Louis Creek, British Columbia: A Biological Study Following Streambank Restoration

by

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ABSTRACT

Recent restoration has taken place along portions of the streambank of Louis Creek, British Columbia, including introduction of large woody debris and riprap (large boulders). To examine the benefits of restoring the streambank, samples were taken from both an unrestored, control site and a restored, treatment site. Site samples were taken by disturbing the top layer of sediment and collecting any macroinvertebrates present. Macroinvertebrates are one of the many important trophic levels that maintain a healthy stream ecosystem, providing food for fish populations and playing a key role in nutrient cycling of biomass and organic material.
Total abundance, total biomass, and order specific abundances were examined; order abundance includes isolating invertebrates from the orders Ephemeroptera, Plecoptera, Trichoptera and Diptera (EPT and EPT/D assessments).

There was both a 600% increase in total abundance, and a 500% increase in biomass within the treatment site compared to the control site (p-values 0.007 and 0.001, respectively). While total EPT had no significant increase in macroinvertebrates/m^2 (p=0.529), there was still a slightly larger number found in the treatment site when compared to the control site. Total EPT/D did not have significant difference between both the control and treatment sites (p-value 0.353). Furthermore, these results not only reveal the site's rapid post-stream restoration response and provide a benchmark for monitoring stream response over time but also provide an insight to the food available to local fish population.

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INTRODUCTION

The importance of protecting and enhancing fish habitat is becoming more and more evident with each passing year. An ever-changing climate (Hanson and Peterson 2014), increasing frequency of natural disasters including flooding, heat waves, heat domes, forest fires and increasing economic pressures (Gopal and Anbumozhi 2019), are proving to have increasing negative impacts on fish and fish habitat around the province of British Columbia (BC) (Department of Fisheries and Oceans (DFO) 2018). Hydrological impacts including droughts, snowpack loss and overall water availability (Hanson and Peterson 2014) are proving to be increasingly problematic for native fish populations.

Common practices including logging, increasing urban development causing habitat loss and recreational activities provide challenges to maintaining adequate fish habitat including both spawning and rearing habitat.

VALUE OF RESTORATIONS

The Department of Fisheries and Oceans released a report in 2010 stating that escapements (fish returning to spawning grounds) of the Fraser River stream-type chinook to Louis Creek has declined steeply to very low levels within the previous 6 years. Louis Creek has since been designated within the Fraser watershed priority area having had specific sites being identified as high priority for restoration efforts with a high potential for success (DFO 2019). Moreover, due to past removal of the riparian area, habitat restoration efforts and site monitoring efforts are vital for any potential future Louis Creek sites.

Prior to restoration efforts, the site’s bankside had significant erosion caused by a combination of both hydraulic and geotechnical failure. Hydraulic failure occurs when the flow exceeds the transportation of sediments in the outer bank causing water to degrade the toe of the bank (Babakaiff 1997). Geotechnical failure occurs when degradation of bank’s toe (undermining) has occurred in addition to the soil being saturated, allowing gravitational forces to exceed the strength of the resisting forces, causing bank collapsing towards the toe of the slope (Babakaiff 1997).

Moreover, salmon habitat needs include optimal water temperature and water quality, adequate depths, stream cover and substrate size (Bjornn and Reiser 1991). Removal of riparian vegetation can lead to increased stream temperatures, subsequent decreased dissolved oxygen, and erosion leading to poor and fine sediment accumulation (Bjornn and Reiser 1991). Current bankside stabilization techniques include bank re-vegetation (live staking, seeding), rock revetments (rip rap, rock toe keys, gabions), and bioengineering (fascines, brush mattress, tree and root revetments, vegetated geogrids) (BC Gov 2004).

SITE SELECTION

Louis Creek is located approximately 60 km North of Kamloops, BC, and drains into the North Thompson River (Pehl 2009). The valley bottom of Louis Creek is mostly privately owned, has high agricultural water demands and has seen logging and agricultural activities such as grazing that have had adverse effects on stream health and channel stability (Pehl 2009).

Within Louis Creek, populations of *Oncorhynchus kisutch* (Coho salmon), *Oncorhynchus tshawytscha* (Chinook salmon), *Oncorhynchus mykiss* (Rainbow trout) have all been found (Henderson and Seidler 1999, North Thompson Indian Band 2002).

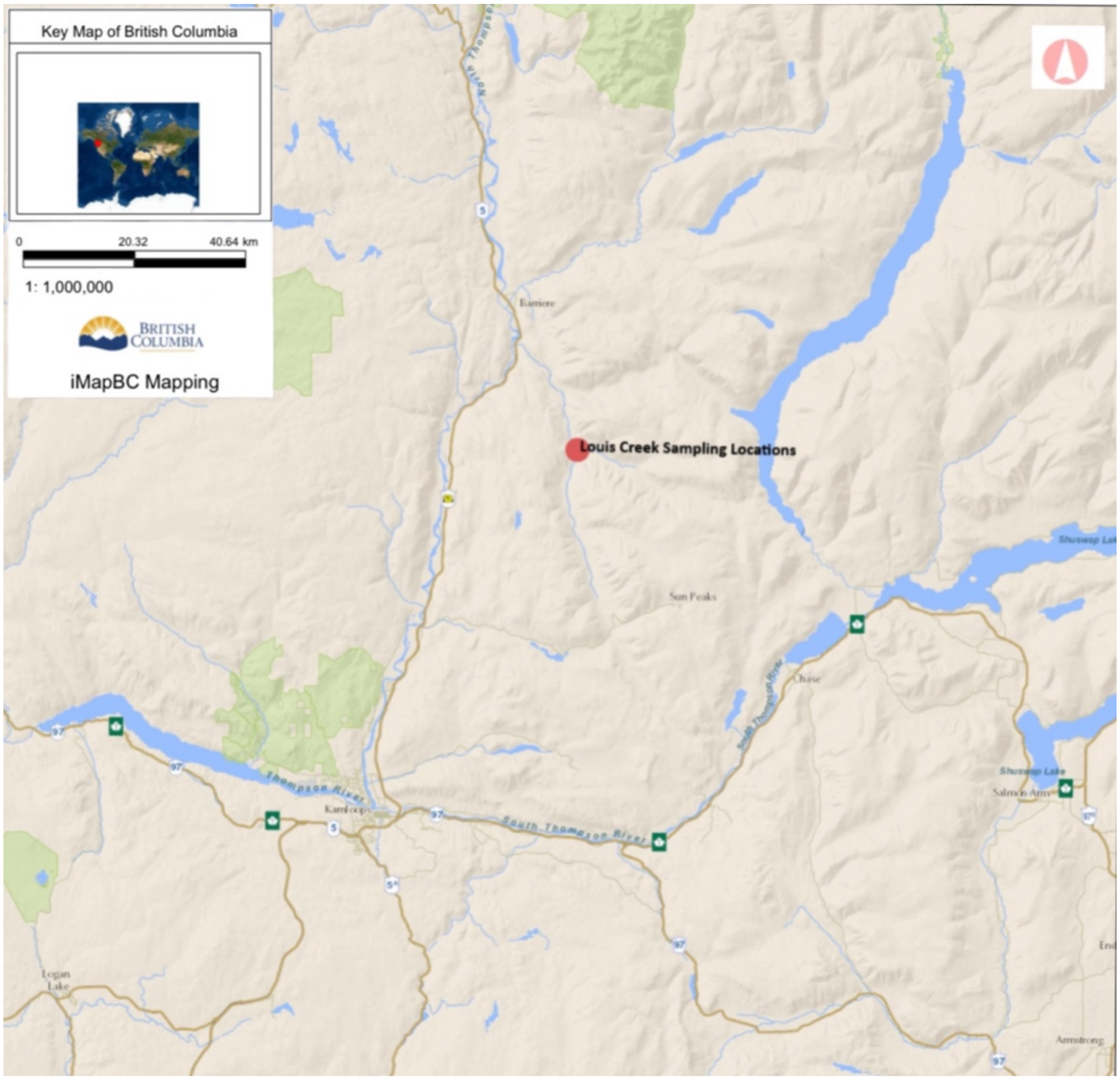

FIGURE 1. Louis Creek study sampling locations in relation to Kamloops, British Columbia. iMap BC.

This study examined two sites, the first being a control site that has been subjected to significant erosion and has not undergone any restoration efforts, pictured in Figure A 1. The second site is a treatment site, that has undergone immense restoration efforts including introduction of logs, bank stabilization by boulder additions (often called riprap) and planting of vegetation (grasses), as pictured in Figure A 2. It should be noted that for the purposes of this study, the two sites studied for this project had been predetermined by Mike Wallis, a project consultant and engineer.

Previous land management saw the removal of the riparian cottonwood trees, causing the stream to bear the consequences of this decision. Removal of the cottonwood trees decreased both litter and organic material from entering the stream including removal of potential fallen trees that would have created pools, sedimentation, etc. Harvesting of the riparian area also led to decreased inputs of terrestrial invertebrates (food for fish), and decreased shade, thereby increasing overall stream temperature. Though, the impact that is directly correlated with this study is the decreased bankside stability due to removal of deep roots associated with cottonwood trees, thus increasing the risk and/or impact of bankside erosion.

FIGURE 2. Control and treatment sites of Louis Creek, BC, 2021. Image: Google Earth, drawing: Jessica McQueen.

OBJECTIVES

The objectives of this study were to

- Describe the aquatic invertebrate community of both a resorted and unrestored stream bank section to determine whether the restoration efforts are effective in respect to returning the aquatic diversity of Louis Creek, and
- to provide baseline data of the present invertebrate community for future recovery and restoration – looking at community level change.

METHODS

SAMPLING SITES

As previously mentioned, the location of both sampling sites had been previously determined by the private consultant, Mike Wallis, who was responsible for the physical restoration of the treatment site. Moreover, the control site is located downstream in respect to the restored, treatment site. Twenty meters was measured along the concave (outer) stream bank via measuring tape and flagged at both ends with both a wooden stake and flagging tape. The middle of the section was measured and flagged (the ten-meter mark) at the apex of the concave bank. Every sampling area was outlined with a painted wooden stake, with the first stake at meter-zero, and stakes placed every two meters apart until ten sampling units had been marked; each sampling unit measured one meter in width and two meters in length, each sample unit totaling 2 m^2. Along the edge of the stream, the samples were taken beginning downstream moving upstream to create as little disturbance within the sampling sites as possible.

It should be noted that the apex of the treatment site was located, and twenty meters were measured out; however, the apex was moved further upstream due to unsafe water depths. Should the samples had been taken where the true apex was located, the water would have been too deep to properly disturb the substrate with hands which would have impacted the sample collections as the sampling technique would have been improper. Additionally, safety of those in the stream had to be considered, as the water was more than chest deep, had faster currents and higher energy, but increased sediment which made the ground unstable; with those considerations in mind, the sampling sites were moved slightly upstream (Figure 3).

FIGURE 3. Louis Creek restored, treatment bankside site displaying including length and width of one sampling area including display of the altered travelling leg sampling technique within one sampling area. Base image: Google Earth, drawing: Jessica McQueen.

DATA COLLECTION

Prior to sampling Whirl-Pak® bags had been labelled both internally via waterproof paper (marked with pencil) and externally Sharpie. Both internal and external labels included the date, location, sample location (treatment or control site) and sample number. Total number of samples was correlated with the number of sampling units; therefore, ten samples were taken at each site. Samples were collected by means of a modified travelling leg technique, an altered version of the "3-minute travelling kick method" (Environment Canada 2012) at both the control and restored sites. A lead team member dragged a 250 μm mesh D-frame net in a zigzagged motion, while an assistant, who was wearing arm waders, walked in front of the leader (while still facing the leader), disturbing and brushing over any substrate, organic material, debris, etc.

to dislodge present aquatic invertebrates. The D-frame net was held close to the area being disturbed to ensure that both the invertebrates and substrate were captured. When zigzagging the D-frame net in an upstream direction within each one-by-two-meter sampling units, there was a thirty-second time limit that was monitored by a second assistant. Should there be any obstacles or the need to pause for any reason, the timer was stopped, and the net was removed from the water until both the leader and both assistants were prepared to continue. Note the second assistant also aided with transferring samples collected from the sampling units to the near-by processing tables.

All material caught within the net was transferred to a basin to have any unwanted materials such as organic material cleaned off and removed, preventing any potential loss of invertebrates. The sample was run through a 250 µm sieve to remove excess water and transferred to a Whirl-Pak® that was filled with ethanol for preservation. Any remaining invertebrates within the basin or sieve were removed with a spoon, water or by hand. Stream width and depth measurements were taken after the samples had been collected to not disturb or alter the sampling site.

SAMPLE SORTING

Samples collected were sorted through with help of an LED magnifying bench lamp to remove and isolate any captured invertebrates to be further placed into small vials. This involved rinsing off any fine sediment over a 250 µm sieve, placing small amount of sample in a shallow basin to remove all invertebrates collected, and place them in a vial that correlated to the sample number. Just as sampling bags had been labelled, each vial was labelled both internally and externally with the corresponding sample number from each respective sampling site (manor as previously mentioned). Invertebrates that had been isolated from the sediment were then identified into respective taxa, specifically down to family level. Taxa identification was accomplished by primary use of Merritt et al (2008) and Thorp and Covich (1991), under a Leica MZ6 modular stereomicroscope. Moreover, family abundance was recorded for each sample; those that could not be identified to family level were recorded to the order level. Length measurements were taken of each invertebrate and recorded for biomass analyses, providing quality control of both the taxa identifications and abundances initially recorded. It should be noted that invertebrates in the pupa life-stage were neither counted towards total biomass and abundance nor total EPT and EPT/D analyses.

DATA ANALYSIS

Data collected was arranged for analysis using Excel and was analyzed with the SPSS (IBM Corp (SPSS), 2019). All samples (control and treatment) were tested for normal distribution and equal variances (Tables B 1 and B 2 respectively). Data not normally distributed (p-value >0.05) or had unequal variances (p-value <0.05) were analysed with a non-parametric test (Mann-Whitney U test); data that was both normally distributed (p-value >0.05) and had equal variances (p-value >0.05) was tested with a parametric test (t-test).

BIOMASS

To begin the process of determining the site biomass estimations, each invertebrate was measured to ocular units and converted to length (mm) via conversion factors, as seen in Table C 1. The length-mass regressions, as seen in Table C 2, were then used to convert the lengths of each invertebrate to weight (grams). The length-mass regressions are completed by using the power model $M = aL^b$ (Benke et al 1999). This equation sees the use of constants (a & b) for each respective order and family of invertebrates identified as well as each invertebrate's individual length [mm] measured (L) to find mass (M) (Benke et al 1999). Should there not be a regression equation for a specific family, then the equation for its respective order was used in place (Table C 2).

The biomasses of each insect within their respective samples were totaled and divided by the respective sampling area ($2\ m^2$) to get g/m^2. The ten samples of each the control and treatment sites were added and averaged to determine final biomass values. Finally, just as the total abundance and total taxa abundances, the data was tested for normality and equal variances and then further analyzed using a parametric or non-parametric test, normality results dependent.

PRELIMINARY ANALYSIS RESULTS

Total abundance, total biomass, total Ephemeropteran (E), Plecopteran (P), and Trichopteran (T) also displayed as total EPT, and finally total Ephemeropteran, Plecopteran, Trichopteran and Dipteran (D), displayed as total EPT/D all had normal distribution and equal variances tested. Both normality and equal variances are tested to determine whether a parametric or non-parametric test was to be used. Should the data be both normally distributed and have equal variances, then a parametric test was to be used; data not meeting one or both assumptions (having normally distributed data and equal variances) would be analyzed with a non-parametric test.

Furthermore, only total biomass had equal variances (p-value 0.03, Table B 1), however did not have normally distributed treatment site data. While normally distributed data was seen within the control site for all four categories (p-values >0.05, Table B 2), the treatment site had non-normally distributed data (p-values <0.05). Both ETP and EPT/D did not have equal variances (p-values 0.073 and 0.319 respectively), nor had normally distributed data (p-values 0.05 and 1.66 respectively). None the less, due to all four categories not meeting the two parametric test assumptions, the Mann-Whitney U non-parametric test was used.

RESULTS

During this study, a total of 1279 macroinvertebrates were collected, representing 9 orders, 5 of which orders are non-insect (i.e., *Nematoda*, *Mollusca*, etc.), and a total of 24 families. There was a significant increase from 9.2 invertebrates/m^2 to 54.75 invertebrates/m^2 within the control and treatment sites respectively (p-value 0.007, Table B 3); this significant increase of invertebrates/m^2 is displayed in Figure 4.

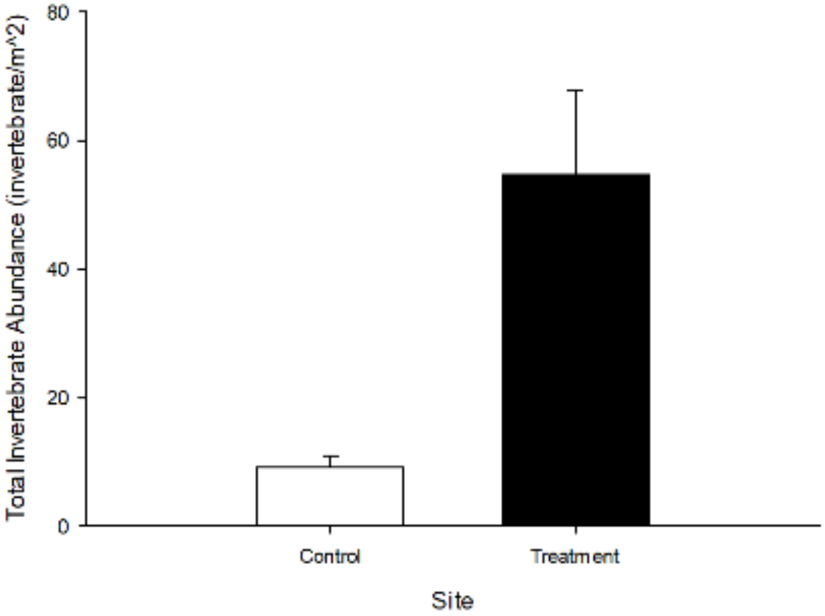

FIGURE 3. Mean total abundance (invertebrate/m^2) of both the control and treatment sites, with error bars that display 95% confidence intervals. Louis Creek, 2021.

There was a significant increase in total biomass from a total 78.32 g/m^2 in the control site compared to the 376.75 g/m^2 from the treatment site, displayed in Figure 5 (p-value 0.001, Table B 3).

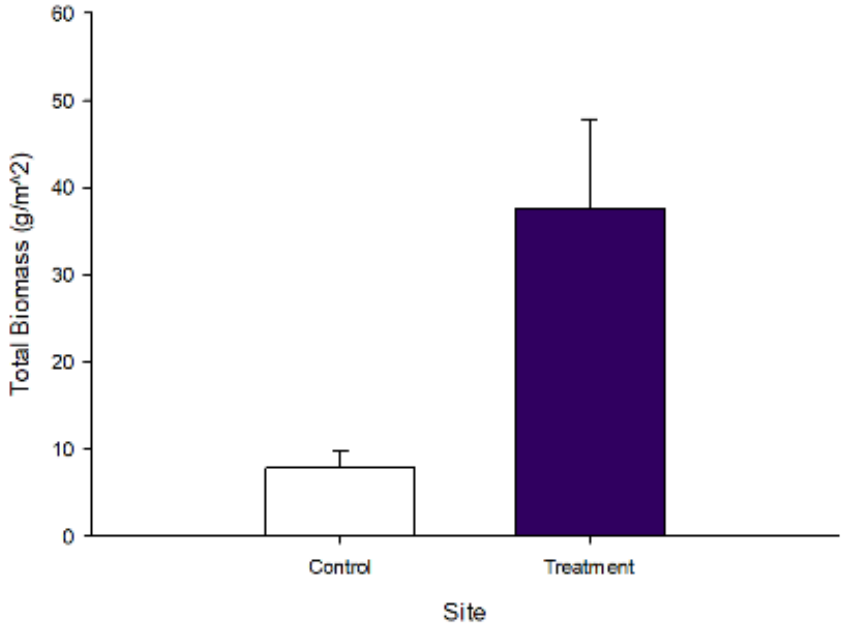

FIGURE 4. Total biomass (invertebrate/m^2) of both the control and treatment sites, with error bars that display 95% confidence intervals. Louis Creek, BC 2021.

While Figure 6 appears promising by showing the control site 1.2 invertebrates/m^2, and the treatment sites 4.65 invertebrates/m^2, the difference in control and treatment site data were not significant (p-value 0.529, Table B 3).

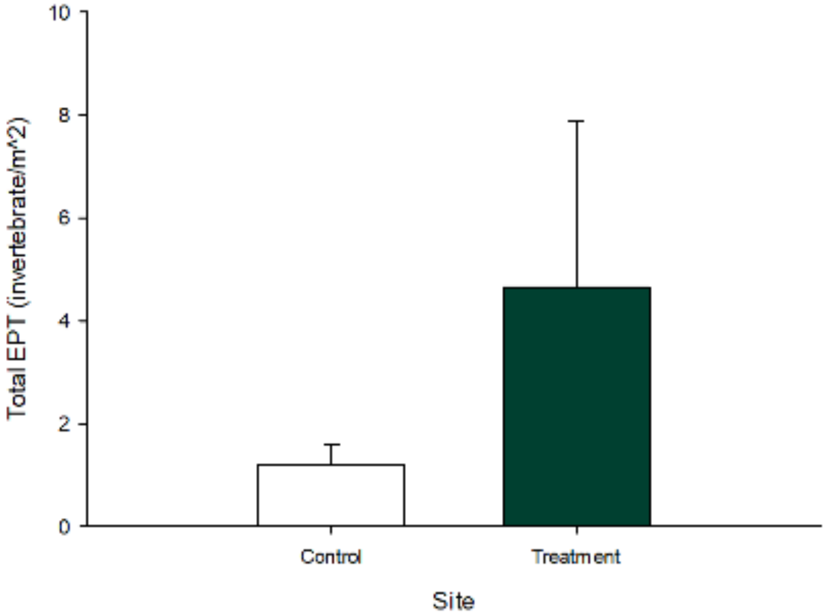

FIGURE 5. Mean total EPT (total EPT invertebrates/m^2) of both the control and treatment sites, with error bars that display 95% confidence intervals. Louis Creek BC, 2021.

Unlike all other invertebrate analyses, EPT/D was the only analysis that saw the control site having a larger mean compared to the treatment site, as displayed in Figure 6. However, the data was not significantly different between the control and treatment sites (p-value 0.353, Table B 3).

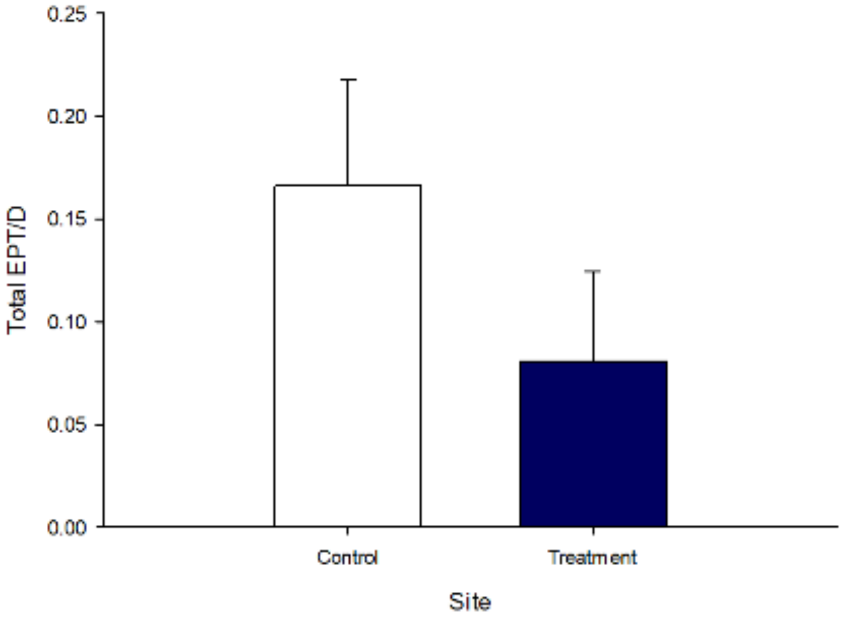

FIGURE 6. Mean total EPT/D of both the control and treatment sites, with error bars that display 95% confidence intervals. Louis Creek, BC 2021.

DISCUSSION

In many cases, when adequate funding is available and specifically allocated to post restoration monitoring, success of stream restoration is qualified in part by use of the present stream aquatic macroinvertebrate communities. Aquatic invertebrate communities are often studied as they reflect an integration of both biotic and abiotic influences including water quality (pollutants), substrate presence and more (Testa et al 2010).

Unfortunately, the lack of streambank metrics for assessing the stream's biological condition and water pollution became evident once data analysis began. While common tests such as the Benthic Index of Biological Integrity (B-IBI) or a Hilsenhoff family-level index (FBI) could have been used, they are structured more specifically towards riffles and pool-riffles rather than streambanks. The lack of applicable indices limited how data was analyzed, leading to abundance analyses and total biomass estimations.

Moreover, this project featured bankside stabilization by use of subangular boulders, addition of bare logs (no branches), and vegetation plantings including grasses during the winter months. As previously mentioned, both the control and treatment site's bankside had significant erosion caused by both hydraulic and geotechnical failure. The riprap not only helped control bankside erosion, provides resistance for rock movement within the revetment (Ministry of Environment, Lands and Parks 2000) but also provided support for the large wood. Within the months of the streambank restoration, the restored, treatment site already displayed improvements from recent restoration efforts when compared to the unrestored, control site. The treatment site is showing improved aquatic macroinvertebrate community health based on increased invertebrate abundance and biomass estimates.

TOTAL ABUNDANCE

While there was an expectation to see increased abundances from the treatment site, such a large difference between sites was unexpected. The results, however, support the initial prediction that there would be an increase between both sites due to streambank restoration. The treatment site saw an increase of just under 600% invertebrates/m^2 when compared to the control site, demonstrating the positive impact site restorations can have on the invertebrate community. The difference in both site's total abundances is shown specifically by the 184 total collected, sorted, and identified macroinvertebrates from the control site samples, while in contrast, a total of 1095 invertebrates from the treatment site samples collected.

Such stark increases in treatment site abundance, while unexpectedly high, reinforce the value of streambank restorations. Such an increase in total abundance could be partially due to the addition of large wood (LW). LW provides food, places to attach to especially during high flows, and predator avoidance, and may be the only suitable stable material available to macroinvertebrates (Testa et al 2010). A lack of LW within the sampling sites is partially due to both past and current land use management, again, with the removal of the cottonwood trees. Portions of the streambank adjacent to agricultural range have been significantly affected by the erosion; certain portions having the bank and the fence on top to collapse into the stream, potentially allowing stream access to livestock which causes additional damage when left unmanaged. Even without direct access to the stream, having fencing within the riparian area allows livestock to eat the riparian vegetation especially the new growth, preventing vital tree and shrub growth. Riparian vegetation such as shrubs and trees prevent additional erosion, provide cover from predators for the fish, provides shade and eventually become woody debris or in-stream wood which improves fish habitat by creating pools and spawning areas (Streamside Native Plants). Additionally, a healthy riparian area provides food, habitat, and predator cover for many species outside of the stream (Streamside Native Plants); while these are vital to the ecosystem, being unable to establish riparian vegetation due to collapsing fences and having cow access will not see any such ecosystem benefits.

TOTAL BIOMASS

Just as with total abundance, while there was an expectancy of biomass to be significantly different, such a large difference between both sampling sites was unexpected. Furthermore, within the entire control site sampling area (20 m^2), there was a total of 78.32 g/m^2, whereas the treatment site saw an increase just under 500% increase in biomass compared to that of the control site, with a total of 376.75 g/m^2. It should also be noted that the aquatic macroinvertebrates collected from the treatment site were observed to be larger in overall size when compared to the invertebrates collected from the control site. While this was an observation, future studies would be beneficial to quantify such observations, however, in this study, there was not enough time to measure potential differences.

Biomass, which is also often characterized as food available for fish, is important to examine as the amount of available food is not only critical for fish production, density and growth but is also a key contributor to a site's carrying capacity (Bjornn and Reiser 1991). The addition of LW allowing for increased coarse particulate organic matter (CPOM) in the treatment site, may have potentially resulted in the increased biomass. LW and in-stream CPOM provide food for shredders and filter feeders, which is supported by a report by Bjornn and Reiser (1991) stating that aquatic invertebrate production is dependent on the availability of in-stream CPOM. However, any increase in invertebrate abundance, whether the increase is due to any of the aforementioned factors, would have led to the subsequent increase in biomass. Therefore, the addition of riprap, LW and riparian area planting leading to increased in-stream CPOM may have ultimately led to the treatment site's increase in total biomass. Furthermore, this study displays a rapid response to streambank restoration, increasing aquatic invertebrate abundance and most importantly, providing increased food for fish.

ORDER ABUNDANCE

When examining taxa abundance, there was a total of 10 EPT families collected between both sampling sites. The treatment site saw increased abundances of all Ephemeroptera, Plecoptera and Trichoptera taxa when compared to the control site; the difference is expressed by the larger mean value of 4.65 invertebrates/m^2 compared to the control site's mean of 1.2 invertebrates/m^2.

Typically, Ephemeropteran, Plecopteran, and Trichopteran orders are used as bioindicators as they have a low tolerance to water pollution and are affected by changes in land use, habitat loss, and nutrient enrichment (Poulton and Tao 2019). EPT results for both sampling sites did not meet the initial expectations and had no significant differences in taxa abundances between sampling sites (p-value = 0.514). The lack of a significant difference is most likely due to the overall lack of EPT abundances, specifically Plecopteran and Trichopteran orders in both sites. While it is difficult to definitively say why the data was not significantly different, it may be since the control site remains in a state of disturbance, and it has been less than a year since restoration efforts have taken place for the treatment site.

EPT/D analysis revealed non-statistically different data between sampling sites (p-value = 0.353), therefore does not support the initial prediction of having a significant difference between sites. This contradicts the initial assumption of having a higher ratio of EPT/D in the treatment site compared to the control site. Neither means of ETP/D for the control nor the treatment site reached over a value of 1 which was surprising; there was a higher mean value in the control site (0.166) compared to that of the treatment site's (0.08). When re-examining the initial data, this result becomes less surprising given the small abundance of Ephemeropteran, Plecopteran, and Trichopteran taxa collected compared to the large number of Dipteran invertebrates collected, especially the treatment sites data. When examining the control site, the ratio of EPT to D invertebrates was 24 to 93 respectively; the treatment site saw a ratio of 93 EPT to 761 D invertebrates collected.

Mayflies, stoneflies, and caddisflies, from orders Ephemeroptera, Plecoptera, Trichoptera respectively are all pollutant sensitive insects, while Dipteran taxa such as chironomids are pollutant tolerant (Chadde). Invertebrates such as mayflies, stoneflies and caddisflies will take longer to inhabit a newly restored site compared to a more tolerant invertebrate such as chironomids due to their habitat sensitivity. Pollutant sensitive organisms require higher percentages of dissolved oxygen, cooler water, and a more neutral pH, while tolerant organisms can live in lower oxygen, non-neutral pH, and warmer waters to a degree (Courtney and Clements 1998, Poulton and Tao 2020, Thorp and Rogers 2009). Moreover, the lack of EPT invertebrates may be due to the current level of site disturbance and lack of time for the restoration efforts to take full effect. This could also explain why there is a higher number of EPT/D invertebrates in the control site compared to the treatment site. The ratio of EPT compared to D invertebrates is much larger within the control site due to Dipteran invertebrate's ability to tolerate less than ideal conditions. This ratio is more balanced in the treatment site, as the restoration has improved habitat conditions allowing for EPT invertebrates to grow in numbers.

It is important to remember that the treatment site had undergone sampling within a year of the restoration efforts. As previously mentioned, this is a very newly restored site, and while EPT and EPT/D may not yet be significantly different between sites, restorations, specifically vegetation often take many years to become fully functional and healthy sites once again (Baird et al 2015). With that in mind, the stark difference in significant data between both sampling sites for total abundance and total crude biomass show the immense potential of streambank restorations. While this is just the first year since the restoration, data collected from the 2021 sampling year will provide a benchmark for the upcoming sampling, giving an additional insight to macroinvertebrate community response to restorations over time.

LIMITATIONS

While both the control and treatment sites had high levels of fine sediment, the treatment site had larger sediment (increased amount of gravel, yet still had higher amounts of clay/silt), while the control site had more uniformly sized fine sediment. The high level of fine sediment clouded the water, even after rinsing the samples many times, and made isolating invertebrates challenging.

Additionally, as previously mentioned, the treatment site had deeper water, constraining the location along the bankside of which the samples were taken. Subsequently, the midpoint of the sampling areas was moved slightly upstream to allow for proper sampling technique. Pooling at the treatment site required the stakes used for marking the sample locations to be moved three meters closer to the point bar (inner portion of the stream bend) as not only was the water too deep to see the stakes, but the high amount of very fine sediment prevented the stakes from remaining in place. Sampling took place three meters into the stream from where the stakes were placed, in the originally designated sampling location.

UPCOMING RESTORATION SUGGESTIONS

Future restorations of portions of the Louis Creek streambank could see a few areas of restoration implementation being improved upon. For example, planting a combination of shrubs, trees, and grasses rather than exclusively planting grasses. While grasses typically grow quite quickly, adding vegetation to a site, they tend to have shallow root systems, and do not provide much bankside stabilization compared to the root systems of trees and shrubs. Grasses also lack the ability to produce large amounts of litter that a tree or shrub could produce, limiting in-stream CPOM and shade in the stream. Additionally, while the addition of LW is critical for many reasons, perhaps using log revetments rather than the tree stakes (pictured in Figure A2) may be more beneficial. Log revetments create resting places, add CPOM, provide shade, pool formation and subsequent sediment accumulation.

FUTURE RESEARCH

Increased research surrounding streambanks (especially in Western North America) is needed in respect to measuring streambank health including order diversity, richness, and responses to pollution and/or disturbance. While instream structures such as tree revetments, riffles and pools are important for fish populations, streambank structures provide habitat for the aquatic invertebrates that the fish, as well as other species such as amphibians, eat. Future research examining invertebrate response to solely restoring riparian vegetation and subsequently increasing in-stream woody debris and LW could be of value due to their importance in a fish's diet.

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APPENDICES

APPENDIX A. ADDITIONAL SITE INFORMATION

FIGURE A 1. Untreated, control site of Louis Creek, BC. Red stakes display the beginning of each sample area, ten stakes in total outlining the entire 20 m^2 sampling area. Image: Jessica McQueen, 2021.

Appendix A. Additional Site Information Cont.

FIGURE A 2. Restored, treatment site of Louis Creek, BC. Red stakes display the beginning of each sample area, ten stakes in total outlining the entire 20 m^2 sampling area. Stakes moved 3m inwards due to water depth and instability when placed in fine sediment. Image: Jessica McQueen, 2021.

APPENDIX B. STATISTICAL ANALYSIS & LENGTH-MASS REGRESSION SUPPORTING INFORMATION

TABLE B 1. Test for equal variances for total abundances, total EPT, total EPT/D and biomass from Louis Creek, BC 2021

	Significance (P) value
Total Abundance	0.03
Biomass	0.56
Total EPT	0.073
Total EPT/D	0.319

TABLE B 2. Test for normality for total abundances, total EPT, total EPT/D and biomass from Louis Creek, BC 2021

	Site	Significance (P) value
Total Abundance	Control	0.403
	Treatment	0.201
Biomass	Control	0.227
	Treatment	0.003
Total EPT	Control	0.05
	Treatment	<0.001
Total EPT/D	Control	1.66
	Treatment	<0.001

Appendix B. Statistical Analysis & Length-Mass Regression Supporting Information Cont.

TABLE B 3. Test results from non-parametric tests for total abundance, total EPT, total EPT/D and biomass for Louis Creek, BC 2021

	Test used based on normality p-value	Significance (P) value
Total Abundance	Mann-Whitney U Test	0.007
Total biomass	Mann-Whitney U Test	0.001
Total EPT	Mann-Whitney U Test	0.529
Total EPT/D	Mann-Whitney U Test	0.353

APPENDIX C. LENGTH-MASS REGRESSION SUPPORTING INFORMATION

TABLE C 1. Magnification levels and respective conversion factor when measuring length (mm) of invertebrates

Magnification	Conversion Factor
0.63	0.1429
0.8	0.1176
1.0	0.09523
1.25	0.07692
1.6	0.06060
2.0	0.04878
2.5	0.03846
3.2	0.03077
4.0	0.02409

Appendix C. Length-Mass Regression Supporting Information Cont.

TABLE C 1. Length-weight regression equations for biomass conversion from Louis Creek, BC

Order	Taxon	Regression Equation (a*L^b)	Reference
Amphipoda	Order level	$0.0058*L^{3.015}$	Benke et al 1999
Coleoptera	Order level	$0.0077*L^{2.91}$	Benke et al 1999
	Dytiscidae	$0.0618*L^{2.502}$	Benke et al 1999
	Elmidae	$0.0074*L^{2.879}$	Benke et al 1999
	Haliplidae	$0.0271*L^{2.744}$	Benke et al 1999
Trichoptera	Order level	$0.0056*L^{2.839}$	Benke et al 1999
	Brachycentridae	$0.0083*L^{2.681}$	Benke et al 1999
	Hydropsychidae	$0.0046*L^{2.926}$	Benke et al 1999
Ephemeroptera	Order level	$0.0071*L^{2.832}$	Benke et al 1999
	Ameletidae	$0.0077*L^{2.588}$	Benke et al 1999
	Baetidae	$0.0053*L^{2.875}$	Benke et al 1999
	Ephemerellidae	$0.0103*L^{2.272}$	Benke et al 1999
	Leptophlebiidae	$0.0047*L^{2.686}$	Benke et al 1999
Odonata	Order level	$0.0078*L^{2.792}$	Benke et al 1999
Plecoptera	Order level	$0.0094*L^{2.754}$	Benke et al 1999
	Chloroperlidae	$0.0065*L^{2.724}$	Benke et al 1999
	Nemouridae	$0.0056*L^{2.762}$	Benke et al 1999
	Perlidae	$0.0099*L^{2.879}$	Benke et al 1999
Hemiptera	Order level	$0.0108*L^{2.734}$	Benke et al 1999
Diptera	Order level	$0.0025*L^{2.692}$	Benke et al 1999
	Ceratopogonidae	$0.0025*L^{2.469}$	Benke et al 1999
	Chironomidae	$0.0018*L^{2.617}$	Benke et al 1999
	Empididae	$0.0066*L^{2.436}$	Benke et al 1999
	Tabanidae	$0.005*L^{2.591}$	Benke et al 1999
	Tipulidae	$0.0029*L^{2.681}$	Benke et al 1999
Annelida	Oligochaeta	$0.008*L^{1.888}$	Miyasaka et al 2008
Mollusca	Gastropoda	$0.137*L^{2.355}$	Eklof et al 2017
	Bivalvia	$0.069*L^{2.820}$	Eklof et al 2017
Ostracoda	Order level	$0.1738*L^{4.2678}$	O'Gorman and Emmerson 2010
Nematoda	Order level	$0.0021*L^{2.395}$	O'Gorman and Emmerson 2010