 | | 9 |
| northern pike | 1 | | | 1 |
| pumpkinseed | 23 | 50 | | 73 |
| rock bass | 21 | 41 | | 62 |
| rosyface shiner | | 3 | | 3 |
| shorthead redhorse | | 1 | | 1 |
| silver redhorse | | 1 | | 1 |
| smallmouth bass | 1 | 49 | | 50 |
| spottail shiner | | 1 | | 1 |
| tessellated darter | | 76 | | 76 |
| white sucker | | 52 | | 52 |
| yellow perch | | 1 | | 1 |
| TOTAL | 171 | 323 | 2 | 496 |

Appendix table 3. Values calculated for the Percent Model Affinity index (Novak and Bode 1992). Taxon listed as "other" includes Simuliidae, Gammaridae, Asellidae, Physidae, and Empididae. Levels of effect are 'none' = 65% or greater, 'slight' = 50 to 64%, 'moderate' (mod) = 35 to 49%, and 'severe' <35%.

Percent abundance of each taxa by transect

| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 67 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Trichoptera | 1.27 | 26.64 | 61.73 | 10.50 | 0.34 | 0 | 52.25 | 0.37 | 60.29 | 2.91 | 2.27 | 1.88 | 21.54 | 5.42 | 28.15 | 67.90 |
| Ephemeroptera | 9.92 | 20.29 | 14.42 | 20.46 | 0.34 | 1.92 | 19.49 | 0.37 | 15.43 | 1.28 | 0.97 | 1.88 | 53.29 | 3.25 | 42.38 | 15.92 |
| Plecoptera | 0 | 0 | 0.18 | 0 | 0 | 0 | 0.07 | 0 | 0.09 | 0 | 0 | 0 | 0.84 | 0 | 0.11 | 0 |
| Coleoptera | 8.40 | 5.53 | 0.85 | 0.55 | 0.34 | 0 | 7.67 | 0.37 | 8.54 | 10.56 | 6.49 | 4.84 | 11.38 | 6.29 | 16.71 | 2.02 |
| Oligochaeta | 0.76 | 2.87 | 1.44 | 0.55 | 6.44 | 16.03 | 0.96 | 51.67 | 0.65 | 17.12 | 62.01 | 72.85 | 0.34 | 28.85 | 0.61 | 2.64 |
| Chironomidae | 74.81 | 38.93 | 13.30 | 60.44 | 92.54 | 82.05 | 16.55 | 47.21 | 11.55 | 54.10 | 26.95 | 18.28 | 5.65 | 37.31 | 10.20 | 8.84 |
| Other | 4.83 | 5.74 | 8.09 | 7.50 | 0 | 0 | 3.01 | 0 | 3.45 | 14.03 | 1.30 | 0.27 | 6.95 | 18.87 | 1.84 | 2.68 |

Absolute difference between percent abundance and model percent

| | Model percent | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Trichoptera | 10 | 8.73 | 16.64 | 51.73 | 0.50 | 9.66 | 10.00 | 42.25 | 9.63 | 50.29 | 7.09 | 7.73 | 8.12 | 11.54 | 4.58 | 18.15 | 57.90 |
| Ephemeroptera | 40 | 30.08 | 19.71 | 25.58 | 19.54 | 39.66 | 38.08 | 20.51 | 39.63 | 24.57 | 38.72 | 39.03 | 38.12 | 13.29 | 36.75 | 2.38 | 24.08 |
| Plecoptera | 5 | 5.00 | 5.00 | 4.82 | 5.00 | 5.00 | 5.00 | 4.93 | 5.00 | 4.91 | 5.00 | 5.00 | 5.00 | 4.16 | 5.00 | 4.89 | 5.00 |
| Coleoptera | 10 | 1.60 | 4.47 | 9.15 | 9.45 | 9.66 | 10.00 | 2.33 | 9.63 | 1.46 | 0.56 | 3.51 | 5.16 | 1.38 | 3.71 | 6.71 | 7.98 |
| Oligochaeta | 5 | 4.24 | 2.13 | 3.56 | 4.45 | 1.44 | 11.03 | 4.04 | 46.67 | 4.35 | 12.12 | 57.01 | 67.85 | 4.66 | 23.85 | 4.39 | 2.36 |
| Chironomidae | 20 | 54.81 | 18.93 | 6.70 | 40.44 | 72.54 | 62.05 | 3.45 | 27.21 | 8.45 | 34.10 | 6.95 | 1.72 | 14.35 | 17.31 | 9.80 | 11.16 |
| Other | 10 | 5.17 | 4.26 | 1.91 | 2.50 | 10.00 | 10.00 | 6.99 | 10.00 | 6.55 | 4.03 | 8.70 | 9.73 | 3.05 | 8.87 | 8.16 | 7.32 |
| sum difference | | 109.6 | 71.1 | 103.5 | 81.9 | 148.0 | 146.2 | 84.5 | 147.8 | 100.6 | 101.6 | 127.9 | 135.7 | 52.4 | 100.1 | 54.5 | 115.8 |
| sum difference X 0.5 | | 54.8 | 35.6 | 51.7 | 40.9 | 74.0 | 73.1 | 42.2 | 73.9 | 50.3 | 50.8 | 64.0 | 67.8 | 26.2 | 50.0 | 27.2 | 57.9 |
| 100 - sum difference | | 45.2 | 64.4 | 48.3 | 59.1 | 26.0 | 26.9 | 57.8 | 26.1 | 49.7 | 49.2 | 36.0 | 32.2 | 73.8 | 50.0 | 72.8 | 42.1 |
| Effect level | | mod | slight | mod | slight | severe | severe | slight | severe | mod | mod | mod | severe | none | slight | none | mod |