Restoration Planning for Urban Salmonid Habitat: Effects of Stormwater Runoff on Water Quality and Benthic Invertebrates

by

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Abstract

Restoration of salmonid habitat has been completed in many urban areas; however, the success of these projects may be limited without consideration of water quality. Urban watersheds are affected by stormwater runoff which transfers toxic substances such as heavy metals, hydrocarbons, and fine particles from impervious surfaces into streams. Previous research has documented impacts of stormwater causing premature death in spawning coho (*Oncorhynchus kisutch*), and related extent of impervious surfaces to impacts on benthic invertebrates. This research aims to expand our knowledge on the effects of stormwater runoff on water quality and benthic invertebrate communities, and make recommendations for restoration of Mosquito Creek, in North Vancouver, British Columbia. Stream water quality was monitored, site habitats were assessed, and impervious surfaces were mapped. Benthic invertebrate samples were collected and analyzed for abundance, diversity, and pollution tolerance, comparing upstream and downstream of a stormwater inflow and two sites on a reference stream. Average water quality measurements showed minor impacts related to elevated temperatures. However, benthic invertebrate metrics revealed chronic water quality issues, reflecting cumulative impacts. Pollution tolerance index and abundance were reduced at the downstream Mosquito Creek site suggesting impacts from the stormwater inflow, while the Ephemoptera, Plecoptera, Trichoptera (EPT) to total ratio and overall stream health (Streamkeepers Site Assessment Rating) were significantly lower at Mosquito Creek overall suggesting watershed impacts from impervious surfaces and point-source pollution events. Restoration recommendations including a rain garden are discussed to improve water quality for salmonids.

Keywords: Restoration; urban streams; salmonids; benthic invertebrates; water quality; stormwater

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Chapter 1. Introduction

Stormwater runoff is an important consideration in the restoration of stream habitats in urban environments around the world, which have been heavily impacted by logging, and subsequent development of the watershed. Salmonids in British Columbia (BC), and throughout the Pacific Northwest region, including several commercially and culturally important species, have experienced significant population declines in the past several decades (Price et al., 2017). Declines have continued in 2020, with sockeye salmon (*Oncorhynchus nerka*) returns the worst on record at 283,000, a substantial decrease from the 9.6 million annual average between 1980 and 2014 (PSC, 2020; Grant et al., 2018). There are numerous factors causing these declines including overexploitation, decreased marine survival, habitat loss, fish farms, and climate change. Land conversion in urban watersheds is also an increasing concern, as salmonids are exposed to variable flows and non-point source pollution in the form of runoff from roads, parking lots, and industrial and residential areas. Historically, the traditional rainwater management strategy was to remove runoff as quickly as possible from developed areas, so designs were made to be efficient in collecting and discharging rainwater to receiving waters (BC MWLAP, 2002). In addition, developments often encroach into streamside vegetation, which previously buffered some of this runoff. Episodic, point-source pollution from oil spills and other contaminants have also become the norm in many urban creeks (Granger, 2019; CBC, 2019).

Restoration of salmon habitat has been completed in many urban areas; however, the success of these projects may be limited without a full consideration of the abiotic conditions such as water quality. This underlying issue has the potential to limit recovery of fish populations and restoration success, due to the toxic effects of urban runoff on fish and deleterious effects on stream invertebrates, which are an important prey item for salmonids. For example, in a review of typical urban channel reconstruction projects that focused on reconfiguring the channel and installation of log complexes, benthic invertebrate assemblages did not improve as stormwater was not addressed (Violin et al., 2011). Urban runoff contains a complex mixture of many toxic substances such as heavy metals, polycyclic aromatic hydrocarbons (PAHs), other organic hydrocarbons, and fine particles (Spromberg et al., 2016; Feist et al., 2011). Rain mobilizes these non-point source pollutants from roads, parking lots, and industrial and residential areas and transfers them to aquatic habitats.

There are many potential toxicological effects of runoff on fish due to the variety of chemicals involved. PAHs are disruptive to the fish cardiovascular system, while metals affect the respiratory and osmoregulatory functions of the gills (Brette et al., 2014; Niyogi and Wood, 2004). PAHs can also cause reduced growth and lipid stores in juvenile chinook (*O. tshawytscha*), which increases their risk of predation (Meador et al., 2006). Toxic stormwater has also been implicated in the high rates of premature death in spawning coho (*O. kisutch*) (Spromberg et al., 2016). As fall migration coincides with the period of high seasonal rainfall and stormwater runoff in the Pacific Northwest, there is an acute risk of toxic exposures (Spromberg et al., 2016). Peter et al. (2018) determined that tire wear particle leachates were most chemically similar to waters that induce the coho mortality syndrome indicating that tire wear particles may be a significant contributor. Key contaminant groups identified within the mortality signature were (methoxymethyl)melamine compounds, bicyclic amines, and long-chain glycols and ethoxylates (Peter et al., 2018). Seminal research on the candidate compounds has revealed that 6PPD-quinone, found in tire rubber, is the primary toxic contaminant (Tian et al., 2020). The precursor compound 6PPD is used in tires to provide protection from ozone in car exhaust.

Runoff pollution also affects benthic invertebrate communities in streams. These small aquatic organisms are an excellent biological indicator as many species are very sensitive to contaminants, they are relatively sessile, reflecting the local site conditions, and they are highly responsive to changes in their environment (Branton et al., 2006; Page et al., 2008). For instance, EPT Richness is a key metric used to estimate stream health, which incorporates the pollution sensitive taxa from the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). Each of these orders can also be represented as their own richness metric as each has specific tolerances to oxygen, temperature, and habitat complexity (Branton et al., 2006). Percent Chironomidae (lake flies), which tends to increase when disturbance increases can also be used to determine impacts to the stream, as many genera of this family are highly tolerant and opportunistic (Branton et al., 2006). In combination with measures of benthic invertebrate abundance, family richness, % dominant taxa, and other measures of diversity and evenness, these metrics can collectively be used to determine if impacts to water quality and stream invertebrate biota are occurring.

Previous research in 1999-2006 determined the benthic index of biological integrity (B-IBI) scores for several creeks in the Lower Mainland and found that in North Vancouver, Mosquito Creek had a B-IBI score of 22.0-29.0 and Wagg Creek had a score of 11.5-19.0. These values were lower than in other less urbanized areas and compared to controls in reservoir catchment areas (Page et al., 2008). This suggests that there may be impacts on invertebrate communities and salmon already occurring in the Lower mainland and North Vancouver. Further work by the City of North Vancouver determined a B-IBI score of 27.5 in Mosquito Creek, 22.0 in Mackay Creek, and 19.0 in Wagg creek in 2014 (City of North Vancouver, 2016). In this study, I examined the invertebrate communities in the downstream portion of Mosquito Creek, in the District of North Vancouver, during May and September 2020.

The effects of urban runoff are also dependent on the extent of impervious surfaces in the watershed as this influences the amount of stormwater and collection of pollutants. Impervious surfaces, such as rooftops, paved roads, parking lots, and asphalt, allow very little to no water to infiltrate through them and collect pollutants on their surfaces. This is a particular concern in urban areas of Metro Vancouver, which are expanding with increasing development and land conversion. Previous studies have found relationships between invertebrate metrics and level of urbanization in major urban centers across the continental US (Cuffney et al., 2010). A maximum of 5-10% of impervious surfaces has been suggested to protect aquatic ecosystems; however, results have shown that these levels may already change invertebrate assemblages by 13-33% (Cuffney et al., 2010). Other reviews place the threshold for biological degradation at 10-20% impervious surfaces (Metro Vancouver, 2019; Paul and Meyer, 2001). One study has also found a correlation between extent of impervious surfaces within a watershed and the severity of coho mortality (Feist et al., 2011). Reports from Metro Vancouver indicate that impervious surfaces cover 20% of the regional land area in Metro Vancouver, and is likely increasing (Metro Vancouver, 2019). The District of North Vancouver has 11% impervious surfaces, while the City of North Vancouver has 65% impervious surfaces (Metro Vancouver, 2019).

Hydromodification, the alteration of natural flow regimes, including channelization, straightening, and hardening of banks also influences the impact of non-point source pollution in urban streams. While the area of impervious surfaces affects the amount of pollution collected, hydromodification, such as channelization, can affect the rate of non-point source pollution by increasing the timing and delivery of pollutants entering the stream (USEPA, 2007). For instance, the "first flush" in the fall results in a rapid release of pollutants as water flows quickly into the system.

In addition to conversion of impervious surfaces at the watershed level, riparian buffers are often encroached on by development, which limits natural filtration processes. An extensive review of riparian buffer widths found that at least 30 m of streamside forest vegetation are needed for effective nitrate removal and sediment trapping functioning (Sweeney and Newbold, 2014). For instance, with an average water flux the nitrogen removal efficiency is 48% for a 30 m buffer and 90% for a 100 m buffer. The water flux is influenced by soil texture, organic content, and depth. For sites with low water flux, narrower buffers may be sufficient, but these areas do not contribute substantially to streamflow. Effective watershed removal of nitrogen requires at least 30 m buffers, with removal efficiencies continuing to increase for wider buffers (Sweeney and Newbold, 2014). This contrasts with the streamside protection enhancement area (SPEA) policy in the City of North Vancouver, which only applies to 15 m from top of bank (City of North Vancouver, 2015). This buffer may also be reduced under the discretion of Qualified Environmental Professionals (QEPs) under pressure from developers and where pre-existing infrastructure has been built. Therefore, riparian buffer quality at the site scale, as well as percent of impervious surfaces at the watershed scale, combine to determine water quality and the resulting stream biological community.

This research aims to expand our knowledge on the effects of stormwater runoff on benthic invertebrate communities and provide a current baseline for stream condition in Mosquito Creek, North Vancouver, which supports several salmonids including coho (*O. kisutch*), steelhead (*O. mykiss*), and cutthroat trout (*O. clarkii*) (DFO, 2017; Cassidy, 2019). In this work I investigated the percent of impervious surfaces, riparian buffer condition, and the resulting water quality and benthic invertebrates in Mosquito Creek and nearby Mackay Creek as a comparison. I specifically compared upstream and downstream sites, in relation to the Highway 1 bridge and related stormwater outflow, which enters Mosquito Creek.

In addition to baseline studies, I have also provided recommendations for restoration and mitigation strategies needed to improve water quality and address urban runoff in Mosquito creek. Mitigation techniques can be used to improve water quality by controlling the water at its source using bioretention facilities such as rain gardens and infiltration galleries (Metro Vancouver, 2012). A rain garden is a concave landscape area planted with shrubs, sedges, and plants, where runoff is retained temporarily to allow infiltration into soils below. An infiltration trench uses a perforated distribution pipe installed underground and surrounded by drain rock to allow runoff to soak into the ground more gradually (Metro Vancouver, 2012). Other low impact development (LID) solutions include: bioswales, a shallow channel planted with grass used to convey stormwater; pervious paving such as porous asphalt or modular pavers, which allow water to percolate through; and planting trees to increase interception and absorption of rainfall (Metro Vancouver, 2012).

Though the specific constructions differ, the main purpose of these mitigation strategies is to slow the flow of water, increase infiltration and water storage, and trap some pollutants. This benefits urban streams by restoring a more natural hydraulic regime, increasing shallow and deep groundwater infiltration, filtering out and trapping pollutants, and reducing turbidity and erosion issues. This leads to improved water quality, natural invertebrate communities, and provides improved habitat conditions for salmonids in urban streams. The City of North Vancouver stormwater management plan has the goal that 30% of all road drains (catch basins) will have a source control (i.e. raingardens) and that 50% of all outfalls will have a treatment structure (e.g. stormwater oil and grit separators, treatment wetlands, or treatment ponds) by 2031 (City of North Vancouver, 2016). This research provides ongoing evidence and support for the continuation of these strategic plans and goals.

Chapter 2. Objectives

The main objectives of this research are to (1) improve our understanding of how urban runoff influences salmonid habitat (through benthic invertebrates), (2) record baseline conditions, and (3) determine what restoration practices would be needed to improve water quality in Mosquito Creek, North Vancouver. My three key research questions are: (1) how is water quality affected by urban runoff in streams of a developed area, such as North Vancouver, BC; (2) how does urban runoff influence the biological condition (benthic invertebrate community) of Mosquito Creek; and (3) what is the proportion of impervious surfaces in the watershed and at the site level, and how do these factors relate to the biological condition.

My hypothesis is that water quality in this developed urban watershed will have higher turbidity, lower dissolved oxygen, and more variable temperatures than a reference stream. I predict that the benthic community will have shifted to more pollution tolerant species with a loss of pollution sensitive taxa. I predict a more stressed benthic community with lower abundance, reduced family richness and other diversity metrics (Simpson's diversity and Simpson's evenness), lower diversity of EPT species (low EPT index, EPT abundance, and EPT total ratio), an increase in % Chironomidae and % Dominance and an overall poor stream health rating. Similarly, I expect that greater proportions of impervious surfaces in the watershed and reduced area of forested riparian buffers at the site will correlate with changes to water quality and benthic invertebrates.

The main goal of this project is to use these results to make recommendations for restoration of Mosquito Creek. One area of particular interest is where runoff is conveyed from the Highway 1 bridge into Mosquito Creek. Due to the high levels of traffic and runoff, this may represent an effective candidate site for restoration. Restoration plans will also need to take into account existing integrated stormwater management plans in North Vancouver, stormwater guidance in BC, and ongoing projects in the watershed (City of North Vancouver, 2016; BC MWLAP, 2002; Cassidy, 2019). It is also imperative that restoration projects will be effective in future climate scenarios. Restoring riparian areas and a more natural flow regime is highly valuable in a changing climate, where rainfall and peak flows are predicted to increase, and shading is needed to maintain cool water temperatures (Beechie et al., 2012).

This work will improve our understanding of the relationships between impervious surfaces, loss of riparian buffers, and the resulting impacts to biotic condition of urban streams from stormwater. In particular, studies of areas of moderate development are needed to improve our understanding of critical thresholds involved. This information is essential to improving the ecological outcomes of the numerous stream restoration projects in the region. For example, restoration on lower Mosquito Creek including installation of large woody debris (LWD) and boulders, was completed in 2020. By incorporating a watershed perspective and improving water quality, we can increase the success of these and other restoration efforts. This study and recommendations may also help to inform regional planning and future restoration projects in other urban areas of the Lower Mainland, BC, and throughout the Pacific Northwest.

Chapter 3. Methods

3.1. Study site

The main study site of interest was Mosquito Creek in North Vancouver, BC (Figure 1). Although completely natural sites in North Vancouver are limited, Mackay Creek, which is comparatively less altered, was selected as a nearby comparison site to serve as a control. The headwaters of both creeks lie close to Grouse Mountain in the District of North Vancouver and then flow south into City of North Vancouver before draining into Burrard Inlet. The proximity of the two creeks (which are less than 1 km apart), similarity in aspect, and similar creek size support the comparison between the two streams.

Results were also compared between an upstream and downstream sampling site on both creeks. On Mosquito Creek, the upstream sampling point (MOSUP) was located south of West Queens road, and the downstream point (MOSLW) was located south of the Trans-Canada highway bridge in William Griffin Park. On Mackay Creek, the upstream point (MACUP) was located north of the Trans-Canada highway, in Murdo Fraser Park, and the downstream point was located in Heywood Park (MACLW). Specific sampling sites were selected based on locating ideal riffle habitat for invertebrate sampling, and to maintain approximately equal latitude, and equal distance between the points on the given stream. Results were analyzed using a standard BACI design where Mackay and Mosquito Creeks represent control and impact conditions respectively and upstream and downstream position represents before and after.

Mosquito Creek also has two main tributaries, Mission Creek in the north, and Thain Creek in the south (with its tributary Wagg Creek). Additional sampling points along Mission Creek were used in order to provide more information about the contributions of runoff pollution from this highly developed tributary. The Mosquito Creek upstream site (MISUP) was located near Shannon Crescent and Evergreen Place, and the downstream site (MISLW) was located just before the confluence with Mosquito Creek in William Griffin Park. Study of Mission Creek was included to help explain the influence of this tributary on the downstream Mosquito site, as Mission Creek joins between the upstream and downstream Mosquito Creek sampling sites.

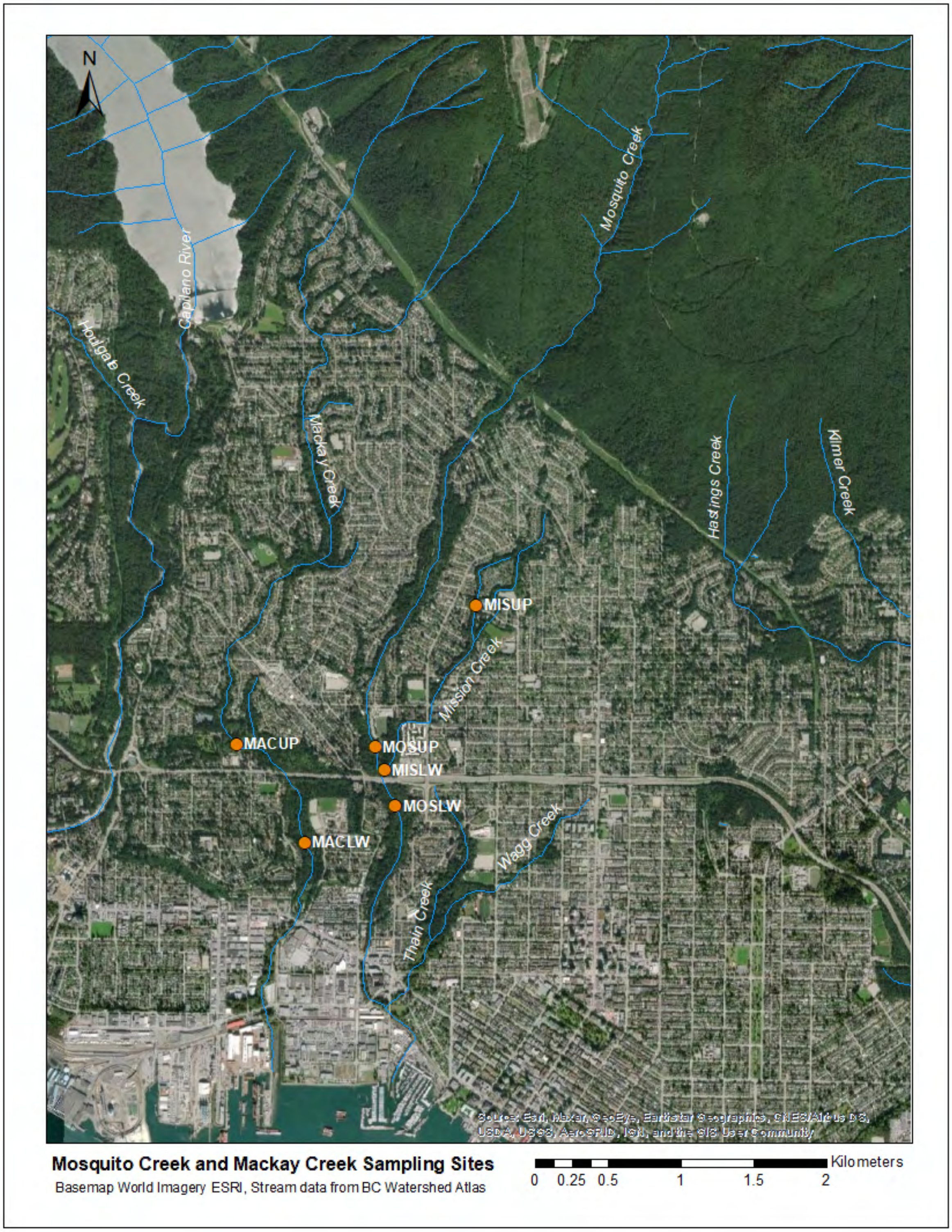

Figure 1. Map of Mosquito Creek watershed and Mackay Creek watershed and sampling points

3.2. Site and Habitat Measurements

General site information was collected for each site including GPS coordinates, elevation, stream order, ecoregion, and surrounding land-use. Stream order was determined following the Strahler method as outlined in the CABIN method (RISC, 2009). Photographs were taken on each site visit at each site as a reference. This included across the creek, upstream, downstream, canopy, representative photo of the substrate, and underwater using a Fujifilm FinePix XP140 camera.

Hydrology measurements were taken to document the flow regime and determine discharge of the creek twice per month from May to October 2020, and on the invertebrate sampling dates. This included measurements of bankfull width, wetted width, and average depth. Bankfull width was defined as the estimated width at high flow conditions from a 1-2 year flood, determined by locating points of change on the stream banks, where areas of scour are evident, sediment texture abruptly changes, or vegetation changes indicate a shift from water tolerant species to upland conditions (RISC, 2009). Slope was measured using a Suunto Clinometer by taking the average of three measurements over the reach. Reach length was defined following the CABIN protocol of six times the bankfull width (RISC, 2009). Water velocity was recorded using a Hach flow meter at a minimum of ten points along a cross-section. Discharge was then calculated by multiplying the cross-sectional area of each segment by the average of three velocity measurements.

In order to gain a better understanding of stream habitats available to invertebrates, I took stream and substrate measurements following the CABIN protocol (RISC, 2009). This included habitat types, macrophytes, periphyton coverage, dominant substrate, substrate embeddedness, and surrounding material. A Wolman substrate survey, measuring the intermediate axis of 100 substrate particles, was also completed to determine Wolman D50 and Wolman Dg values. I recorded presence of LWD greater than 10 cm diameter and longer than 2 m that was functional within the channel according to FHAP procedures (Johnson and Slaney, 1996). Functional LWD are those that lie at least partly within the bankfull channel that may influence channel geomorphology by causing scour or impoundment (Johnson and Slaney, 1996). In the riparian area, I recorded key habitat variables such as dominant riparian vegetation, percent canopy cover, and estimated percent cover of each vegetation class (conifer, deciduous, shrub, fern/grass) according to the CABIN method (RISC, 2009). An area-constrained survey for vegetation types was also completed for each site in July, when vegetation could most easily be identified, to describe vegetation species present in each reach. I also recorded incidental sightings of invertebrates and other wildlife to provide additional qualitative descriptions of each site. I assessed each site for potential impacts such as erosion, undercut banks, or pollution. Noting the exceptionally high recreational use of the creeks and associated trails by hikers and dogs, I also completed a recreational use survey to quantify this potential impact at each site.

3.3. Water Quality Measurements

Water quality measurements were completed using a YSI Professional Plus multi-meter to record temperature, dissolved oxygen, conductivity, and pH of the stream. Dissolved oxygen was calibrated at each site to account for changes in atmospheric pressure, pH was calibrated with a 3-point calibration and conductivity was calibrated with a single point. Turbidity measurements were also taken using a LaMotte2020we turbidity meter. Turbidity measurements were taken in triplicate and the average was used to increase precision of the estimate. Turbidity was calibrated at each site to a zero turbidity standard solution. Water quality measurements were taken at each site visit, twice per month from May to October 2020, in order to get an average of the conditions during the summer and leading up to the first major rainfall event. This included measurements at the same time as spring invertebrate sampling. Water quality was also tested as soon as possible after the first major rainfall event in late September, concurrent with fall invertebrate sample collection. All flow and water quality measurements were completed after invertebrate sampling, to avoid disturbing the substrate. Concurrent measurements of air temperature and weather was also described as these may influence stream conditions.

3.4. Benthic Invertebrate Collection

Invertebrates were sampled at one upstream site and one downstream site for Mosquito Creek and Mackay Creek, for a total of four sampling points. Additional sampling at both sites on the Mission Creek tributary were also completed but were not analysed due to time constraints. Invertebrate sampling is typically done in the fall or late summer when the largest portion of the taxa in the stream are likely to be present in an aquatic life stage and flow conditions are optimal for safe sampling (Branton et al., 2006). In order to determine the effects of urban runoff on the invertebrate community, samples were collected following the first large storm in the fall, which occurred in late September. This is when contaminants will potentially be the highest in runoff and there is the greatest chance of detecting changes in the benthic invertebrate community.

Rainfall events have two important aspects to consider, the total amount of precipitation per storm event, and the intensity or rate of rainfall. In terms of rate, moderate rainfall is defined as a precipitation rate between 2- 10 mm per hour and heavy rain is defined as a precipitation rate 10- 50 mm per hour (Met Office, 2012). The first large storm event meeting this condition of moderate or heavy intensity rainfall was selected as this level is expected to mobilize potential urban pollutants and lead to flowing runoff conditions beyond water pooling on paved areas. Prior to the first storm it of fall, it was generally sunny. Though smaller overnight precipitation events did occur, they were likely not large enough to mobilize a significant amount of the contaminants. Therefore, precipitation events on the order of 1- 20 mm total were considered minor events. We only considered large events over approximately 20 mm as this volume of precipitation would be likely to lead to excessive runoff and transport of urban contaminants into the stream.

Fall Benthic invertebrate samples were collected at each sampling site on September 26, 27, and 29, 2020 (Table 1). Samples were collected following the first large storm event marked by a large volume (>50 mm in one 24-hour period) and moderate-heavy rainfall intensity (5-10 mm/hr), which occurred 23 September to 25 September 2020 (ECCC, 2020). There was some additional rainfall on September 26; however, the vast majority of the storm occurred in the three days prior to sampling. Leading up to the storm, it had been relatively dry since the previous large storm ending August 21 (total of 32 days since last major rainfall event) (ECCC, 2020).

The DFO (2000) stream invertebrate sampling guide also recommends sampling in early spring and early fall (before heavy rainfall) as organisms may be washed away. Therefore, I also sampled benthic invertebrates in spring, in May, in order to better understand the complete invertebrate community using the stream over the course of the year. Runoff pollution may cause acute effects as well as chronic effects in the invertebrate community with many events over time. Measuring after the first storm in fall, would represent acute effects, as pollutants have had a chance to build up to higher concentrations over the preceding dry period in late summer. Consistent trends in both benthic invertebrate samples, and in particular differences in the spring sample, would provide a more integrated picture of water quality conditions over the long-term and chronic effects of runoff pollution.

Table 1. Weather pattern of the first storm in September and sampling dates.

Date, Sept 2020	23	24	25	26	27	28	29
Weather	rain	rain	rain	rain	sun/cloud	sunny	sunny
Rainfall (mm)*	59.9	24.3	30.9	1.9	0	0	0
Forecast rainfall rate (mm/Hr)	max 5-10	max 1-3	max 1-3	max 1	none	none	none
Sampling dates				X	X		X

*rainfall data from nearest Environment Canada rainfall station in West Vancouver (ECCC, 2020)

Benthic Invertebrate sampling followed a standard collection method using a Surber sampler (DFO, 2000; MoE, 1999). Samples were preferentially collected in shallow riffle areas with moderately fast flow and cobble substrate (rocks 5 to 25 cm in diameter). Riffle habitats are ideal as they support the most diverse array of species. To minimize environmental differences between sites and between streams, riffles with coarse gravel-cobble substrates were targeted over sand-gravel or boulder substrates (Page, 2008). Five, 30 cm by 30 cm sub-samples were collected at each sampling station. Specific placements were randomly selected and were spread out across the reach to capture a variety of conditions (Figure 2).

The Surber sampler was placed on the downstream edge of the sampling area, so the opening faces upstream into the flow and the frame is pushed lightly into the stream substrate (DFO, 2000). Each stone and debris 5 cm or larger in the sampling area was held underwater and brushed off to loosen invertebrates which were swept into the net. Following inspection of each large stone, rocks were discarded downstream or to the side of the sampler (MoE, 1999). The streambed was then gently agitated to a depth of 2 to 5 cm to loosen any remaining invertebrates. In order to maintain comparability between stations, time spent on handling and rubbing the substrate was limited to 5 minutes per sub-sample (MoE, 1999). In order to minimize potential bias, all work was completed by the same person, effort was made to spend at least 3 minutes agitating the gravel, and the full 30 cm square area was thoroughly covered in all samples.

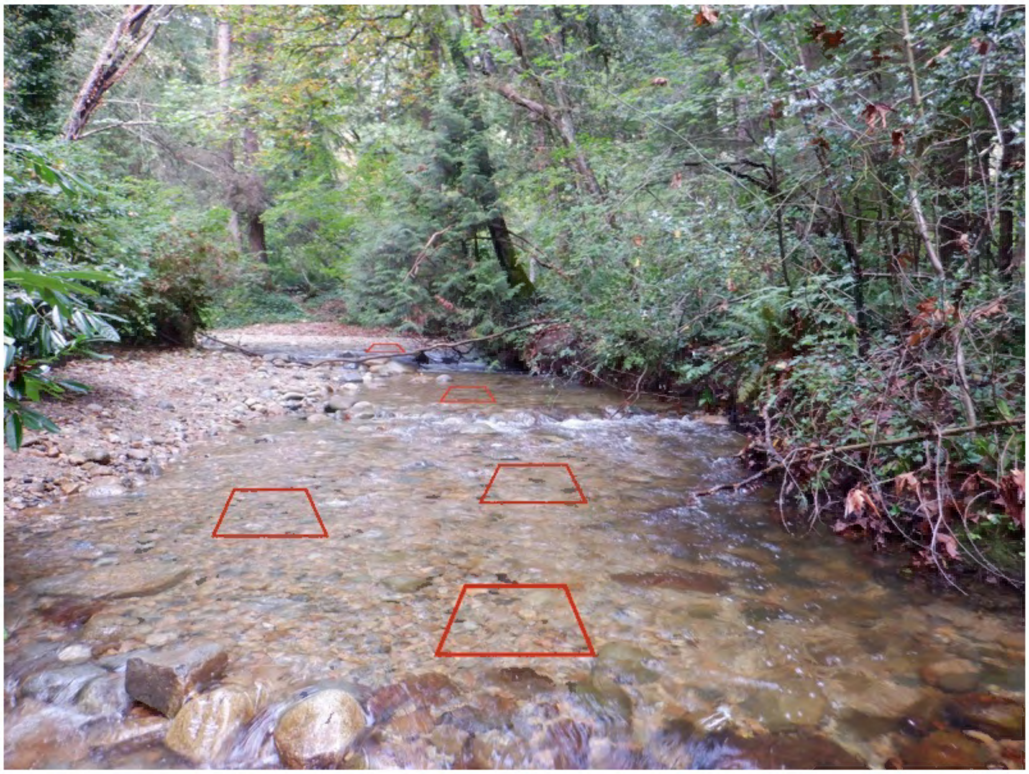

Figure 2. Example Surber Sampler placements for benthic invertebrate sampling at the Mackay Creek upper site.

The net was then removed with a forward scooping motion and the stream water was used to rinse down any invertebrates into the collection bottle at the end of the net. After rinsing, nets were checked for any remaining invertebrates clinging to the net, which were transferred into the sample jar immediately. Invertebrates were then preserved for future analysis in a glass jar (250 mL). In order to preserve the sample, the concentrated 99% isopropyl alcohol was added to make a 70% solution of alcohol in water. Sample vials were then labeled with: unique station ID, replicate number (1-5), date of collection, and initials of collector. Where additional jars were needed, in the case of exceptionally high debris, jars were also labelled A and B. Samples were immediately placed into a cooler bag for transport.

3.5. Benthic Invertebrate Identification

Preserved benthic invertebrate samples were stored in a cool, dark environment to limit potential for degradation. Samples were then transferred into a large clear tray for visual inspection and viewed under a microscope at 10-20X magnification in small batches to ensure all benthic invertebrates were included. Invertebrates were sorted, identified, and enumerated. Taxonomic Identifications were made to family or the lowest practical level. Key reference texts for aquatic insect identification were consulted, in particular, Merritt et al. (2008) Guide to Aquatic Insects of North America, Voshell (2002) A Guide to common Freshwater Invertebrates of North America, as well as electronic reference materials such as Haney et al. (2013) An Imaged-Based Key to Stream Insects, and Bouchard (2004) Guide to aquatic macroinvertebrates of the Upper Midwest. The results were then be recorded and the total abundance, EPT richness, and other metrics were calculated. Two of the five sub-samples were randomly selected for analysis to account for within-site variability, as invertebrates are not distributed evenly in a stream, and represent replicates for each site.

One of the key benthic invertebrate metrics used was the EPT Index. As mentioned previously, this measures taxa from the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) which are pollution sensitive. High EPT richness, therefore, indicates a diverse benthic community with pollution sensitive taxa, and a healthy stream environment, while low EPT richness suggests environmental degradation. We also used several other metrics from the Streamkeepers methodology including: Density (estimated abundance /m2), Pollution tolerance index (PTI), EPT abundance, EPT to Total Ratio, Predominant taxon Ratio, Streamkeepers Site Assessment Rating, and an overall assessment of Stream Health. We also described the invertebrate samples using several key metrics including abundance (the total number of benthic invertebrates), Total Family Richness (the number of families), % Chironomidae and % Dominant 3 taxa. We used diversity metrics from the CABIN methodology to further describe the communities present including Simpson's Diversity Index, Simpsons Reciprocal Index, and Simpson's Evenness, which measures equitability or how evenly taxa are distributed in the community.

Terrestrial invertebrates are also often collected in benthic invertebrate samples as they will drop into streams during flight, fall from overhanging vegetation, and are carried in stream currents until they lodge in the substrate (McLemore and Meehan, 1988). All terrestrial invertebrates were identified to order or family, and subsequently excluded from further analysis of benthic invertebrates as recommended in the CABIN protocol (RISC, 2009). Zooplankton species such as copepoda and cladocerans (water fleas) were also routinely observed. Presence and absence of these was noted, and they were also excluded from further benthic invertebrate analysis (RISC, 2009). Statistical analyses were completed using JMP (version 16) statistical software.

3.6. Mapping Analysis

All mapping was completed using ArcGIS software (ArcMap, ArcGIS Version 10.8, Esri 2019). Arial imagery from 2014, which was categorized into landcover types using maximum likelihood analysis, was provided by City of North Vancouver for the regional area covering Mosquito and Mackay creek watersheds. Using ArcGIS, this data was analyzed to produce a map of impervious surfaces for the entire study area covering both Mackay creek and Mosquito Creek watersheds. Mapping data showing extent of impervious surfaces from the District of North Vancouver within the urban boundary of the district for 2016 was also used as a point of comparison. Percent cover of impervious and pervious areas were then calculated for each watershed.

Site buffers were also mapped extending 30 m out from around each sampling point in a circle (A = 2827.4 m^2; r = 30 m). As described by Sweeney and Newbold (2014), riparian widths up to approximately 580 m from the stream can have ecological benefits, but effects are diminishing beyond approximately 30 m. Percent impervious area, where developments removed portions of the riparian buffer, were then calculated for the immediate site area, within the circle around each sampling point to estimate the site level impacts. The District of North Vancouver 2016 dataset was also used, where available, to improve accuracy of these site level estimates.

Chapter 4. Results

4.1. Site Descriptions

Various site parameters were determined for each of the six sites, Mackay Creek upper and lower, Mosquito Creek upper and lower, and Mission Creek upper and lower, as described in Table 2. All of the sampling sites are located within the Pacific Maritime Ecozone, Lower Mainland Ecoregion. More specifically, all sites were located within the Coastal Western Hemlock dry maritime (CWHdm) Biogeoclimatic Ecosystem Classification (BEC) zone with the upper reaches of the watersheds transitioning to very wet maritime, montane (CWHvm-2); MFLNORD 2021; BC MoF 1991). Riparian vegetation characteristics estimated for each site are summarized in Table 3 and a full list of vegetation species observed is provided in Appendix A. Detailed substrate characteristics are provided in Table 4. A representative upstream photo for each site is shown in Figure 3 and a complete set of photos for each site is provided in Appendix B.

The Mackay Creek upper site was dominated by coniferous vegetation with species such as western redcedar (*Thuja plicata*), big leaf maple (*Acer macrophyllum*), and salmonberry (*Rubus spectabilis*). The stream substrate at the Mackay upper site was dominated by cobbles with some coarse gravel and the LWD was poor. The Mackay upper site was impacted by undercut banks and erosion, invasive species including English ivy (*Hedera helix*) smothering trees, small sprouts of Japanese knotweed (*Reynoutria japonica*), as well as several escaped garden plants such as common oak (*Quercus robur*) and cherry Laurel (*Prunus laurocerasus*). Recreational use of the site was limited with an average of 2 hikers and 1 dog per hour during the summer (see Appendix C for recreational use survey results). Surrounding land-use of the upper Mackay Creek site was primarily forested park with some residential, and a small industrial site about 120 m upstream (Figure 4). Abundant local wildlife was observed at the upper Mackay site including a mallard pair (*Anas platyrhynchos*), a river otter (*Lontra canadensis*), and an American dipper (*Cinclus mexicanus*) feeding in the stream. Salmonid fry were also observed, particularly under the cut bank.

Table 2. Habitat and sample reach parameters at the sampling sites.

Measure	MACUP	MACLW	MOSUP	MOSLW	MISUP	MISLW
Location Coordinates	0492288 5464618 10U	0492752 5463944 10U	0493239 5464598 10U	0493378 5464192 10U	0493933 5465569 10U	0493304 5464436 10U
Elevation (m)	61 m	33 m	70 m	63 m	143 m	84 m
Stream Order	3	3	2	3	1	2
Average Wetted Width (m)	4.98 m	5.97 m	8.27 m	11.64 m	2.65 m	5.41 m
Bankfull Width (m)	6.13 m	10.08 m	9.53 m	18.90 m	6.15 m	8.65 m
Reach length (m)	36.8 m	60.5 m	57.2 m	113 m	36.9 m	51.9 m
Stream habitat type	straight run, riffle, pool	straight run, riffle	step-pool	riffle, straight run, pool	step-pool, riffle	step-pool, riffle
Dominant substrate	cobbles (some coarse gravel)	cobbles (some coarse gravel)	cobbles (some boulders)	cobbles (some boulders)	cobbles	cobbles (some boulders)
Slope (%)	4%	5%	7.3%	3.7%	7.7%	6.7%
Macrophytes (%)	0%	0%	0%	0%	0%	0%
Periphyton coverage	2	3	3	3	3	2
LWD per bankfull channel width	0.17	1.83	0	0.67	0.17	0.33

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, MIS- Mission Creek, UP- Upper, LW- Lower

Table 3. Riparian vegetation characteristics at the sampling sites.

Measure	MACUP	MACLW	MOSUP	MOSLW	MISUP	MISLW
Dominant vegetation	coniferous	coniferous	deciduous	deciduous	coniferous	deciduous
Canopy cover %	73%	51%	88%	55%	71%	87%
% Conifer	40%	50%	10%	10%	40%	35%
% Deciduous	35%	25%	65%	40%	30%	45%
% Fern/grass	10%	5%	20%	10%	20%	5%
% Shrub	15%	20%	5%	40%	10%	15%

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, MIS- Mission Creek, UP- Upper, LW- Lower

Table 4. Substrate characteristics at the sampling sites.

Measure	MACUP	MACLW	MOSUP	MOSLW	MISUP	MISLW
Dominant substrate	cobbles (some coarse gravel)	cobbles	cobbles	cobbles (some boulders)	cobbles	cobbles (some boulders)
Substrate embeddedness	4	4	2	4	4	3-4
Surrounding material	2 (sand)	2 (sand)	2-3 (sand-gravel)	3 (gravel)	3 (gravel)	3 (gravel)
Wolman D50 (mm)*	55	55	79	79	53	63
Wolman Dg (mm)*	51	52	71	73	65	55
Mean particle size (mm)	63	65	94	88	58	72

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, MIS- Mission Creek, UP- Upper, LW- Lower
*Wolman D50 describes the median diameter; Wolman Dg describes the geometric mean diameter

The Mackay Creek lower site was also dominated by coniferous vegetation with species such as western redcedar (*T. plicata*), as well as vine maple (*Acer circinatum*) and false lily of the valley (*Maianthemum dilatatum*). The dominant stream substrate at the Mackay upper site was cobbles and the LWD was fair, with several functional logs and a log jam at the top of the reach. The Mackay lower site had relatively few impacts but did have several invasive species such as policeman's helmet (*Impatiens glandulifera*) and yellow archangel (*Lamiastrum galeobdolon*) and had a point-source pollution event causing high turbidity (see Section 4.4). The site was also connected to a trail network with moderate recreational use with an average of 16 hikers per hour and 4 dogs per hour during the summer. Surrounding land-use of the lower Mackay Creek site was primarily forested park surrounded by residential areas. Incidental wildlife observed at the lower Mackay site included Anna's hummingbird (*Calypte anna*), an American robin (*Turdus migratorius*) eating salmonberries, and Swainson's thrush (*Catharus ustulatus*).

The Mosquito Creek upper site was dominated by mature deciduous vegetation such as black cottonwood (*Populus trichocarpa*) and red alder (*Alnus rubra*), while the few conifers such as red cedar (*T. plicata*) were much younger, as the area had previously been logged. The site was just downstream of the large 1.2km Evergreen culvert above Queens road and had a variable hydraulic environment. This contributed to the dominant stream substrate of cobbles with some boulders, and the lack of LWD in the reach.

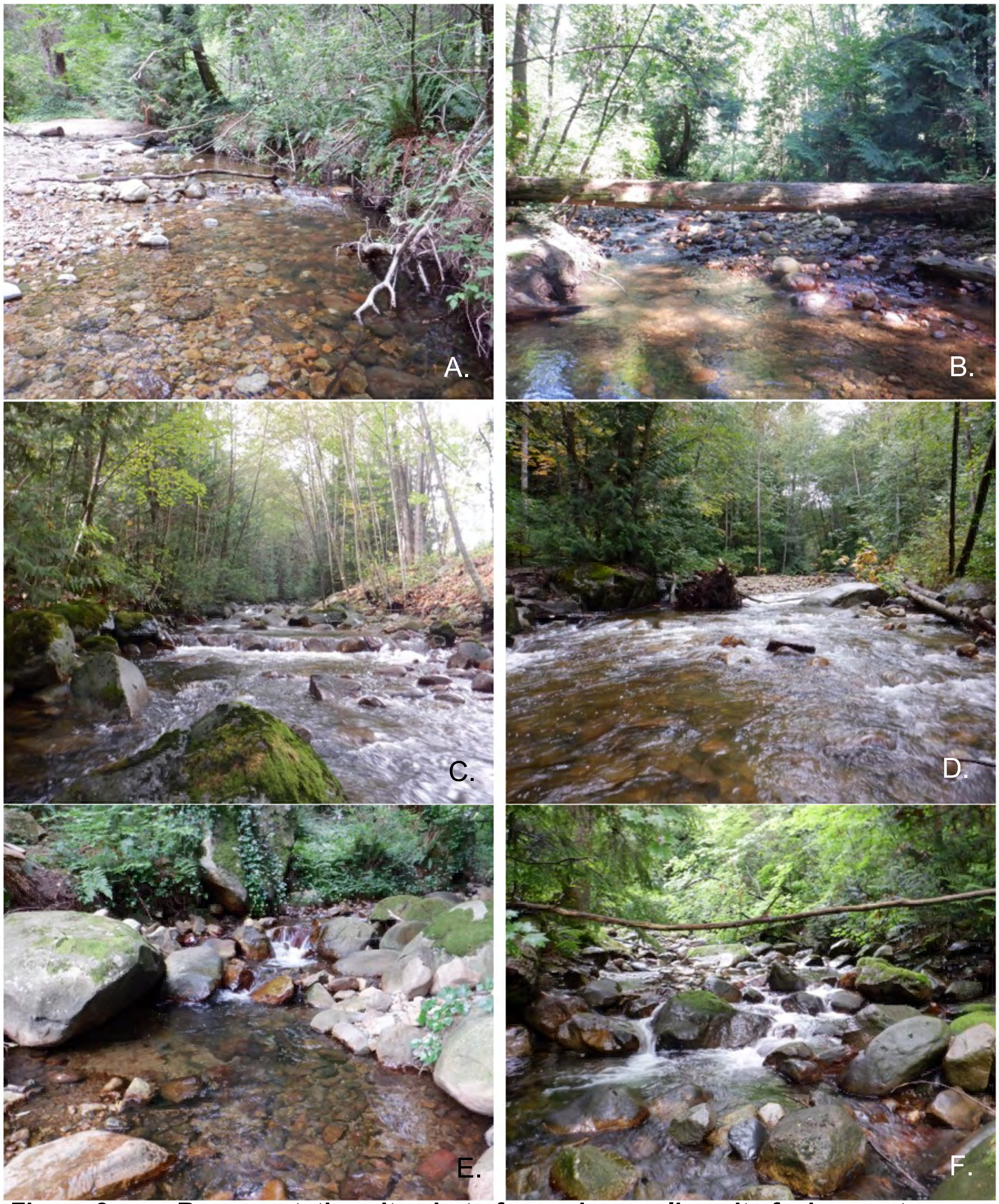

Figure 3. **Representative site photo for each sampling site facing upstream. A. Mackay Creek upper site 21 July 2020. B. Mackay Creek lower site 4 August 2020. C. Mosquito Creek upper site 26 September 2020. D. Mosquito Creek lower site 26 September 2020. E. Mission Creek upper site 7 July 2020. F. Mission creek lower site 9 June 2020.**

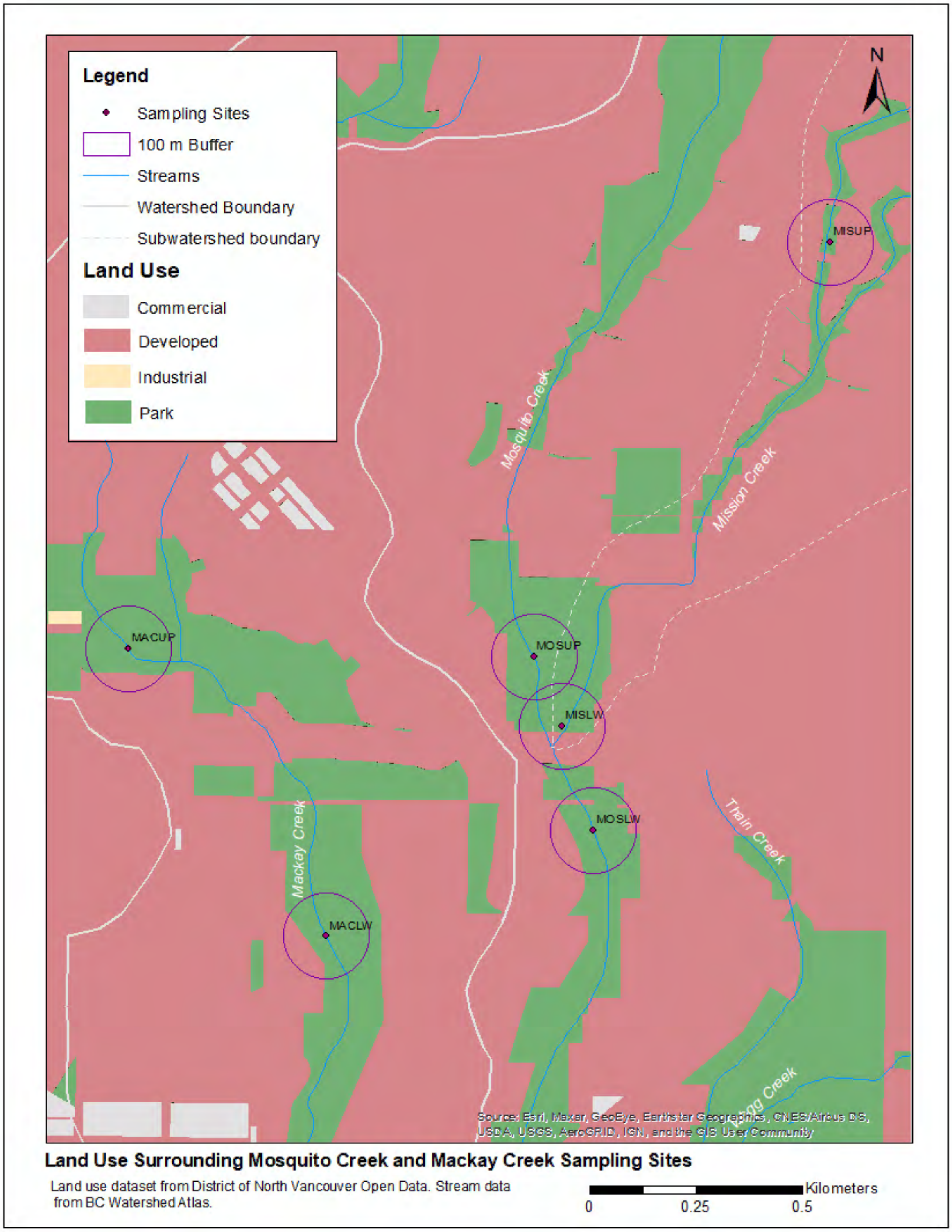

Figure 4. **Map of land use in Mosquito Creek and Mackay Creek watersheds surrounding sampling sites.**

The Mosquito upper site had a number of impacts including built areas for the culvert and the hiking bridge, and the stream was channelized and straightened. There were also erosion issues, and a slope stabilization structure was eroding causing pieces of the metal mesh to break apart into the stream. There were also several invasive species such as Himalayan blackberry (*Rubus armeniacus*) and removal work had occurred leading to some areas of bare soil. Trails near the site also had moderate recreational use with an average of 33 hikers and 14 dogs per hour during the summer (Appendix C). Surrounding land-use of the site was primarily forested park surrounded by residential areas. Incidental fish and wildlife observations included a northern flicker (*Colaptes auratus*) nest with the parent feeding its young, as well as several small coho (*O. kisutch*) fry (2 inches long), and a large fry (5 inches long) under the bridge.

The Mosquito Creek lower site was dominated by deciduous vegetation with species such as red alder (*A. rubra*) and salmonberry (*R. spectabilis*), with a few younger conifers such as western hemlock (*Tsuga heterophylla*). The dominant stream substrate at the Mosquito lower site was cobbles with some boulders. The LWD was poor in terms of functional wood in the stream though additional logs near the stream were abundant. The site was impacted by variable and high flows following rain. In one instance, in October 2020, the flows were so great, they lifted the large metal lid off of the intake box for the restored side channel and carried it up onto the opposite bank. LWD was also rafted downstream on another occasion. The lower Mosquito site had a variety of non-native species, though most were only a single plant. Trails at the site also had high recreational use with an average of 64 hikers and 23 dogs per hour during the summer (Appendix C). Surrounding land-use of the site was primarily forested park surrounded by residential areas. Incidental fish and wildlife observed at the lower Mosquito site included salmon fry swimming in the shallows on several occasions, and a Great Blue Heron (*Ardea herodias*) flying downstream and later upstream likely looking for spawning salmon in October.

The Mission Creek upper site was dominated by coniferous vegetation with species such as western hemlock (*T. heterophylla*), vine maple (*A. circinatum*), and sword fern (*Polystichum munitum*). The dominant stream substrate at the Mission upper site was cobbles and the LWD was poor. The site was located just upstream of a newly restored culvert with riprap and plantings with coconut matting on the steep banks. The site was impacted by the loss of large trees towards the bridge, the close proximity of houses in the upslope riparian area, and an abundance of invasive species such as sprawling English ivy (*Hedera helix*) and common periwinkle (*Vinca minor*). Recreational use of the site was extremely limited (an average of 0 hikers and 0 dogs per hour) but on one occasion a local family used the area for picking salmonberries (recreation survey Appendix C). Surrounding land-use of the site was residential with a narrow riparian strip as it is a small tributary stream. Incidental wildlife observed at the upper Mission Creek site included pacific wren (*Troglodytes pacificus*), black-capped chickadee (*Poecile atricapillus*), and eastern grey squirrels (*Sciurus carolinensis*).

The Mission Creek lower site was dominated by deciduous vegetation with species such as red alder (*A. rubra*), western redcedar (*T. plicata*), and salmonberry (*R. spectabilis*). The dominant stream substrate at the Mission lower site was cobbles with some boulders, and the LWD was poor. The Mission creek lower site was impacted by a small footbridge, a point-source pollution event (see Section 4.4), pollution from a camp at the upstream end with beer cans, clothing, and other garbage, as well as invasive species such as English ivy (*H. helix*) and a few spots of Japanese knotweed (*R. japonica*). Recreational use of the site was low with an average of 7 hikers and 4 dogs per hour (Appendix C), as the site was situated along a side trail. Surrounding land-use of the lower Mission Creek site was forested park and residential. Incidental fish and wildlife observed at the site included pacific wren (*T. pacificus*), salmonid fry swimming in pool areas, as well as numerous moths following an outbreak of western hemlock looper (*Lambdina fiscellaria*) in August 2020.

4.2. Stream Discharge

Flow measurements were completed, and average discharge was calculated at each of the six stream sites: Mackay Creek upper and lower, Mosquito Creek upper and lower, and Mission Creek upper and lower, from May to October (Figure 5, Table 5). A pattern of decreasing flow was observed throughout the summer months, and then a strong increase in flow in late September, which coincided with the first major storm of the fall. Increased flow then continues into October. The increase in flow for both sampling dates in June corresponds with rain events on the day of sampling or the days prior.

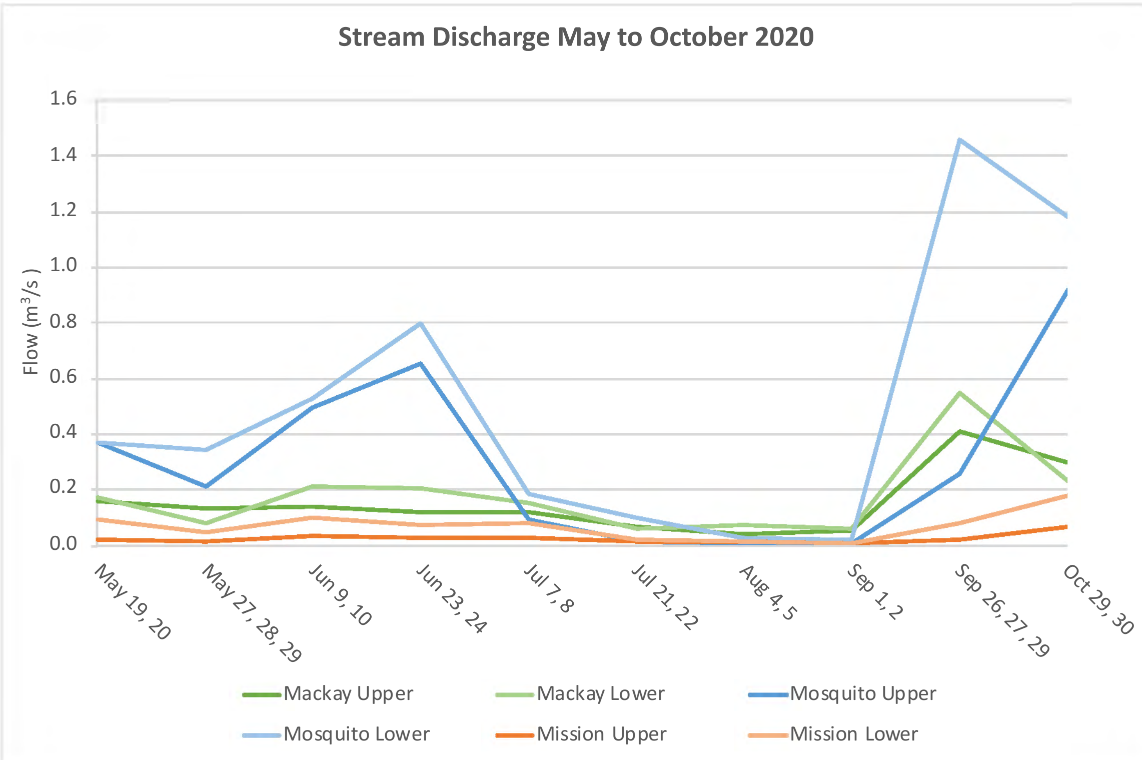

Figure 5. Average stream discharge from May to October in cubic meters per second at each stream site.

Note: Actual dates may differ slightly as flow measurements were completed over 2-3 days.

Table 5. Flow statistics and average discharge across each stream site for the period of May to October, 2020.

Measure	MACUP	MACLW	MOSUP	MOSLW	MISUP	MISLW
Average discharge (m^3/s)	0.153	0.179	0.303	0.501	0.023	0.071
SE	0.04	0.05	0.10	0.16	0.01	0.02
Maximum discharge (m^3/s)	0.409	0.551	0.918	1.460	0.065	0.178
Minimum discharge (m^3/s)	0.042	0.059	0.008	0.021	0.004	0.009
Range (m^3/s)	0.367	0.492	0.910	1.439	0.061	0.169

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, MIS- Mission Creek, UP- Upper, LW- Lower
Range is the difference between the maximum and minimum discharge.

Average discharge was greatest for the Mosquito Creek lower site (0.501 m^3/s) and the greatest range of flow was also found at the Mosquito Creek lower site. Mackay Creek upper and lower sites were more similar and the flow was less variable. As expected, Mission Creek discharge was considerably less than the others, as it is a tributary of

Mosquito Creek. On average discharge on Mission Creek increases about three-fold from the upper site to the lower site. The lowest flows were recorded in early September and ranged from just 0.004 to 0.059 m^3/s at the upper Mission Creek site and lower Mackay Creek site respectively. At this date, even the Mosquito Creek discharge was less than at Mackay Creek. The greatest discharge was recorded in October or late September at all sites.

4.3. Water Quality

Water quality measurements including water temperature, dissolved oxygen, pH, conductivity, and temperature were measured at each of the six sites over the summer from May to October, 2020. Water temperature shifted gradually over the seasons, from cooler temperatures in the early spring, in May to warmer temperatures in summer in August (Figure 6).

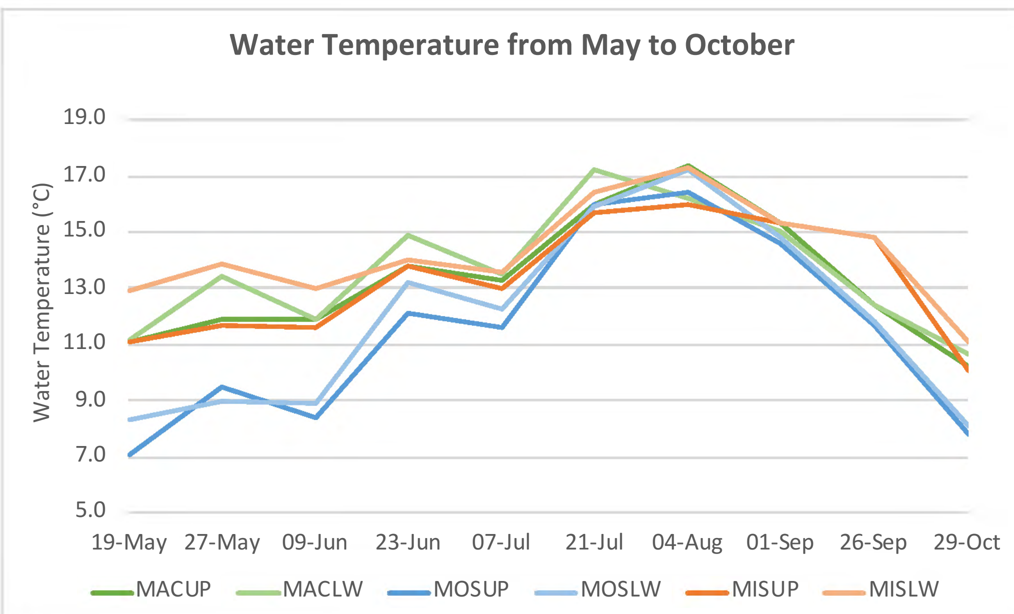

Figure 6. Water temperature from May to October 2020 at each stream site.
Note: Actual dates may differ slightly as measurements were completed over two days.

Mosquito Creek was generally cooler on average, but this difference disappeared by the late summer. Snow meltwater likely contributed to the significantly cooler water temperatures observed Mosquito Creek until about the first half of June. Snow on Grouse Mountain was completely gone by 23 June 2020 observations, when stream temperatures increase. Stream temperatures across all sites began to decrease again in

September and continuing in October. Overall, water temperatures remained below 18°C across all sites on the dates measured.

We also measured the concurrent air temperature at each site, which ranged from 10 to 23°C on the dates measured from May to October 2020. There appeared to be no substantial differences between the average air temperatures at each site. We also plotted air temperatures relative to the water temperatures measured (Figure 7).

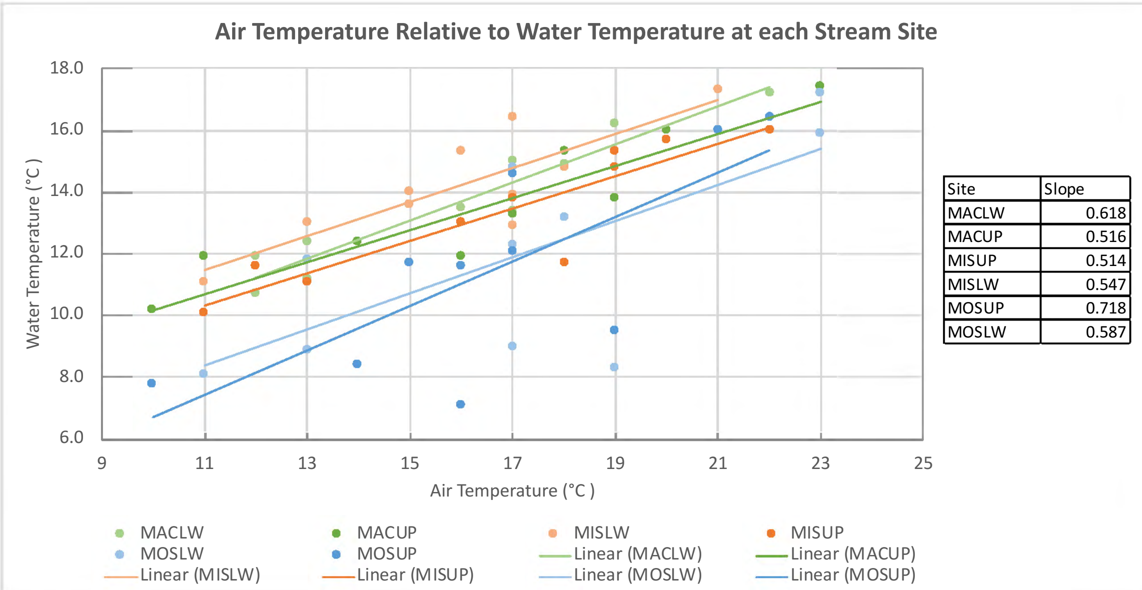

Figure 7. Relative air temperature and water temperatures at each site.

As shown in Figure 7, all of the trend lines follow a similar pattern. For most stream sites, an increase in 2 degrees of air temperature approximately corresponds with a 1 degree increase in stream water temperature (slope = 0.52). This likely reflects the seasonal pattern of more sun when there are warmer temperatures, as amount of light incident on a stream is a more important factor controlling stream temperature, rather than the air temperature itself (Beschta, et. al., 1987; Teti, 1998). At the Mosquito Creek upper site, the slope of this relationship is slightly greater (slope = 0.72) so an increase in about 1.4 degrees of air temperature corresponds with 1 degree increase in water temperature. This means that as air temperatures increase, the upper Mosquito Creek site has a greater increase in water temperatures compared to the other sites.

Water quality measurements including dissolved oxygen, pH, and Conductivity are shown in Figure 8 both as an average and a time series for all sites on Mosquito Creek, Mackay Creek, and Mission Creek. Dissolved oxygen was generally high across all sites

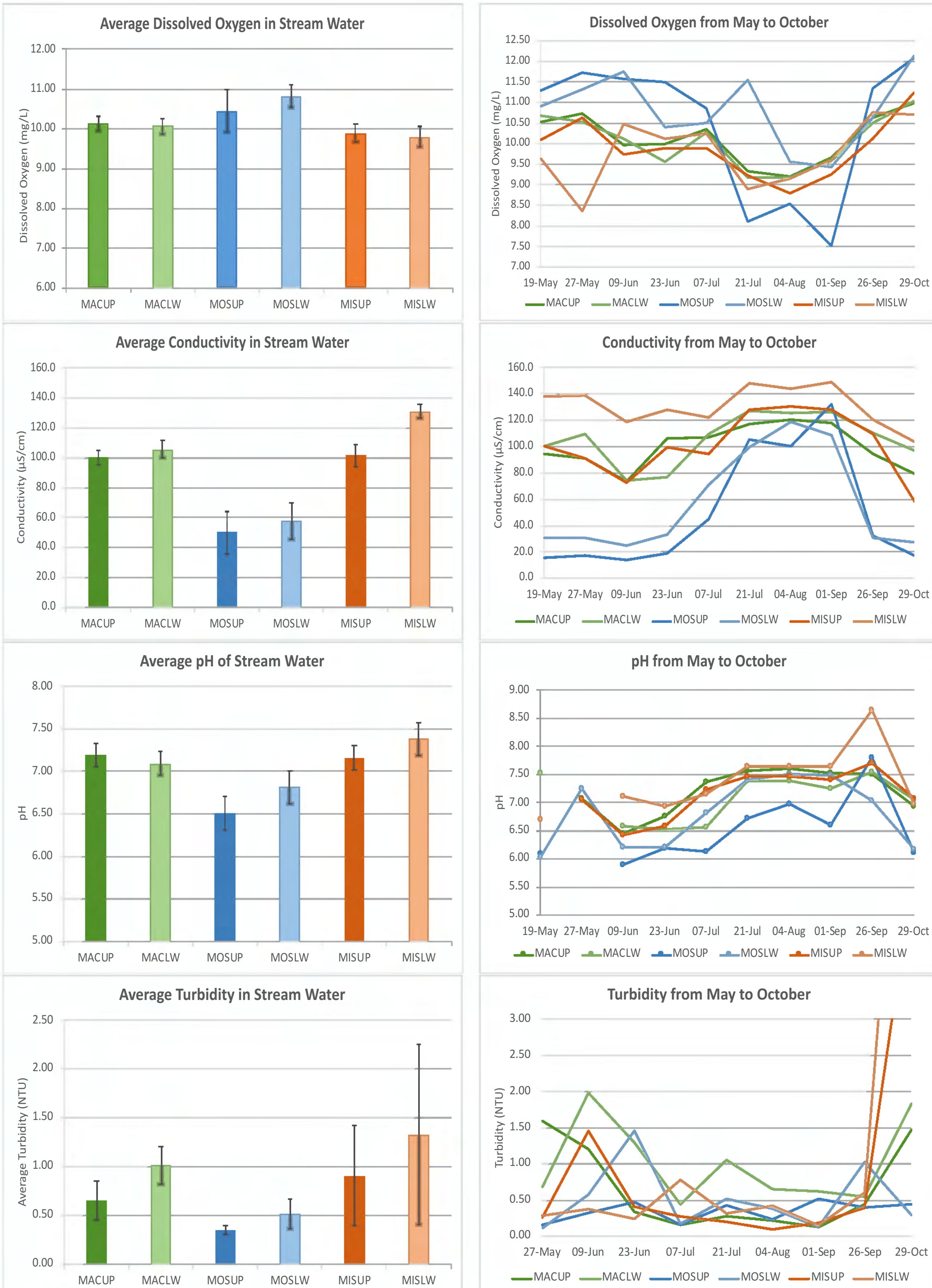

Figure 8. **Water quality at each site Mackay Creek upper and lower, Mosquito Creek upper and lower, and Mission Creek upper and lower. A. dissolved oxygen B. conductivity. C. pH. D. turbidity.**

Note: Error bars represent standard error of the mean (not the range in values measured).

and ranged from 7.5 mg/L at the Mosquito Creek upstream site at the beginning of September to 12.1 mg/L at the downstream site in late October. There was an overall pattern at all sites of greater dissolved oxygen in the spring, a decrease in the late summer, and then an increase in fall during late September and October. This swing was more pronounced at Mosquito Creek upper site.

There was a wide range of conductivity values across all sites and it ranged from 14.2 - 148.6 μS/cm. Conductivity generally was lower in the spring and then increased in the late summer. Conductivity then decreased again in the fall likely due to the cooler temperature and fall rains which began in late September. Average conductivity was notably lower at the Mosquito Creek sites compared to both the Mackay Creek sites and Mission Creek sites.

Across stream sites, the pH was reasonably close to neutral and ranged from 5.89 to 8.65 for specific measurements. Note that pH values were missing on a few dates due to a faulty meter, but overall does not show any trend over the time series. The pH does appear to be lower at the Mosquito Creek upper site (mean pH= 6.50; SE = 0.20).

Average turbidity was quite low for all stream sites as shown in Figure 8. The lowest turbidity values were recorded at Mosquito Creek lower site in May and Mission Creek upper site in August (0.11 NTU). White the greatest values were recorded in at the upper and lower Mission Creek sites in October, with 4.85 NTU and 8.69 NTU respectively. Turbidity generally shows a pattern of increased turbidity when flow levels were increased. Turbidity was higher in spring, then decreases into the late summer, and then shows an increase again in late September coinciding with the first major rainfall event of the fall. Interestingly, average turbidity appears lower at the Mosquito Creek upper site and turbidity was higher at the Mackay Creek downstream site.

4.4. Point Source Pollution Events

In addition to standard water quality monitoring two point-source pollution events were observed during the summer sampling period. The first event occurred at the lower Mackay Creek site on 16 July, 2020, at approximately 13:40. This event was characterized as an unusual turbid discharge grey/brown in colour, that was observed in the creek, along the east bank. Attempts to locate the source were unsuccessful and it appeared to clear up by 15:20. No data on the water quality during the event are available.

The second pollution event observed took place at Mission Creek at the lower site on 1 September, 2020 at approximately 14:05. A substantial amount of a white cloudy substance was observed entering the stream at the upper end of the reach and flowed down to the survey site (Figure 9). The white cloudy water was accompanied by an extremely strong rotten fruit odour. Upon further inspection, a white substance was observed to be coming from a retaining wall at the top of the reach.

Figure 9. Mission Creek lower site on 1 September 2020 during a point-source pollution event.

As the event occurred immediately after normal water quality monitoring, we were able to compare water quality before and after this pollution event. In a matter of only 20 minutes, the creek water quality was markedly reduced. The results of this event were a substantial increase in turbidity and conductivity, and an extreme decrease in pH (Figure 10; Table 6). Given that pH is on a log scale, this change represents an increase in the hydrogen ion concentration by over 75,000%. In addition, these measurements were taken at the sampling site, downstream of the pollution entry point so some dilution would have already occurred. Due to the unknown nature of the substance and health and safety concerns we did not sample directly at the source.

Table 6. Change in water quality at Mission Creek lower site on 1 September 2020 before and after pollution event.

	Time	Average Turbidity (NTU)	Turbidity SE	Temp (°C)	Conductivity (µS/cm)	pH
Before pollution event	13:42	0.16	0.02	15.3	148.6	7.65
After pollution event	14:05	38.2	3.50	15.4	321.8	4.78

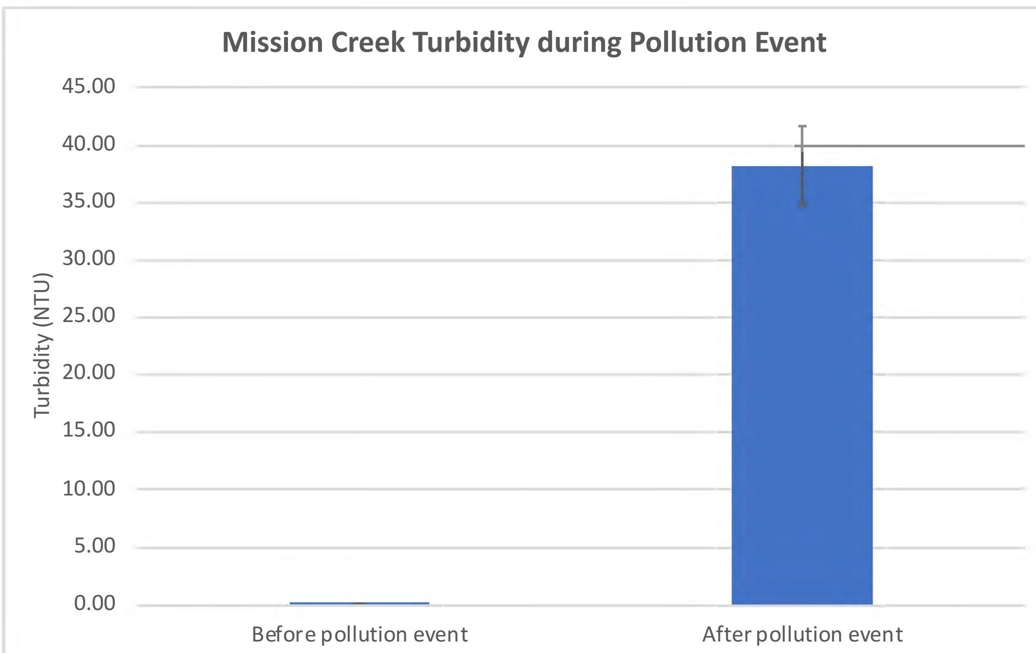

Figure 10. Change in turbidity before and after pollution event on Mission Creek 1 September 2020.

Upon further assessment of the area, we determined that the source of this substance was a Safeway on Westview Drive at the top of the ravine, as we saw the same white fluid and recognized the foul-smelling odour in the driveway behind their building. We also observed a garbage truck removing a large dumpster (likely containing rotten produce). After calls with the Emergency Management BC line to report the problem, the City of North Vancouver sent a representative to speak to the Safeway manager. The pollution issue had not been reported previously and was likely a recurring event when produce waste is disposed of and rinse water is washed down the drain (pers. comm. F. Ramsay, City of North Vancouver, 2 September 2020).

4.5. Spring Benthic Invertebrates

Benthic invertebrate samples were collected in the spring on 27, 28, 29 May 2020. Results are available for the four main sites, the Mosquito Creek upper site, Mosquito Creek Lower site, and Mackay Creek upper site and Mackay Creek lower site. In total over 900 invertebrates were examined from the spring samples including at least 30 families of aquatic invertebrates. This encompassed a diverse array of benthic invertebrates including abundant Chironomidae (lake flies) and Naididae (Naidid worms), as well as Chloroperlidae (green stoneflies) and Ameletidae (comb-mouthed minnow mayflies) (Figure 11).

Figure 11. Benthic invertebrate specimens including these stoneflies collected during spring 2020.

In addition to Surber sampling, incidental observations of Hemiptera, Gerridae (water striders) and Decapoda (crayfish) were summarized and were included in site diversity metrics as these are less likely to be captured in the Surber sampling net method but were present at many of the sites. Terrestrial invertebrate drop-ins were excluded as is recommended in the CABIN protocol to ensure that numbers and diversity are representative of only the aquatic invertebrate families present (RISC 2009). Copepods, were found at the majority of sites, and were also excluded, as these zooplankton can be very abundant at some locations which may bias estimates (RISC 2009). Complete benthic invertebrate data is provided in Appendix D.

Several metrics of benthic invertebrate abundance and diversity were calculated for the spring invertebrate samples (Table 7; Figure 12). The largest number of benthic invertebrates was found in the Mackay Creek lower site, while the lowest abundance was found at Mosquito Creek at the lower site. Total Family Richness was similar across the sites and was slightly lower at the Mosquito Creek lower site and greater at the Mackay creek lower site.

Table 7. Spring Benthic invertebrate results.

Metric	MACUP	MACLW	MOSUP	MOSLW
Abundance (Total #)	307	348	143	107
Total Family Richness	16	23	17	13
% Chironomidae	39%	24%	22%	50%
% Dominance	68%	61%	66%	70%

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, UP- Upper, LW- Lower

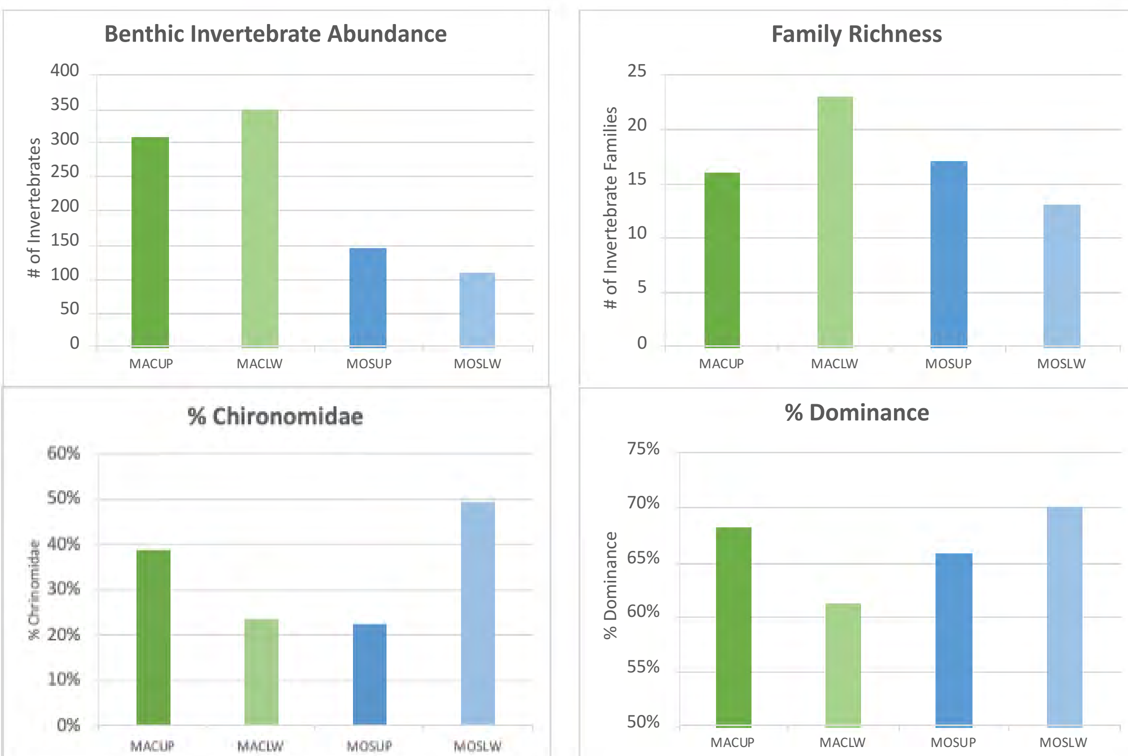

Figure 12. Spring benthic invertebrate metrics: Total abundance, % Chironomidae, total family richness, and % dominance (proportion of top three taxa) for spring invertebrate samples.

The % Chironomidae represents the proportion of the Chironomidae family in proportion to the whole sample, and was elevated at all sites, particularly the Mosquito Creek lower site. This is expected to be higher at polluted sites as Chironomidae are tolerant of pollution. However, Chironomidae are also found at healthy stream sites to some extent. The % Dominance metric measures the proportion of the top three taxa so high dominance suggests a less diverse community. The % Dominance was generally high across all sites; however, at Mackay Creek this included a rather high proportion of Ephemeroptera. There was also a decrease in dominance going downstream at Mackay Creek but an increase going downstream at Mosquito Creek.

Streamkeepers assessment methods were also used to further analyze the benthic invertebrate samples. Results of these Streamkeepers metrics are shown in Table 8. Overall, stream health ratings, which combine the four major Streamkeepers metrics (Pollution tolerance index, EPT index, EPT total ratio, and predominant taxon ratio) were Marginal-Acceptable at Mosquito Creek, and Acceptable-Good at Mackay Creek. Based on these coarse metrics we can see some of the differences between the sites where they are doing well, and where they fall short. For example, Mosquito Creek upper site had a poor EPT to total ratio; while at the family level the EPT groups were present, their abundances were very low in proportion to the full sample. Pollution tolerance index was also acceptable at Mackay Creek but it was very close to the cut-off of greater than 22 families to be considered good. Additional metrics such as density, the estimated abundance per m^2, show a pattern of decreased density at Mosquito Creek and further decreases at the downstream site.

According to the CABIN methodology, additional metrics on invertebrates were determined including Simpson's Diversity Index (1-D), Simpsons Reciprocal Index (1/D), and Simpsons evenness, which measures how evenly the individuals are spread between families in a community (Table 9). Evenness (E) can be calculated by taking Simpson's Reciprocal Index (D) and expressing it as a proportion of the maximum value D could assume if individuals in the community were completely evenly distributed (D_{max}, which equals S the family richness in a case where there was one individual per species). Evenness takes a value between 0 and 1, with 1 being complete evenness.

As determined by these statistics all sites have very low evenness, although the Simpson's diversity Index is fairly high. This highlights the disproportionate numbers of some species, such as Chironomidae, Naididae, and to some extent the stonefly group Chloroperlidae and some Ephemoptera which were found in very high numbers in some samples. Whereas many other families, such as Baetidae (a mayfly family) and Elmidae (an aquatic beetle family) were represented by only one or very few specimens per sample.

Table 8. Streamkeeper metrics of benthic invertebrate density, EPT, and stream health for spring benthic invertebrate samples.

Metric	MACUP	MACLW	MOSUP	MOSLW
Density (/m^2)	3411	3867	1589	1189
Pollution tolerance index (PTI)	22	22	21	16
PTI rating	Acceptable	Acceptable	Acceptable	Marginal
EPT Index	7	9	5	6
EPT Index rating	Acceptable	Good	Acceptable	Acceptable
EPT abundance	150	122	24	35
EPT to Total Ratio	0.49	0.35	0.17	0.33
EPT ratio rating	Marginal	Marginal	Poor	Marginal
Predominant taxon Ratio	0.39	0.24	0.38	0.50
Predominant Taxon Ratio rating	Good	Good	Good	Acceptable
Streamkeepers Site Assessment Rating	3.0	3.25	2.75	2.25
Stream Health	Acceptable	Acceptable-Good	Marginal-Acceptable	Marginal-Acceptable

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, UP- Upper, LW- Lower

Table 9. Diversity index values for spring benthic invertebrate samples.

Metric	MACUP	MACLW	MOSUP	MOSLW
Simpson's Diversity Index	0.78	0.86	0.79	0.72
Simpsons Reciprocal Index	4.63	7.04	4.81	3.60
Simpson's Evenness	0.29	0.31	0.28	0.28

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, UP- Upper, LW- Lower

The spring benthic invertebrate sampling results should be considered observational as only one sample was analyzed per site and in stream variation can be high for benthic invertebrate samples. However, these results can be used to provide an overview of general trends in benthic invertebrates and a comparison to fall samples.

4.6. Fall Benthic Invertebrates

4.6.1. Introduction and overview results

Fall Benthic invertebrate samples were collected on September 26, 27, and 29 2020. Samples were collected following the first large storm event marked by a large volume and moderate-heavy rainfall intensity, which occurred 23 September to 25 September 2020. Results are available for the four main sites, the Mosquito Creek upper and lower site, and Mackay Creek upper and lower site. In total over 3600 benthic invertebrates were examined from at least 50 families of aquatic invertebrates. Overall, the most abundant benthic invertebrate families were Chironomidae (lakeflies) at 15%, Baetidae (small minnow mayflies) at 11%, Leptophlebiidae (prongilled mayflies) at 11%, and Glossosomatidae (saddlecase maker caddisflies) at 9% of all sampled benthic invertebrates (Figure 13). Other common groups found in all samples included naididae (naidid worms), Hydracarina (water mites), Collembola (springtails), and Amphipoda (scuds). Many families were rare represented by only a few individuals, such as the Lepidoptera, Crambidae, which was only found in one sample. Complete benthic invertebrate data is provided in Appendix D.

While invertebrate samples were not analyzed for the Mission Creek upper and lower sites, observational evidence suggests that these sites have abundant, healthy populations of benthic invertebrates. For example, at the Mission Creek upper site water striders were observed on several occasions, as well as numerous caddisfly cases on the rocks, which were identified at other sites as Trichoptera, Glossomatidae. At the Mission Creek lower site, incidental observations included crayfish and waterstriders, as well as a large stonefly nymph and an Ephemoptera, Heptanaginae nymph. In addition, adult forms of both Ephemoptera and Plecoptera, Chloroperlidae were noted on multiple occasions. During invertebrate sampling, active mayflies and worms were readily apparent at the lower site. Similarly, at the upper site, lots of swimming mayflies, a very large stonefly, worms, and Glossomatidae stuck to rocks were noted.

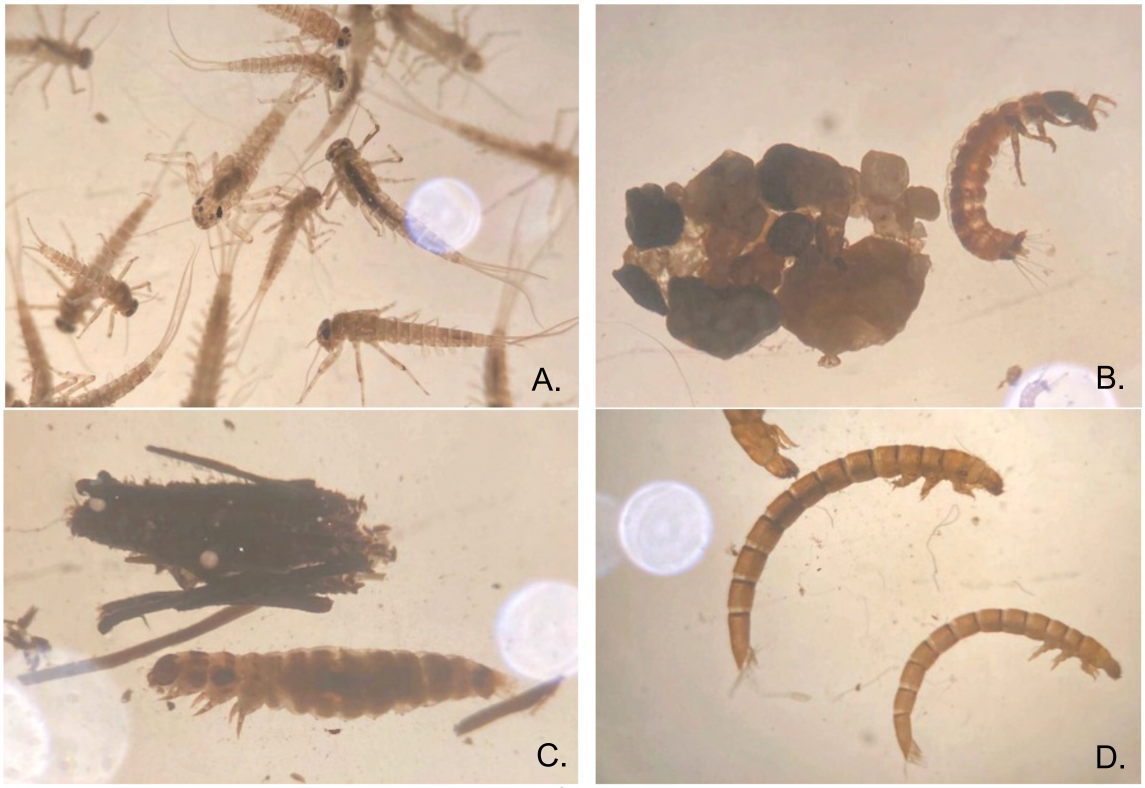

Figure 13. **A. Ephemoptera, Baetidae from the Mackay upper site. B. Trichoptera, Glossosomatidae with case made of small stones from Mackay upper site. C. Lepidoptera, Crambidae, with case of sand grains and long twigs from Mackay lower site. D. Coleoptera, Elmidae from the Mackay lower site.**

4.6.2. Basic Benthic Invertebrate Metrics

Several metrics of benthic invertebrate abundance and diversity were calculated for the fall invertebrate samples (Table 10; Figure 14). High benthic abundance in the fall was found at both the Mosquito Creek upper site and the Mackay Creek upper site. The Mackay Creek lower site was intermediate, and the Mosquito Creek lower site had low benthic abundance. Total Family Richness was similar across the sites with about 26 families on average across all samples. This was greater than in the spring samples which was about 17 families on average. The % Chironomidae was fairly low at all sites, although the Mosquito Creek upper site was slightly higher. This measure is expected to be higher at polluted sites as Chironomidae are tolerant of pollution. The % Dominance metric measures the proportion of the top three taxa so high dominance suggests a less diverse community. Dominance was generally high across all sites reflecting localized, but extremely high abundances of particular families such as: Chironomidae, Psychodidae, Baetidae, Leptophlebidae, and Glossosomatidae.

Table 10. Fall benthic invertebrate results.

Metric	MACUP	MACLW	MOSUP	MOSLW
Benthic Abundance	581	365	658	208
Total Family Richness	26	25	29	24
% Chironomidae	12%	14%	19%	14%
% Dominance	57%	48%	50%	45%

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, UP- Upper, LW- Lower
Note: These values are averages for each metric.

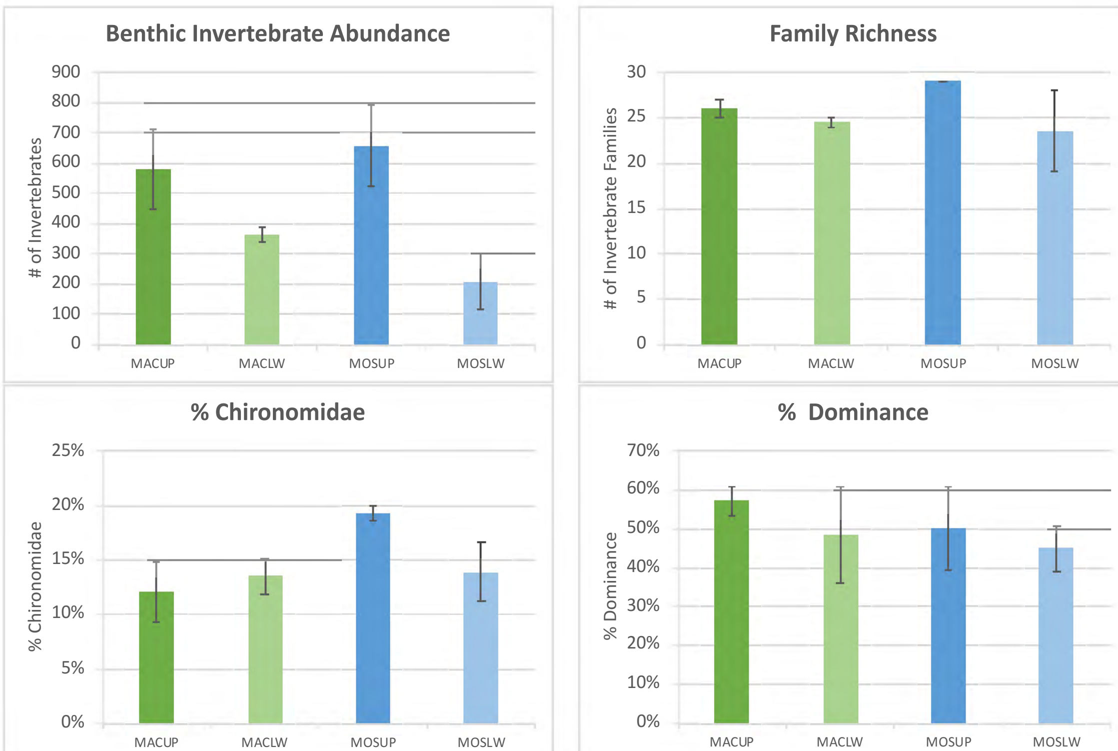

Figure 14. Fall benthic invertebrate metrics: Total abundance, % Chironomidae, total family richness, and % dominance (proportion of top three taxa) for fall invertebrate samples. Error bars show Standard Error.

Statistical Analyses were completed to further determine the significance of these patterns given the BACI design where position upstream or downstream represents the before and after condition, and Mackay and Mosquito Creeks represent control and impact conditions respectively (using a significance level of $\alpha = 0.05$). In order to analyze BACI data, t-tests have been suggested as a simple approach (Stewart-Oaten et al., 1986; Brown and Manly, 2001). First, a two sample t-test was used to compare the Mackay upper and lower control sites. Mackay upper and Mackay lower were found not to differ significantly in abundance (p value = 0.2194). Finding no difference, we can assume there was no difference between the control before and after (upstream and downstream) sites. Using a two sample t-test it was then found that there was a significant difference in abundance between Mosquito upper and lower sites (p value = 0.0389). This suggests that the benthic invertebrate abundance decreases between the upstream and downstream locations at the impact site, but not at the control site.

Others have recommended using a two-factor Anova to analyze BACI data in order to overcome various limitations with the t-test method for biological data (Brown and Manly, 2001; Underwood, 1991). A two-factor Anova was completed using the factors position (upstream/downstream) and stream (Mackay/Mosquito) as well as an interaction term for positionXstream. The interest is in the interaction term. The interpretation of a significant interaction is that the difference between the averages of the Control and Impact data are not the same in the Before period as in the After period. The results of this analysis of abundance found a significant result for position, not significant for stream, and the interaction term was not significant (Table 11). As observed in the plot of least square means the slope of the lines are not substantially different, whereas non-parallel lines would indicate a significant interaction (Figure 15). This means that we do not have evidence that the effect of position (upstream or downstream) on abundance differs depending on the stream (reference site or treatment site), or vis versa.

Table 11. Two-factor Anova effect tests results for fall benthic invertebrate abundance.

Source	Nparm	DF	Sum of Squares	F Ratio	Prob > F
Stream	1	1	3160.13	0.1433	0.7242
position	1	1	221445.13	10.0450	0.0339*
positionXstream	1	1	27261.13	1.2366	0.3285

*significance is shown in red for alpha = 0.05 level

Note that this lack of significance is most likely due to the unexpected result of there being a similar upstream to downstream pattern on Mackay Creek itself. The invertebrate abundance appears to decrease on Mackay Creek going downstream, and this overshadows any difference on Mosquito Creek. Given the variability with invertebrates, and with the limited sample size of two, it was not possible to determine an effect in this way. Further the additional data from the spring, where invertebrate abundance was not just equal but greater at the lower site, provides additional support for the conclusion that the upstream and downstream sites on Mackay Creek are not different in abundance. Based on the combined results including the t-tests this suggests there may be an impact on benthic invertebrate abundance. However, based on the ANOVA results, we cannot completely rule out that the differences aren't due to some other factor, such as natural progression downstream, as was also found at the Mackay Creek control.

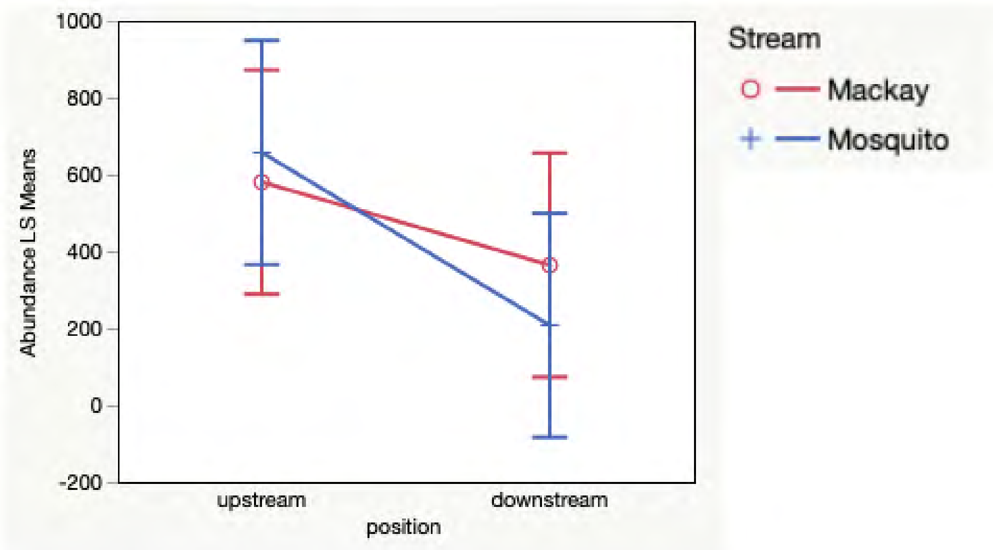

Figure 15. Least squares means plot for the two-factor Anova model of abundance based on stream and position.

Next, we completed a comparable analysis using a two sample t-test followed by two-factor Anova for benthic family richness. The two-sample t-test found that Mackay upper and Mackay lower did not differ significantly in family richness (p value = 0.6711). It was also found that there was no significant difference in family richness between Mosquito upper and lower sites (p value = 0.1688). This suggests that there was no significant decrease in benthic invertebrate family richness at the impact site. However, the one sided t-test that Mosquito lower was less than Mosquito upper was marginally significant (p value = 0.0844). Similarly, the two-factor Anova analysis did not find a significant result for position, stream, or the interaction term (p value = 0.4370). This means that there is no evidence that the effect of position (upstream or downstream) differs depending on the stream (reference site or treatment site) for invertebrate family richness.

The next analysis was for % Chironomidae. The two-sample t-test showed that Mackay upper and Mackay lower were not significantly different in % Chironomidae (p value = 0.6563). The difference in % Chironomidae between Mosquito upper and lower sites was also not significant (p value = 0.1573). This suggests that there was no significant increase in % Chironomidae at the impact site. However, it was also found that Mosquito upper had significantly greater % Chironomidae than Mackay upper (one tailed p value= 0.0394). A one-tailed test is valid as the direction of the expected difference is known in advance. The two-factor Anova analysis did not find a significant result for position, stream, or the interaction term (p value = 0.1919). This means that there is no evidence that the effect of position (upstream or downstream) on % Chironomidae differs depending on the stream (reference site or treatment site).

For % Dominance, using the two sample t-test there was no significant difference found between the Mackay Creek sites (p value = 0.5305) or the Mosquito Creek sites (p value = 0.6998). Similarly, the two-factor Anova analysis did not find a significant result for position, stream, or the interaction term (p value = 0.8572). This means that the effect of position (upstream or downstream) does not differ depending on the stream (reference site or treatment site) for % Dominance.

4.6.3. Streamkeeper Metrics

As with the spring results, fall benthic invertebrate data was also evaluated using Streamkeepers assessment methods (Table 12; Figure 16). Overall, stream health ratings, which combine the four major Streamkeepers metrics (Pollution tolerance Index, EPT index, EPT total ratio, and predominant taxon ratio) were Acceptable-Good at Mackay Creek and Acceptable at Mosquito Creek. These coarse metrics reveal areas of high and low performance on the various benthic invertebrate parameters for each site. For example, EPT to total ratio was marginal at both the upper and lower Mosquito creek sites. While the EPT groups were present, their abundances were very low in proportion to the full sample. Pollution tolerance index at the control sites was generally “good”, or close to the cut-off at 22 for “good”, at both the Mackay Creek sites and the Mosquito Creek upper site, but went down to “acceptable” at the Mosquito Creek lower site. EPT abundance follows a pattern of the greatest abundance at Mackay Creek upper site, intermediate values at Mackay Creek lower and Mosquito Creek upper sites, and then the lowest values at the Mosquito Creek lower site. Other metrics were more similar across the sites. For instance, the predominant taxon ratio, which measures the proportion of the most abundant taxon, was below the threshold of 0.40 for a rating of “Good” for all sites.

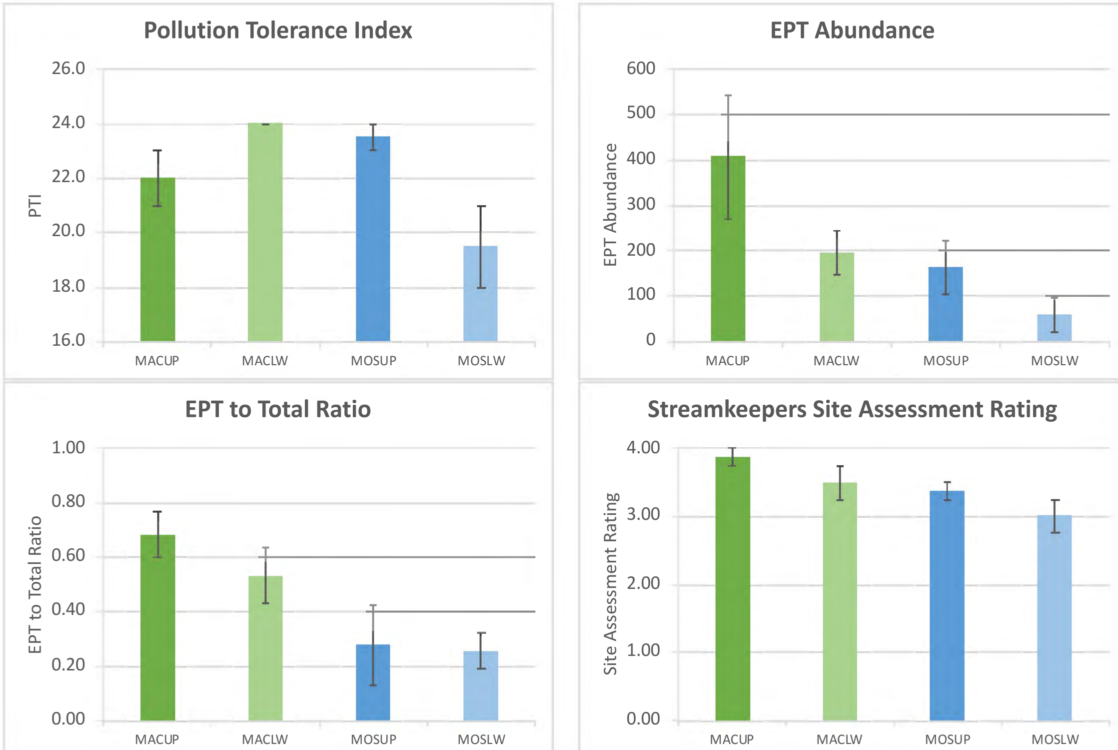

Figure 16. Select streamkeeper metrics for fall benthic invertebrates: PTI, EPT abundance, EPT to total Ratio, site assessment rating. Error bars show standard error.

Statistical Analyses were completed to determine the significance of these patterns using a two sample t-test to compare Mackay Creek sites, the Mosquito Creek sites, and a two-factor Anova using the factors position (upstream/downstream) and stream (Mackay/Mosquito) and the interaction term positionXstream.

Since the streamkeeper metric of density is a transformation of abundance (abundance/0.09 m^2), the statistical analysis was identical to that for benthic invertebrate abundance. Mackay upper and Mackay lower were found not to differ significantly in density (p value = 0.2194), while Mosquito upper and lower sites were significantly different in density (p value = 0.0389). This suggests that benthic invertebrate density was reduced at the impact site comparing the upstream to downstream locations, but not at the control site. As discussed previously, the two-factor Anova found a significant result for position (upstream/downstream), not significant for stream (Mackay/Mosquito), and the interaction term was not significant (p value = 0.3285).

Table 12. Streamkeeper metrics of density, pollution tolerance, EPT, and stream health for fall benthic invertebrates.

Metric	MACUP	MACLW	MOSUP	MOSLW
Density (/m^2)	6450	4050	7306	2311
Pollution tolerance index (PTI)	22.0	24.0	23.5	19.5
PTI rating	Acceptable	Good	Good	Acceptable
EPT index	10.0	9.0	9.5	9.0
EPT index rating	Good	Good	Good	Good
EPT abundance	407	197	163	60
EPT to total ratio	0.68	0.53	0.28	0.26
EPT ratio rating	Acceptable	Acceptable	Marginal	Marginal
Predominant taxon ratio	0.26	0.25	0.28	0.19
Predominant taxon ratio rating	Good	Good	Good	Good
Streamkeepers site assessment rating	3.88	3.50	3.38	3.00
Stream health	Acceptable-Good	Acceptable-Good	Acceptable	Acceptable

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, UP- Upper, LW- Lower

Pollution Tolerance Index (PTI) measures the number of broad taxonomic groups in each pollution tolerance category with higher scores reflecting greater numbers of more sensitive taxa. The two-sample t-test found that there was no significant difference between Mackay Creek upper and lower sites (p value = 0.2051), as may be expected, as these are both controls representing before and after conditions. However, the two-sample t-test also revealed that there was a significant difference in PTI between the upstream and downstream sites on Mosquito Creek (p value = 0.0390). This suggests that pollution impacts have occurred on lower Mosquito Creek that have led to poorer water quality and the loss of pollution sensitive taxa. The Mosquito lower site was also significantly lower than the Mackay lower site (p value = 0.0272). Using the two-factor Anova method also showed that the interaction term (streamXposition) was significant (p value = 0.0327; Table 13). As observed in the plot of least square means, non-parallel lines indicate a significant interaction (Figure 17). This is evidence that the effect of position (upstream or downstream) differs depending on the stream (control site or impact site) for PTI.

Table 13. **Two-factor Anova effect tests results for PTI of fall invertebrates**

Source	Nparm	DF	Sum of Squares	F Ratio	Prob > F
stream	1	1	4.500000	2.5714	0.1841
position	1	1	2.000000	1.1429	0.3453
stream*position	1	1	18.000000	10.2857	0.0327*

*significance is shown in red for alpha = 0.05 level

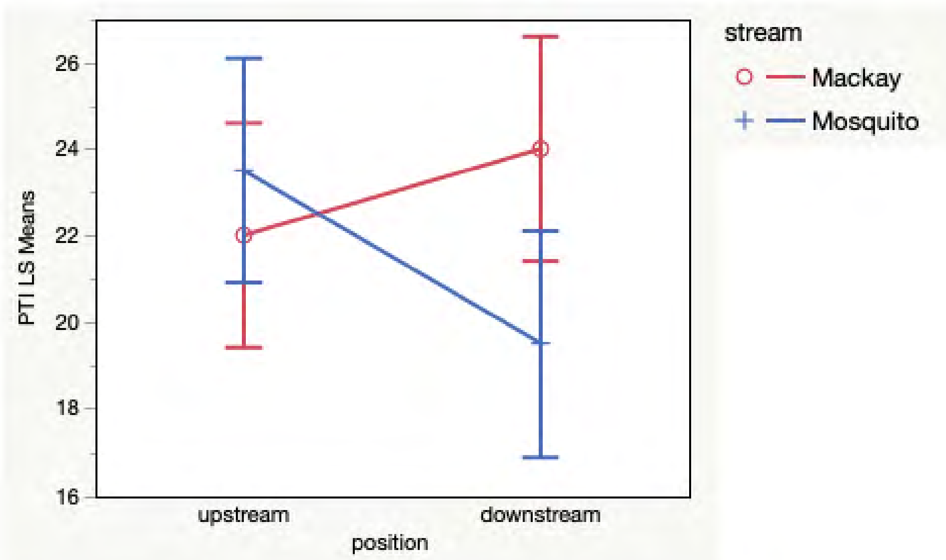

Figure 17. Least squares means plot for the two-factor Anova model of PTI based on stream and position for fall benthic invertebrates.

Results for EPT index, the number of families in the orders Ephemoptera, Plecoptera, and Trichoptera, were analyzed. Using the two-sample t-test, we found that there was no significant difference between Mackay Creek upper and lower sites (p value = 0.7269), or between the upstream and downstream sites on Mosquito Creek (p value = 0.8605). Similarly, the two-factor Anova also determined no significant effect for position (upstream/downstream), stream (Mackay/Mosquito), and the interaction term was not significant (p value = 0.9010). This suggests that there was not a substantial difference in the number of EPT families present at the sites, and that there was not a distinct impact on the EPT index going upstream to downstream that differed between the sites.

Following this, we analyzed EPT abundance, which records differences in the total of individuals of Ephemoptera, Plecoptera, and Trichoptera. Results from the two-sample t-test show that there was no significant difference between the upstream Mackay Creek site and the lower Mackay Creek site (p value = 0.1409). While it appeared that EPT abundance decreased from Mosquito Creek upper to lower, this difference was non-significant (p value = 0.4186). There was however a significant difference between Mackay upper site and the Mosquito lower site (p value = 0.0390). The two-factor Anova effect test found that stream was marginally significant (p value = 0.0790), position was not significant (p value = 0.1255), and the interaction term was also not significant (p value = 0.546). These results seem to suggest that there may already be some impacts to EPT abundance occurring at upper Mosquito Creek site, as well as decreasing EPT abundance at both streams going from upstream to downstream.

Additional detail is provided by the metric of EPT to total ratio, which compares EPT as a proportion of the total abundance. The two-sample t-test determined that there was no significant difference between Mackay upper and lower sites (p value = 0.3692) or the Mosquito Creek upper and lower sites (p value = 0.8901). However, the difference between Mackay upper and Mosquito lower was significant (p value = 0.0448), and the difference between Mackay upper and Mosquito upper was marginally significant (p value = 0.0521). This suggests that rather than a specific effect on EPT ratio at Mosquito lower, we are seeing a difference between the streams overall. This agrees with the two-factor Anova results, which show a significant result for stream (p value = 0.0311), nonsignificant for position (upstream/downstream), and nonsignificant for the interaction term.

Predominant taxon ratio is the ratio of the most abundant taxon to the total abundance, therefore a high score reflects lower evenness in the population. There were no significant differences between any of the sites measured, and all sites were in the "Good" category (0 - 0.40). This included no difference between the Mosquito sites (p = 0.3690). The two-factor Anova also found no significant effects for stream, position, or the interaction term. This suggests that extremely uneven community composition is not an issue at these sites. This is similar to the % dominance metric which also did not find any significant differences.

Finally, the last Streamkeeper metric analyzed was the overall Streamkeepers Site Assessment Rating. Following the methods used previously, the two-sample t-test did not find a significant difference between Mosquito Creek upper and Mosquito Creek lower (p value = 0.2508). However, there was a significant difference between the Mackay upper site and the Mosquito lower site (p value = 0.0352). This suggests that the overall health of Mosquito Creek lower site is impacted compared to more natural control sites such as Mackay Creek upper site. Assessing the results with the two-factor Anova revealed that stream was marginally significant (p value = 0.0647), meaning that overall scores were lower at Mosquito Creek; however, the interaction term was not significant so the before and after impact at Mosquito Creek may not be the cause.

4.6.4. Invertebrate Diversity Metrics

Additional diversity metrics were determined for fall invertebrate samples, following the CABIN methodology, namely Simpson's Diversity Index (1-D), Simpsons Reciprocal Index (1/D), and Simpsons evenness (Table 14; Figure 18) as described previously (Section 4.5).

Table 14. Diversity index values for fall benthic invertebrate samples.

Metric	MACUP	MACLW	MOSUP	MOSLW
Simpson's Diversity Index	0.86	0.88	0.87	0.90
Simpson's Reciprocal Index	7.31	9.46	8.61	10.66
Simpson's Evenness	0.28	0.39	0.30	0.45

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, UP- Upper, LW- Lower

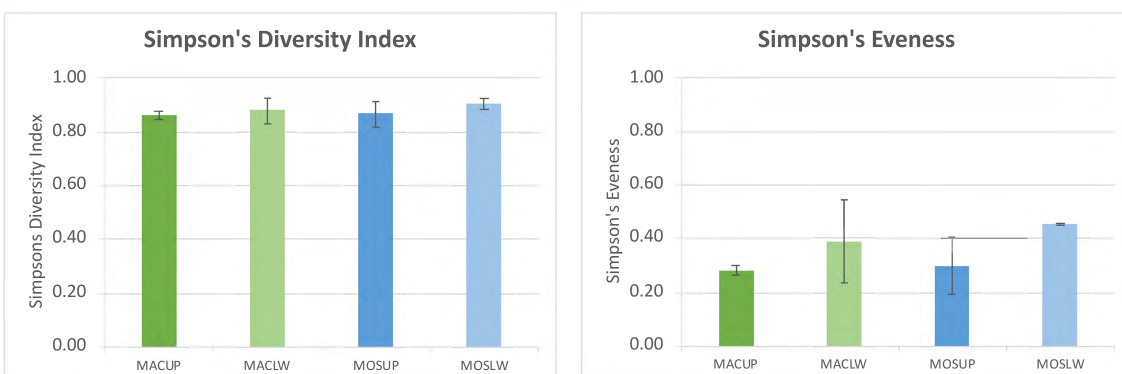

Figure 18. Diversity Indexes for Fall Benthic Invertebrates: Simpson's Diversity Index and Simpson's Eveness. Error bars show standard eror.

Simpson's Diversity Index, calculated as 1-D, may range from 0 to 1, with greater values indicating greater diversity. Average diversity was high at all sites, reflecting the large number of families observed in each sample. Simpson's Reciprocal Index is calculated as 1/D; the lowest value for this index is 1 and the highest value is equal to the number of species. The higher the value for this index, the greater the diversity of the species. Simpson's Reciprocal Index was similar across sites but appears slightly higher at downstream sites and at the Mosquito creek lower site, and lower at the upper Mackay creek site. Evenness, takes a value between 0 and 1, with 1 being complete evenness. Evenness was generally lower than Simpson's diversity Index, emphasising that the disproportionately high abundances of some species and none of the communities of benthic invertebrates were highly even. Evenness varied across the sites, following the same pattern as the reciprocal index, with higher values at the Mosquito lower site, and lower values at the Mackay upper site.

Statistical Analyses were then completed using a two sample t-test to compare Mackay Creek upstream and downstream as the before and after control, and then the Mosquito Creek upstream and downstream representing the before and after impact conditions. This was complemented by a two-factor Anova using the factors position (upstream/downstream) and stream (Mackay/Mosquito) and the interaction term positionXstream to test the effects of these factors. Based on the t-test results, Simpson's Diversity did not differ significantly between any of the sites, including the Mosquito upper and lower sites (p value = 0.2547). Similarly, the Anova model effect test showed that none of the model effects, or the interaction term were significant (p value = 0.7751). Though there appeared to be a slight increase in Simpson's diversity towards the downstream site, the effect of position was not significant (p value = 0.5128).

Although Simpsons reciprocal index appeared to show some differences between the sites, none of these were significant. Mackay upper and lower were not significantly different (p value = 0.5916) as expected. There was also no significant difference between Mosquito upper and Mosquito lower (p value = 0.6075). This suggests that the Simpson's reciprocal index was roughly equivalent at control and treatment sites, and that any potential impacts to Mosquito creek lower did not affect this aspect of the community. The Anova effects test was comparable, reporting that none of the factors were significant and the interaction term was also not significant (p value = 0.9863). This means that the effect of position (upstream or downstream) does not differ depending on the stream (reference or treatment site) for Simpson's reciprocal index.

The analysis of evenness was nearly identical to that of Simpsons reciprocal as they are both closely related. There were no significant differences between the Mackay upper and lower site (p value = 0.4624) or the Mosquito upper and lower site (p value= 0.3056). Even the apparent difference between Mackay upper and Mosquito lower cannot be considered statistically significant (one sided confidence interval p value = 0.1321). Comparing factors in the Anova model showed that while position (upstream/downstream) appeared to affect evenness it was not at the level of significance (p value = 0.2328). Further, stream was also not a significant factor and the interaction term was also not significant (p value = 0.8110). This means that we do not have evidence of a different effect of position (upstream or downstream) depending on the stream. In other words, any pattern in evenness was the same at both streams. It is also important to note that with such small sample sizes, it may not be possible to detect differences in these seemingly nuanced measures.

4.7. Impervious Surfaces Mapping

Mapping analysis was completed to characterize the extent of impervious surfaces in the watersheds and riparian buffer conditions surrounding each sampling site. As shown in the watershed map, there was extensive development through both the Mackay and Mosquito Creek watersheds (Figure 19). Residential areas, commercial buildings, and roads contributing a large total area of impervious surfaces. Large undeveloped areas in the headwaters, helped to increase the percent of impervious area, when analyzed by watershed. Total impervious area was surprisingly similar between each of the Mackay Creek watershed, Mosquito Creek watershed, and Mission Creek subwatershed. Impervious area ranged from 26-29% and pervious area ranged from 71-74% (Table 15). It is important to note that these are estimated values, based on aerial imagery interpretation and are subject to minor errors when classifying pixels. Analysis was based on imagery from 2014-2016 so may not account for more recent changes. Further, this estimates total impervious area, rather than effective impervious area so will not account for existing water infiltration modifications. Regardless, these watershed levels of 26-29% impervious surfaces appear to exceed limits suggested to impact aquatic systems 5-10% (Cuffney et al., 2010) or 10-20% (Paul and Meyer, 2001) depending on the report.

Table 15. Extent of impervious surfaces by watershed for Mosquito Creek, Mackay Creek, and Mission Creek.

Metric	Mackay Creek watershed	Mosquito Creek watershed	Mission Creek sub-watershed
Total area (m^2)	6,888,976	14,756,948	2,730,047
Impervious area (m^2)	1,931,813	4,326,425	705,153
Impervious percent (%)	28%	29%	26%
Pervious area (m^2)	4,957,163	10,430,523	2,024,894
Pervious percent (%)	72%	71%	74%

Percent impervious area, where developments removed portions of the riparian buffer, were then calculated for the immediate sampling site area, within 30m, to estimate the site level impacts to riparian buffer condition (Table 16). At the site level, there were few impacts to the majority of sites within 30 m as most sites were located within forested parks. The greatest Impervious percent was near the Mission Creek upper site with 9% impervious surfaces nearby. Mosquito Creek upper site also had some impervious surfaces within the 30 m riparian area.

Table 16. Extent of impervious surfaces at the site-scale within the 30m riparian area of study sampling points.

Metric	MACUP	MACLW	MOSUP	MOSLW	MISUP	MISLW
Total area (m^2)	2827	2827	2827	2827	2827	2827
Impervious area (m^2)	0	0	212	0	248	0
Impervious percent (%)	0%	0%	8%	0%	9%	0%
Pervious area (m^2)	2827	2827	2615	2827	2579	2827
Pervious percent (%)	100%	100%	92%	100%	91%	100%

Site codes: MAC- Mackay Creek, MOS- Mosquito Creek, MIS- Mission Creek, UP- Upper, LW- Lower

Given that the three watersheds were so similar, with such small sampling sizes, and the lack of invertebrate data for the Mission Creek subwatershed, it was not possible to compare statistically between extent of imperviousness and invertebrate data. Exploratory analyses showed that while a few apparent correlations were detected between watershed and local impervious conditions and various benthic metrics, these were most likely due to random chance, or a result of wherever both Mosquito Creek sites had similar values and the slightly higher impervious area in that watershed.

Therefore, it was not obvious whether the large-scale watershed % impervious surfaces or the local site-scale conditions surrounding a sampling point correlated with the benthic invertebrate conditions. This suggests that the relationship between impervious surfaces and the resulting site level impacts are complex and other more specific factors are likely involved. Future work with additional watersheds would be necessary to resolve this.

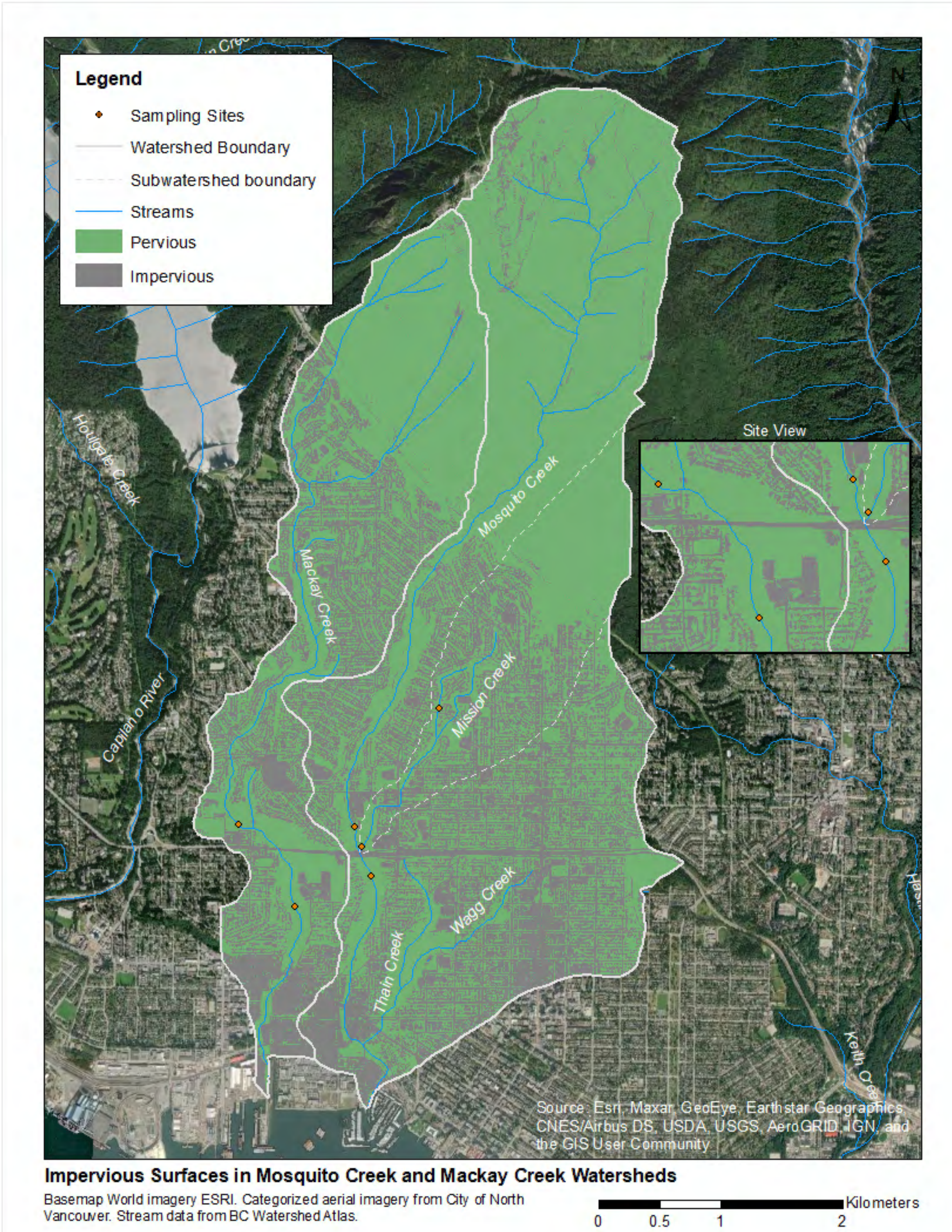

Figure 19. **Map of impervious surfaces in the Mosquito Creek and Mackay Creek watersheds. Inset map showing key sampling points.**

Chapter 5. Discussion

5.1. General Site Conditions and Impacts

The results of the general site surveys show that many urban creek sites face a number of ongoing impacts. For Mosquito Creek, both sampling sites were affected by previous logging and its forests were dominated by deciduous species rather than the coniferous species found at the control Mackay Creek and upper Mission Creek sites. Logging is known to reduce vegetation transpiration and root water storage which can increase stream flows (Rothatcher, 1970). Over time, revegetation would likely have minimized these impacts, with some research indicating a return to predisturbed conditions around 15-30 years (Rothacher, 1970; Kovner, 1956). However, since the watersheds are so developed, with extensive impervious surfaces the infiltration capacity would remain limited. The lack of coniferous forest at Mosquito Creek also contributed to the lack of stable LWD as large coniferous logs have much greater longevity than deciduous LWD which break up rapidly (Wohl and Goode, 2008). The lack of large woody debris at most of the sites may also influence the benthic community as many species live on LWD such as scrapers feeding on surface algae, LWD increases organic matter, and provides habitat heterogeneity which supports increased diversity (Deane et al 2021, Pilotto et al. 2014). However, most of the sites also had poor LWD levels so this effect was not limited to the Mosquito Creek sites.

The Mosquito Creek site was also affected by the large 1.2 km Evergreen culvert upstream of the site which would be expected to reduce sediment load and increase erosive power. Combined with the high flow volumes from the large extent of impervious surfaces in the watershed, this has led to entrenchment of the creek and larger average substrate sizes as shown by the substrate survey. The stream was also channelized due to development and is straightened, lacking the heterogeneity of the control sites. We also found that Mosquito Creek had a much greater range in discharge, both higher peak flows and very minimal flows in late summer compared to the reference sites. The creek was also very flashy following rainfall, which has been consistently associated with increased urbanization (Cuffney et al., 2010; Walsh et al., 2005).

All of the study sites were also impacted by invasive species, despite the many attempts at control through treatment and removal. The most common invasive species were

English Ivy, Himalayan blackberry, and Japanese knotweed, in addition to the more common weed species such as wall lettuce and herb Robert which were essentially ubiquitous. This agrees with previous inventory reports by City of North Vancouver who found that English ivy, Himalayan blackberry, and ground elder were the most common invasives throughout the city (2013). Mosquito Creek was typical and had several of these species, though the Japanese knotweed was not documented there. In terms of recreational use, Mosquito Creek was the highest of all the sites, where hiking and dogwalkers used the site extensively. Some areas of the creek had fencing, but where open, dogs and people often accessed the creek, which may have affected microsite conditions at these points due to greater disturbance. Therefore, recreational use of Mosquito Creek sites and others, will be an additional factor in restoration planning.

5.2. Water Quality Measurements

The results of this study show that average water quality did not appear severely impacted at Mosquito Creek. As shown by average temperature at Mosquito creek, the temperature was lower relative to the other sites. This was largely driven by the lower water temperature at Mosquito creek during spring, most likely as a result of snow meltwater from its headwaters in the Grouse Mountain area. This snow meltwater, likely buffers the stream temperature and helps to maintain cool temperatures.

The water quality guideline for temperature is + or - 1 degree Celsius change beyond optimum temperature for the life history phase of the most sensitive species present (Ministry of Environment, 2001). According to guideline documents, the optimum rearing temperature is 9-16°C for coho, 7-16°C for cutthroat, and 16-18°C for rainbow/steelhead, so the guideline would be set at 17°C. Other sources list the preferred temperatures for juvenile rearing at 11.8-14.6°C for coho, 7.3-14.6°C for steelhead, and 9.5-12.9°C for cutthroat trout (Beschta et al., 1987). At Mosquito Creek these preferred temperatures were exceeded from 21 July to 1 Sept (temperatures 14.6 or greater were recorded) and the guidelines were exceeded in August at Mosquito Creek, which indicates that water temperatures are too high in the late summer. However, temperatures at Mosquito Creek were below 18°C at all points measured during the summer, which is well below the lethal limit (23.0-25.8°C) for cutthroat and coho (Beschta et al., 1987). These results are comparable to the other sites which also had elevated stream temperatures during the late summer months. Mackay Creek also exceeded 17°C in July and August, and Mission Creek in August. These findings suggest that temperature of Mosquito creek is generally sufficient to support salmon, though high temperatures in late summer may result in decreased productivity.

The upstream and downstream sites were also very similar in temperature, although the downstream site appeared to be marginally warmer. This was likely due to the reduced canopy cover at the lower site, allowing more sunlight to reach the stream. Since there was no difference in temperature between the upper and lower Mosquito sites, or with the runoff beginning in the fall, this indicates that the temperature issues were affecting the entire creek, and not due to the inflow point between the upstream and downstream sites. Other research has also shown a consistent connection between water temperature impacts and urbanization (Cuffney et al., 2010; Nelson and Palmer, 2007).

Dissolved oxygen (DO) was also not severely impacted at any of the sites on the dates measured. On average dissolved oxygen was higher at the Mosquito Creek sites likely due to the very high flows and turbulence in the spring and fall seasons. There was a decrease in dissolved oxygen in the late summer, when flows were greatly reduced, and upper Mosquito Creek had the lowest dissolved oxygen values. However, on 1 September, 2020, dissolved oxygen was still at 7.52 mg/L at the Mosquito Creek upstream site and 9.43 mg/L at the downstream site. In general, 7 mg/L dissolved oxygen is needed to support salmonid juveniles, with reduced growth at levels below 5 mg/L and lethal affects at 2-3 mg/L (Reiser and Bjornn, 1979). Further the Canadian water quality guidelines for the lowest acceptable DO concentrations are 6 mg/L for the early life stages (CCME, 1999a). This indicates that under the present conditions, dissolved oxygen is maintained at Mosquito Creek at levels necessary for survival and growth of juvenile salmonids. Given that Mosquito Creek levels were lower at the upstream site from mid July to mid September and approaching these thresholds, additional caution is needed at this site. Further, it is important to note that these are daily site measurements, so there may be other days where measurements were not taken that have higher or lower values.

Average conductivity was generally lower at the Mosquito Creek sites. It was also lower in the spring, then increased in the late summer, and then decreased again in the fall. Levels measured at Mosquito creek ranging from 14.2 to 132.5 µS/cm are typical of rain or snow water and freshwater streams (Card et al., 2014). Conductivity measures the capacity of the water to pass electrical current which is dependent on the concentration of ions in the water such as dissolved salts, which increase conductivity (Card et al., 2014). Other compounds, such as organic compounds like oil do not conduct electrical current well and therefore would decrease water conductivity (Card et al., 2014). Conductivity is also affected by temperature, and the warmer the water, the higher the conductivity. Plotting the temperature and conductivity shows that the conductivity values from Mosquito Creek follow the theoretical relationship based on temperature quite well. The time of year of the greatest conductivity also matches with the greater temperatures in the summer. This can explain the seasonal pattern of conductivity as most likely related to temperature. Rain can also affect conductivity levels and may have contributed to the lower conductivity in the spring and fall as heavy rainfall can dilute the concentration of dissolved salts (Card et al., 2014). The lower than average conductivity at Mosquito Creek might be related to the chemical composition, such as more organic compounds; however, given that water temperature was also consistently cooler (except for late summer), and that both Mosquito Creek sites were very similar, it seems that temperature is the most likely explanation for this pattern. Another common factor affecting conductivity is the surrounding geology and soils, however given their proximity, it is expected that these are the same between Mackay and Mosquito Creek.

Other studies have found an association between urbanization impacts and increased conductivity as well as sulfate, chloride, pesticides, and PAHs (Cuffney et al, 2010; Konrad and Booth, 2002). However, as noted by Cuffney (2010), none of these variables were involved in all metropolitan areas. Overall, conductivity does not suggest any impacts to Mosquito Creek water quality, and there does not appear to be a difference between upstream and downstream sites, that would indicate an impact of the stormwater inflow between the sites.

The pH of the Mosquito Creek sites were slightly lower (more acidic) on average than the other sites at 6.5 and 6.8 at the upper and lower sites, respectively. Since, a unit change in pH corresponds to a tenfold change in the hydrogen ion concentration, small changes in pH can significantly alter the water conditions. BC water quality guidelines set the limit for aquatic life at pH 6.5-9, permitting unrestricted change is within this range (BC Government, 2006). Based on these guidelines, the pH of Mosquito Creek is still within suggested levels, though caution may be warranted to ensure it does not fall below the 6.5 limit. Specific daily values under the pH 6.5 average did occur on several occasions in May, June, July, and October. One possible explanation is that the pH meter was faulty, as there were some calibration issues; however, other sites on the same days measured normal values around pH 7 and, regardless of the calibration, Mosquito Creek sites were consistently lower.

There are a number of potential explanations for a more acidic stream including those that are naturally more acidic due to organic acids in soils (Kaufmann et al., 1992; Hodgson and Harding, 2012). Another causal factor is acid rain, which can reduce the pH of surface waters, especially after spring snowmelt, when significant amounts of accumulated acid deposition are flushed into receiving waters (CCME, 1999b; Kaufmann et al., 1992). Forest growth can also acidify soils which may then add to acidification of surface waters (Kaufmann et al., 1992). The effects on species depend on the extent and source of the aciditiy as organisms are typically unable to adapt to rapid changes in pH, which extend beyond natural ranges. However, in naturally occurring acidic streams, many freshwater invertebrates and fish are adapted to a wide range of pH (Hodgson and Harding, 2012). It is possible that the Mosquito Creek waters are naturally more acidic or are affected in part by some of these other influences. Further research would be needed to determine the contributions of these factors.

The turbidity was consistently low across sites, and lowest at the Mosquito Creek sites. The greatest values at Mosquito Creek were 1.46 NTU in June at the lower Mosquito Creek site. Turbidity generally shows a pattern of increased turbidity when flow levels were increased in the spring and fall, likely related to increased erosion from precipitation or mobilization and transport of fine particles from roadways and the streambed. There was no evidence of an increased or very high turbidity at the downstream site indicating that stormwater inflow did not increase turbidity as expected. Instead, due to the large volume of water at Mosquito Creek any turbidity is sufficiently diluted. The BC Guideline for turbidity is a Maximum increase of 8 NTUs from background levels for a short-term exposure. For a long term exposure a Maximum average increase of 2 NTUs from background levels is the limit (CCME, 2002). None of the values measured during the study period exceeded the level of 8 NTUs at Mosquito Creek. Most of the other sites also had very low turbidity on average and also did not exceed this level. However it was exceeded on one occasion at the Mission Creek lower site in October at 8.69 NTU. These results further support the idea that average water quality is not impacted at Mosquito Creek and there is not a strong increase at the downstream site. Any increases or fluctuations were well below recommended guidelines. While these results do not show an increase in average turbidity, it is also important to consider that point source events could have occurred on the days where turbidity wasn't measured.

As an illustration of this we can consider the evidence from the point source pollution events. For the pollution event on Mission Creek we observed a significant increase in turbidity up to 38.2 NTU, well in exceedance of the BC water quality guideline of 8 NTU (CCME, 2002). The pollution was derived from Safeway grocery store that was trucking away its rotten produce waste and then rinsing leftover liquid food waste down the drain. The solution was highly acidic and turbid and increased conductivity well above background levels. This case highlights how temporally variable these point-source pollution events can be. Another similar event was recorded on Mackay Creek in July with a brown colour, possibly due to erosion or construction but the source was unknown. Other Creeks in North Vancouver, Burnaby, and Coquitlam, have reported numerous similar sporadic dumping events such as Wood creek, Wagg creek and Stoney creek in recent years (Brend, 2019; Labbé, 2021; Richter, 2021). This suggests that these point-source pollution events are occurring periodically on all urban Creeks and likely Mosquito Creek as well. Initially we had focused on Mission Creek as it is a tributary of Mosquito Creek, to see how it may be affecting the main creek and potential negative impacts. However, due to the comparatively small flow volume, as measured from the discharge data, this seems unlikely to have as much of an impact. The flow volume at the Mission creek lower site is only about 14% of the volume of Mosquito Creek lower discharge. Therefore, for an isolated incident, turbidity and acidity of these sporadic events on Mission Creek would likely be diluted. However, given the numerous road crossings, stormwater inflows, and the Mission Creek tributary being repeatedly impacted by this type of event, it is possible that the cumulative effect of all this pollution is impacting Mosquito Creek water quality.

Overall, the physical water quality analyses in this study did not find evidence of a clear effect on instantaneous measurements at the times sampled, or average measurements of water quality at Mosquito Creek. Responses to urbanization are known to be variable as environmental setting plays an important role in establishing baseline conditions (Cuffney et al., 2010). For instance, differences such as average temperature and pH observed may be due to differences in the background environmental factors. Other potential issues such as warm water in the late summer were common to all of the creek sites and may represent a potential future issue with anthropogenic climate change expected in increase warm weather and reduce rainfall in the summer months in BC (Metro Vancouver, 2016). Point source pollution events continue to be an issue that affect creeks in North Vancouver including Mission Creek, a tributary of Mosquito Creek. Given the frequency of measurements, and lack of water chemistry analysis, it is difficult to be certain of the water quality from these results alone, as infrequent but toxic point-source events are known to occur in the region.

5.3. Benthic Invertebrates

5.3.1. Pollution Tolerance

The results of this study indicate that there are changes in the biological condition, the benthic invertebrate community at Mosquito Creek, due to water quality impacts over the long term. These changes are complex and multilayered, with some impacts occurring to all of Mosquito Creek and others specific to the downstream impact site. Some tests were limited by both the coarseness of the metrics, and the limited sample size. Despite these many considerations, there are several notable impacts to the benthic community at Mosquito Creek compared to control sites on Mackay Creek.

The most significant finding is that pollution tolerance (PTI) was reduced only at the downstream Mosquito site. Both methods of testing were significant, even the stricter Anova method. This suggests that pollution impacts have occurred between the upper and lower Mosquito Creek sites that have led to poorer water quality and the loss of pollution sensitive taxa. Since EPT measures, such as EPT to total ratio, were not affected between the upper and lower site, this reveals that the change in PTI is largely driven by other pollution sensitive taxa. For example, based on fall data, several pollution tolerant species and semi-tolerant species were absent from samples, including Elmidae and other aquatic beetles, Asellidae (aquatic sowbug), Sphaeriidae (pea clams), Tipulidae (cranefly larva), and Anisoptera (dragonfly larva), as well as other groups due to lower diversity more generally such as Simulidae (blackfly larva) and Physidae (pond snails). This effect was also confirmed in the spring data where Decapoda (crayfish), Asellidae (aquatic sowbug), Athercidae (watersnipe larva), Elmidae (aquatic beetles), Sphaeriidae (pea clams), Hydracarina (water mites), and Gastropoda (snails) were absent from the lower Mosquito Creek site.

Elmidae, the riffle beetle, are abundant and diverse in clear, cool waters common in the swifter portions of streams and small rivers, and are efficient clingers due to their sharp claws (Voshell, 2002). Their stress tolerance is mostly facultative, others somewhat sensitive, meaning they typically occur in pristine conditions but can withstand moderate levels of disturbance (Voshell, 2002). Similarly, crayfish are facultative; while they can withstand variability in temperature and pH they are sensitive to certain toxic substances including metals, with stream species generally less tolerant than those in lakes and ponds (Voshell, 2002). Other families such as Athercidae are somewhat sensitive (Voshell, 2002). These species contrast with some of the dominant species at the lower Mosquito Creek site, Chironomidae which include some very tolerant groups with some being very tolerant of heavy metals and petroleum products, as well as Oligochaeta which also includes some genera that are very tolerant of pollution (Voshell, 2002). Together these in high numbers are a reliable indicator of polluted conditions (Voshell, 2002). The results for PTI take into account these differences in community composition and pollution sensitivity. While not all of the benthic invertebrate metrics saw significant changes, this one is directly related to pollution.

In particular, the significance of the interaction term in the Anova model, strongly suggests that a change in water pollution occurred between the upstream and downstream site. This is potentially due to the inflow of stormwater from the Highway 1 bridge between these sites. The other input is from Mission Creek in between these points. Given that PTI was reduced in the fall, as well as the spring, this suggests that this pattern in PTI is due to long-term inputs overtime, rather than an immediate change with this particular first flush event in the fall. It is likely that repeated exposures and cumulative impacts from the highway over time as well as any point-source pollution events on Mission Creek would contribute to this result. However, I expect the dominant effects to be from the bridge inflow, as stormwater runoff is known to contain many toxic compounds. Whereas the flow from Mission Creek water may generally be good quality, as the benthics appeared healthy during sampling. Other studies have also shown impacts of stormwater on benthic invertebrates and found that the mean pollution tolerance value was one of the most consistent predictors of urbanization impacts

(Cuffney et al., 2010). These results are also limited by the small sample size and additional testing would be advisable to confirm this pattern.

5.3.2. Abundance Metrics

In addition to the types of species found, effects on benthic invertebrate abundance are also somewhat supported by our results. Abundance and Density metrics will be considered together as these essentially measure the same thing. Benthic abundance was found to be reduced at the downstream Mosquito Creek site, based on the t-test analysis. This means that there are likely effects on abundance due to water quality issues. As mentioned previously, the lack of significant effect in the Anova method was most likely due to the unexpected result of there being a similar upstream to downstream pattern on Mackay Creek itself. The invertebrate abundance appears to decrease on Mackay Creek going downstream, and this overshadows any difference on Mosquito Creek. Given the variability with invertebrates, and with the limited sample size of two, it was not possible to determine an effect in this way. Other studies have also found that abundance was not as useful of a measure to determine urbanization impacts in part because of the variability introduced by estimating abundance (Cuffney et al., 2010).

Considering the additional data from the spring for Mackay Creek, demonstrates that invertebrate abundance is likely equal or even greater at the lower site. This provides additional support for the conclusion that the upstream and downstream sites on Mackay Creek are not different in abundance. Based on the t-test result and this other information, this suggests there is likely an impact on benthic invertebrate abundance at Mosquito Creek as a result of the stormwater inflow. We cannot completely rule out that the differences aren't due to some other factor, such as natural progression downstream, however this seems less likely. Considering the stream continuum model, headwaters have greater tree cover and are generally less productive; while the mid-reaches of streams are expected to have the highest benthic diversity where there are many food sources and habitat types (DFO, 2000). As our study sites are generally around stream order 2 and 3, they would be expected to be getting more diverse/productive heading downstream. The cause of the decreased benthic invertebrate abundance on Mosquito Creek, is likely to be the water quality and pollution impacts from the stormwater inflow as this enters the stream between these points. However, these data must be interpreted with caution because of the limited sample size, and high variability with benthic abundance. It was also difficult to sample on Mosquito Creek as Surber sampler placement was limited by the depth of flow as it should not be deeper than the frame, and should be used in coarse gravel and cobbles, avoiding areas with boulders (DFO 2000; Neverman, 2016).

5.3.3. EPT Community

Next we will discuss the EPT metrics: EPT index, EPT abundance, and EPT to total ratio. These results for these metrics were somewhat unexpected, as there was not a clear impact on EPT species due to the stormwater inflow. For EPT index there was not a significant difference between the Mosquito creek upper and lower sites and the Anova test did not find a significant interaction term. This means that there was not a distinct impact on presence of EPT families; there was at least one individual of each EPT family still found at the lower site. This metric may be less affective to test for differences as it is looking at presence rather than amount. It also does not take into account size of the specimens, as differences in vigour and growth were not quantitatively measured. These results for EPT richness are similar to family richness overall which did not vary much between sites.

EPT abundance showed that the lower Mosquito Creek site was significantly lower than the control upper Mackay site which represents ideal healthy conditions. However, this was not due to changes between the upstream and downstream Mosquito sites. Rather, this was most likely due to an overall low EPT abundance on Mosquito Creek. This is supported by the marginally significant result for the effect of stream ($p = 0.0790$) in the Anova effect test. Closer inspection of the data shows that EPT made up roughly 25% of the samples at both Mosquito upper and lower. Therefore, the very low EPT abundance at the Mosquito Creek lower site, is likely due to the decrease in overall benthic abundance and the pre-existing low EPT abundance for Mosquito Creek. Results for EPT to total ratio are in agreement, finding that the EPT to total ratio was significantly lower at Mosquito Creek overall, as shown by the significant effect of stream in the Anova effect test (p value = 0.0311). The significant result of lower EPT to total ratio at the Mosquito lower site compared to the Mackay upstream control site (p value = 0.0448), is also due to the combined effect of a lower EPT ratio on all of Mosquito Creek and decreased abundance. Spring data also demonstrate lower EPT abundance at Mosquito Creek. Taken together, these results further support the idea of poorer water quality on Mosquito Creek. This suggests that, in addition to the particular outflow from the bridge, there may be other stormwater inflows due to the extent of impervious surfaces, as well as point-source pollution events causing a reduction in water quality on Mosquito Creek and reduced EPT taxa (mayflies, caddisflies, and stoneflies).

The practical significance of this level of change and whether creek functionality would be impaired with such low EPT abundances is the next question. Despite a streamkeeper rating of "good" for EPT index, the EPT to total ratios are rated as "marginal" and close to the cut-off for "poor" at 25% (DFO, 2000). This indicates that long term water quality is not at an acceptable level and other issues throughout the watershed may require consideration. These data should also be interpreted with caution as the low sample sizes and limited locations make it difficult to generalize to all of Mosquito Creek. Further sampling at additional sites and additional replicates would help to improve certainty of these results, determine whether there are impacts upstream, or whether conditions recover downstream.

5.3.4. Diversity Metrics and Minor Trends

Benthic invertebrate metrics relating to diversity provide additional detail on community structure and potential water quality impacts. Total family richness, a coarse metric for diversity, did not vary much across all sampling sites. There were no significant differences measured, except the one-sided test whether upper Mosquito Creek had greater family richness than lower Mosquito Creek was marginally significant ($p = 0.0844$). This provides limited evidence that there was reduced family diversity at the downstream Mosquito Creek site, but with such small sample sizes we cannot confirm this effect. Simpson's diversity also had an equivalent result of all sites being very similar in terms of community diversity and no significant differences found. This suggests that diversity may not be as impacted by the water quality on Mosquito Creek as a whole, or with the stormwater inflow between the upstream and downstream sites. Cuffney et al. (2010), reports similar findings that Diversity indices, functional groups, and dominance metrics were not good indicators of urbanization.

There was also no significant differences for Simpson's reciprocal index or Simpson's evenness. Interestingly, the apparent pattern for Simpson's evenness was the opposite of what I had hypothesized, with the lower Mosquito site having greater evenness. This can be explained by have such low abundances, that there were no species thriving with larger inflated values, as there were at typical healthy sites. For example, at the Mackay upstream site, sample MACUP-3 had incredibly high abundance of Trichoptera, Glossosomatidae; since this pollution sensitive group was thriving, the site has relatively low evenness. This pattern at Mosquito lower, with a diverse community but low abundances can provide insight into the type of pollution. As discussed by DFO (2000), high diversity and low numbers, signals potential toxic pollution (e.g. heavy metals, oil, acids, chlorine, pesticides) or another severe problem. This contrasts with high abundances and low diversity which would signal organic enrichment, or reduced numbers and likely lower diversity which may indicate physical problems such as erosion (DFO, 2000).

Several other metrics did not show significant results for the main test but highlight other considerations. For % Chironomidae, which measures this pollution tolerant species, there were not differences between the Mosquito Creek sites or an effect on the overall Mosquito Creek. However, the test whether the Mosquito upper site was different from the Mackay upper site control was marginally significant ($p = 0.0787$) and the one side test that % Chironomidae was greater at Mosquito upper was significant (p value = 0.0394). These results imply that there may be some issues at specific areas of the upstream site on Mosquito Creek, however the evidence was not strong. There is a considerable risk of Type I errors, when some results may appear to be significant due to random chance, so it is not recommended to run numerous correlational tests. Further, fall results for % Chironomidae were all much lower than in the spring, suggesting that even if Mosquito upper site was higher, it was still within reasonably healthy levels. Other studies have found much higher % Chironomidae, up to 56.6% in urban streams (Violin et al., 2011), suggesting that these levels (14-19%) are not unusually high.

The % Dominance, which also relates to diversity, did not vary between sites. This suggests that any impacts to water quality on Mosquito Creek did not affect dominance (the proportion of the top three taxa). Similarly, the predominant taxon ratio (which looks at the single most dominant taxa), also showed no significant effects. This may be because both pollution sensitive species, and pollution tolerant ones, were sometimes found in large amounts. As a result, dominance was relatively high at all the sites.

Cuffney et al. (2010), also found that dominance was not a good predictor of urbanization.

5.3.5. Streamkeepers Site Assessment Ranking

Overall stream health, as measured by the Streamkeepers Site Assessment Ranking was lower at Mosquito Creek compared with control sites on Mackay Creek. Notably our results comparing the Mackay upper site and the Mosquito Creek lower site were significantly different, and the Anova test indicated that the stream was marginally significant. This means that overall stream health was lower at Mosquito Creek, but there was not evidence of this being due to the before and after impact from the stormwater inflow. Overall results for the spring samples, also showed the same pattern but with more dynamic results, as populations were much lower all around at this point in the season. These findings suggest that overall the biological community at Mosquito Creek is affected, and poor water quality is an issue over the long term. In terms of the magnitude of this change, it was relatively small. Whereas Mackay Creek sites resulted in a stream health rating of “acceptable to good”, Mosquito Creek sites were “acceptable”.

The results of the benthic invertebrate sampling suggest long-term low-level impacts to water quality. At first this may seem to contradict the water quality measurements; however, average water quality metrics and intermittent measurements may not be enough to capture differences in water quality and periodic point-source pollution events. Benthic invertebrates are used as a bioindicator because they are able to capture long-term trends in stream health. This overall metric provides additional evidence for restoration of Mosquito Creek, throughout the watershed.

5.4. Impervious Surfaces Analysis

The impervious surfaces mapping analysis showed that the proportion of impervious surfaces in the Mosquito Creek watershed was approximately 29% based on analysis of aerial imagery. This was similar to Mackay Creek watershed and the Mission Creek subwatershed. It was interesting to analyze by watershed, rather than municipality, as the watershed spanned the District of North Vancouver and the City of North Vancouver, with the boundary at approximately the Trans-Canada highway. As mentioned previously, other reports found that the District of North Vancouver has 11% impervious surfaces, while the City of North Vancouver has 65% impervious surfaces when analysed by municipality (Metro Vancouver, 2019). These differences are due to the large area of forest in the Northern portion of the watershed.

Comparing the extent of impervious surfaces in Mosquito Creek (29%) to suggested thresholds for impacts suggests that Mosquito creek watershed may already be facing impacts. Some reports place the threshold for biological degradation at 10-20% impervious surfaces (Metro Vancouver, 2019; Paul and Meyer, 2001). Similarly, the stormwater planning guidebook for BC, lists 10% impervious surfaces as initial impacts to biodiversity and abundances, and by 30% impervious surfaces most urban watersheds are above the IBI threshold of 30 for stream health (MWLAP, 2002). Other reports are more cautious, suggesting there may be no initial threshold, and found that levels of 5-10% already result in some changes to the invertebrate community (Cuffney et al., 2010). This agrees with our results, which suggest overall changes to the Mosquito Creek benthic invertebrates, including reduced abundance, reduced EPT to total ratio, and lower overall site assessment rating. Our study also confirms that impervious surfaces on the order of 25-30% can be expected to impact benthic invertebrates.

At the site level, there were few impacts to the majority of study sites within 30 m as most sites were located within forested park areas. The greatest impervious percent was near the Mission creek upper site with 9% impervious surfaces in the surrounding area. This emphasizes how residential developments can be potentially infringing on riparian buffers. Further research with different buffer conditions, and additional invertebrate samples would help to determine the relationship with site level impervious surfaces conditions and benthic communities.

As neither of our study watersheds were near to the 10% hypothetical threshold level and were roughly equal in impervious percent, it was not possible to analyse further for trends in effects to benthic invertebrate metrics in relation to urbanization. A large sample size of watersheds would also be needed to statistically analyze trends. However, we can observe that despite having very similar percent of impervious surfaces, the benthic communities were very different between Mackay and Mosquito Creek. Neither the large-scale percent of impervious surfaces, or the site scale riparian buffers were solely dictating the benthic results. This highlights that the relationship between impervious surfaces and the resulting benthic communities are complex, and there are likely many intervening factors such as stormwater outfalls, point-source pollution events, culverts, logging history, as well as infiltration enhancements. For example, at the Mackay Creek site, a restoration project with a settling pond was created in Heywood Park; whereas Mosquito Creek has a stormwater inflow from the Highway 1 bridge. Future work with additional watersheds would be necessary to better understand these additional factors and response to varying levels of impervious surfaces.

Chapter 6. Recommendations for Restoration

Based on this study of Mosquito Creek water quality and benthic invertebrates we have developed a number of key restoration recommendations to improve water quality for salmonid habitat at the study site. We also address several other habitat concerns and suggest additional improvements to salmonid habitat more broadly where our data indicated potential issues. These recommendations can be used to help inform further restoration planning in the Mosquito Creek watershed.

Our recommendations for restoration of the Mosquito Creek study site are:

1. Install rain garden or infiltration gallery at the Highway 1 inflow site to improve site water quality
2. Look for other opportunities to improve watershed imperviousness throughout the watershed
3. Monitoring and enforcement to minimize point-source pollution events
4. Maintain intact riparian areas throughout the watershed to support water quality and other habitat benefits
5. Remove and monitor for invasive species to maintain native vegetation
6. Replanting native conifers to enhance stream habitat by increasing shading and LWD recruitment
7. Integration of recreational use objectives with restoration to protect sensitive habitats and increase restoration success

Our findings suggest changes to the benthic invertebrate community such as a loss of pollution sensitive taxa (lower PTI), decreased abundance at the lower site, as well as a lower EPT to total ratio and a reduction in stream health (Streamkeepers Site Assessment Rating) at Mosquito Creek overall. Based on these results, we recommend a rain garden or infiltration gallery be installed to capture stormwater and road runoff from the Highway 1 bridge outflow, in order to improve water quality for salmonids at Mosquito Creek (recommendation 1). Rain gardens are areas planted with species that tolerate high moisture, while infiltration chambers are underground containers with a permeable bottoms, designed to release stormwater gradually (City of North Vancouver, 2021a). A rain garden will be able to help trap pollutants, filter out sediment, and slow water infiltration into the creek to reduce flashiness. Given the location next to major transportation infrastructure, it is important that rain garden design be completed in conjunction with qualified Professional Engineers and the district of North Vancouver to provide input on sizing and existing infrastructure and utilities at the site.

There are several factors to consider in design of a raingarden to ensure effectiveness of the design and safety of infrastructure. First sizing must be determined, the area of the rain garden. This is based on the area of impervious surfaces draining into the rain garden, in this case the bridge and roadway, combined with the infiltration rate of the soil, and the rainfall capture target. For example, soil infiltration rates vary from 10 mm/hr up to 50 mm/hr, depending on the soil type, with more raingarden area required for slower infiltrating soils (City of North Vancouver, 2021b). In this area of North Vancouver, soil texture is primarily loamy sand (LS) for areas of Capilano soil, with potentially areas of sandy loam (SL) for Buntzen soils nearby (MoE, 1981). The saturated hydraulic conductivity is 61 mm/hr and 26 mm/hr for LS and SL respectively (Metro Vancouver, 2012). The rainfall capture target is based on how much precipitation the area receives. This is approximated by 72% of the 2-year 24 hour event necessary to meet DFO guidelines; for the West Vancouver climate station for example, the rainfall capture target would be 81 mm (Metro Vancouver, 2012). Together these factors can be used to calculate the required area. A general rule is at least 20% of the size of the impervious area that will be providing water for the garden, depending on soils drainage capacity (CRD, 2021).

Once sizing is determined, a suitable location can be selected. Several measurements are required including the elevation of the land at the highest and lowest points where the raingarden will sit and elevation of the inflow and outflow/overflow pipe, as there must be sufficient grade to ensure adequate flow (City of North Vancouver, 2021b). A sump can also be installed to provide additional water storage underground. It is also important to meet the minimum setback requirements. For example, rain gardens must be located at least 3 m away from a residence to keep water away from the foundation (City of North Vancouver, 2021b). If a suitable location and area is not available for a rain garden at this site, other options may be used such as infiltration chambers, which have a smaller footprint.

The soil composition is also important for rain garden design as it allows water infiltration and provides the growth medium for plants. Soil should be approximately 60-70% sand (2mm or finer particles), 10-20% fines (clay and silt), and 15% -20% organics (City of North Vancouver, 2021b). For example, a typical lawn-blend soil (2/3) mixed well with (1/3) compost. A 0.5% component of biochar, which is a charcoal produced by pyrolisis of biomass, is also recommended to increase removal efficiency of pollutants such as high molecular weight PAHs (Mishra, 2021). First, the area should be dug out to the required depth (approximately 60-65 cm), then filled back in with the soil mixture to a depth of 45 cm, leaving a depth of 15-20 cm for ponding (City of North Vancouver, 2021b). There should also be a perforated pipe below the raingarden and an overflow pipe to help drain the raingarden and prevent flooding. Plant species can be selected for the rain garden that can handle very wet conditions in the base of the rain garden, and drier conditions at the edge of the rain garden (Table 17). For example, Slough Sedge (*Cares obnupta*), which was found at many of the local sites, is a good option for the base and Nootka Rose (*Rosa nutkana*) may be used in edge areas. (City of North Vancouver, 2021b). Adding a 50-75mm layer of organic mulch above the soil is also recommended for erosion control, pollutant removal and to maintain infiltration capacity (CRD, 2021).

Table 17. Rain garden plant species examples for North Vancouver

Wet adapted plants (Base)	Moderate Wet adapted plants (Edge)
Deer Fern (*Blechnum spirant*)	Yarrow (*Achillea millefolium*)
Slough Sedge (*Cares obnupta*)	Kinnickinnick (*Arctostaphylos uva*-ursi)
Kelsey Dogwood (*Cornus sericea 'Kelseyii'*)	Douglas' Aster (*Aster suspicatus*)
Tufted Hair Grass (*Deschampsia cespitosa*)	Kelsey Dogwood (*Cornus sericea 'Kelseyii'*)
Soft Rush (*Juncus effuses var. pacificus*)	Tufted Hair Grass (*Deschampsia cespitosa*)
Grooved Rush (*Juncus patens*)	Coastal Strawberry (*Fragaria chilensks*)
Pacific Ninebark (*Physocarpus capitatus*)	Blue Oat Grass (*Helictotrichon sempervirens*)
Sword Fern (*Polystichum munitum*)	Oregon Iris (*Iris tenax*)
Small Fruited Bullrush (*Scripus microcarpus*)	Small Flowering Lupine (*Lupinus micranthus*)
Hardhack (*Spiraea douglasii*)	Creeping Oregon Grape (*Mahonia repens*)
Red-osier dogwood (*Cornus sericea*)	Red-Flowering Currant (*Ribes sanguineum*)
Dwarf willow (*Salix arctica*)	Nootka Rose (*Rosa nutkana*)
Slender rush (*Juncus tenuis*)	Snowberry (*Symphoricarpus alba*)

*adapted from City of North Vancouver 2021 and Hineman 2013

Raingardens have been successfully used in many areas to improve stormwater capture attenuating runoff entering streams and improving stream water quality acting as effective sinks for metals, solids, and petroleum hydrocarbons (Golden and Hoghooghi 2018. For example, several studies have found high removal rates for metals, 60-99% Zn, 65-98% Cu, and 32-100% Pb, depending on the study (Hunt et al., 2008; Li and Davis, 2009; Hunt et al., 2006; Liu et al., 2014). Such improvements in water quality have also been linked to improvements for benthic invertebrate communities. In a study of Tyron creek, Portland, Rios-Touma et al. (2014) measured the combined effects of catchment improvements (including detaining and infiltrating stormwater and revegetation), with instream enhancements (re-meander channel, add LWD, riparian plantings, install rootwads and boulders, and culvert baffles for fish passage). This was part of the City of Portland’s framework for integrated management of watershed health, which first prioritizes stormwater management before constructing instream habitat. Their results showed that streams had increase benthic invertebrate richness as well as increased EPT richness following catchment and instream restoration treatments (Rios-Touma et al., 2014). This contrasts with other studies which have found a lack of change in benthic communities from traditional channel restoration approaches alone, that do not address stormwater (Violin et al., 2011). These reports suggest that restoration using a rain garden or infiltration gallery can be an effective way to trap pollutants, improve water quality, and restore benthic communities at our study site.

In addition to improvements at the Mosquito Creek study site, further improvements to decrease impervious surfaces are needed throughout the Mosquito Creek watershed (recommendation 2). This is supported by evidence from our study such as low EPT abundance (and EPT to total ratio) as well as overall stream health (Streamkeepers Site Assessment Rating) that were reduced at both Mosquito Creek sites studied. These improvements will target water quality impacts that are affecting all of Mosquito Creek such as decreasing the flashiness, trapping pollutants from runoff, and reduce the incising and erosion issues on the creek. This may include additional rain gardens, end of pipe adjustments (such as stormwater oil and grit separators), planting trees to increase interception of precipitation, and pervious pavement options. It is essential to take a watershed approach in stream restoration and consider large-scale dynamic processes at play in stream environments. The City of North Vancouver has created an Integrated stormwater management plan (ISMP) and district of North Vancouver is also in the process of creating their ISMP (City of North Vancouver, 2016). These plans have many important goals such as 30% of all road drains will have a source control (i.e. raingardens) and that 50% of all outfalls will have a treatment structure (e.g. stormwater oil and grit separators, treatment wetlands, or treatment ponds). Continuing to meet these targets, and building on these plans will help to improve water quality. Further cooperation between municipalities to look at cross-boundary watersheds, such as Mosquito Creek will also help to ensure effective management.

Another area for improvement is to minimize point-source pollution events by increasing monitoring and enforcement (recommendation 3). As evident by our study of Mission Creek, these events can result in drastically reduced water quality that can be harmful to fish and other aquatic life. Although challenging, monitoring of the creek and maintaining a database of sightings at Mosquito Creek and others nearby will help to increase our understanding of this issue and identify potential solutions such as targeted education campaigns. Pollution events can be traced back to their sources to try and stop the pollution sources. Citizen scientists and local involvement would also be a valuable resource to reduce the cost of such a program. Another key aspect to this issue is follow-up and enforcement with cases. In addition to regular water quality monitoring, targeted testing for 6PPD-quinone, the chemical component recently identified as the primary toxic contaminant causing coho spawner mortality events by Tian et al. (2020) should be completed for Mosquito Creek and others. If this toxic compound is found it will be even more important to separate road runoff into retention ponds before entering creeks. Minimizing the effects of both runoff and point-source pollution events will help to maintain creek water quality.

Maintaining Riparian integrity is also important for the long-term improvement of Mosquito creek watershed (recommendation 4). As demonstrated by site mapping results there are areas of impingement of the 30 m riparian buffer areas at some of the sampling sites. We also note that removing the Evergreen culvert, upstream of Queens Road and daylighting further sections of Mosquito Creek could be a long-term goal to restore additional riparian areas. Removal of the culvert may also help to improve the erosion and incising issues, as the condensed flows from the culvert contribute to the scouring of the creek. Further a 30 m riparian buffer has been recommended to protect creeks from climate change impacts (Sweeney and Newbold, 2014). As shown by our water quality data, late summer temperatures already exceed ideal thresholds for salmonid juvenile growth, so riparian improvements may an important aspect of protection for climate change. The ISMP has also determined that Mosquito Creek has an average riparian forest integrity of 69% indicating there are likely other areas where riparian areas could be restored (City of North Vancouver, 2016). Other target areas should be identified and opportunities to regain riparian areas in both the City and the District pursued as discussed in the ISMP.

Removing invasive species and restoring native vegetation was also identified as a key recommendation for Mosquito Creek as there were several invasive species identified at the study sites as well as in neighbouring watersheds (recommendation 5). Ongoing work by the local municipalities and volunteer groups is addressing this concern (City of North Vancouver 2013; District of North Vancouver 2015). It will be important to continue these efforts and monitor for any new introductions. Notably, Japanese knotweed was found at the upper Mission Creek site in limited sprouts, but was not yet found at the Mosquito Creek sites. This species is known to be particularly damaging to riparian areas by increasing erosion, and may travel by fragmentation through watercourses to spread to other areas (District of West Vancouver, 2014). Early intervention is key for invasive species management so monitoring should also be a key part of future restoration work.

Treatment methods for invasive species include manual removal, chemical application, and biological control. There are also many best management practices that should be followed such as avoiding unnecessary soil disturbance, bagging or wrapping plants in a tarp before transporting, and removing plant parts and seeds from equipment and vehicles (District of West Vancouver, 2014). Optimal removal times are generally in the spring and fall when replanting to increase survival rates (City of North Vancouver, 2013). In addition to native species already found at the site (Appendix A), there are many available guides for selecting native species for replanting such as the City of West Vancouver's Best Management practices guide (2014). For instance, a recommended replacement for English ivy is Salal (*Gaultheria shallon*), which also forms a similar evergreen groundcover. Invasive removal can increase native plant biodiversity and in may help to maintain water quality.

In working towards a more natural riparian forest, typical of the CWHdm zone, we also recommend replanting riparian coniferous trees as a lack of conifers and LWD was identified during our surveys (recommendation 6). A technique called underplanting can be used to add conifers into existing deciduous forest to speed up mature forest regeneration (Harrington, 1999). Mature coniferous forest may help to increase bank stability and provide increased shading, reducing the risk of high late summer water temperatures that can impact salmonid growth. Coniferous vegetation would also support natural LWD recruitment over the long term. LWD is well known to provide numerous benefits for fish habitat such as increasing channel complexity, providing velocity refugia, increasing salmonid abundance, and has also been found to increase benthic invertebrate populations and diversity (Roni et al. 2015, Whiteway et al. 2010, Deane et al. 2021). Adding LWD complexes at these study sites on Mosquito creek is not advised as it is quite steep, and the intense hydraulic environment has been known to raft large trees up the banks and damage restoration infrastructure. Rather watershed restoration of coniferous forest would support LWD recruitment for lower gradient areas downstream. This would promote water quality and provide numerous benefits to fish habitat.

Given the location of these potential restoration sites in a major urban area, such as the City of North Vancouver, it will also be important to integrate recreational use with any restoration project (recommendation 7). High recreational use of the Mosquito Creek sites and trail network by hikers and dogs suggests that fencing around restored areas will be needed to help protect newly planted areas. Potential micro-site impacts in benthic communities were also identified near stream access points, so fencing may also be useful to maintain sensitive stream locations. Throughout this project it was clear there were many engaged community members interested in the environment so educational opportunities through signage of restoration projects or other engagement would likely be appreciated. Integration of community goals, such as recreational use, with restoration of urban streams can help to protect sensitive habitats, while also strengthening local support to increase long-term project success (Senos, 2005; Fox and Cundill, 2018).

In conclusion, this research shows that there are chronic impacts to water quality and the benthic invertebrate community abundance and pollution sensitive taxa due to a stormwater inflow. Although instantaneous measurements and averages of physical water quality did not provide evidence of impacts, the benthic community demonstrated cumulative impacts from impervious surfaces in the watershed and periodic point-source pollution events. We also documented changes in water quality due to a point-source pollution event which shows how rapid fluctuations in water quality can occur from these sporadic events. This provides evidence of the role of stormwater runoff pollution in urban salmonid habitat and demonstrates the need for holistic restoration approaches that address water quality. We were also able to determine specific site conditions at the Mosquito study site and propose recommendations for future restoration work targeting water quality and salmonid habitat improvements. Major limitations to this study are the small sample size for benthic invertebrate samples, the relatively limited number of study sites, and having only one field season for studying environmental conditions which may vary from year to year. For example, research was completed during summer of 2020 during the Covid-19 pandemic when traffic levels were unusually low for the first part of the summer, which may have resulted in less pollution accumulation. Future research on this topic would include a number of watersheds with varying levels of impervious surfaces to determine the effects on the benthic community and better understand the assumed thresholds, or if there is no threshold before impacts occur. It would also be informative to replicate these results with additional samples and at additional sampling sites to confirm conditions throughout the Mosquito Creek watershed. Further research assessing effectiveness of rain gardens for restoring stream water quality would also help to advance design of effective restoration treatments.

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Appendix A. Vegetation Species Lists

Table A1. Mackay Creek upper site vegetation July 2020

Native Vegetation Species	
Common Name	**Scientific Name**
western red cedar	*Thuja plicata*
western hemlock	*Tsuga heterophylla*
big leaf maple	*Acer macrophyllum*
red alder	*Alnus rubra*
pacific yew	*Taxus brevifolia*
vine maple	*Acer circinatum*
salmonberry	*Rubus spectabilis*
red huckleberry	Vaccinium parvifolium
indian plum	*Oemleria cerasiformis*
western sword fern	*Polystichum munitum*
lady fern	*Athyrium filix-femina*
deer fern	*Blechnum spicant*
salal	*Gaultheria shallon*
dull Oregon grape	*Mahonia nervosa*
goat's beard	Aruncus dioicus
false lily of the valley	*Maianthemum dilatatum*
common foxglove	*Digitalis purpurea*
large-leaved avens	*Geum macrophyllum*
coastal brookfoam	*Boykinia occidentalis*
slough sedge	*Carex obnupta*
common jewelweed	*Impatiens capensis*
Invasive and Exotic Species	
Common Name	**Scientific Name**
English holly	*Ilex aquifolium*
English ivy	*Hedera helix*
Japanese knotweed	*Reynoutria japonica*
common periwinkle	*Vinca minor*
horse chestnut	*Aesculus hippocastanum*
Himalayan blackberry	Rubus armeniacus
cherry laurel	*Prunus laurocerasus*
golden chain tree	*Laburnum anagyroides*
creeping buttercup	*Ranunculus repens*
common oak	*Quercus robur*
herb Robert	*Geranium robertianum*

wall lettuce	*Lactuca muralis*
common plantain	*Plantago major*

Table A2. Mackay Creek lower site vegetation July 2020

Native Vegetation Species	
Common Name	**Scientific Name**
western hemlock	*Tsuga heterophylla*
western redcedar	*Thuja plicata*
bigleaf maple	*Acer macrophyllum*
black cottonwood	*Populus trichocarpa*
red alder	*Alnus rubra*
vine maple	*Acer circinatum*
salmonberry	*Rubus spectabilis*
red huckleberry	*Vaccinium parvifolium*
western sword fern	*Polystichum munitum*
lady fern	*Athyrium filix-femina*
western brackenfern	*Pteridium aquilinum*
deer Fern	*Blechnum spicant*
dull Oregon grape	*Mahonia nervosa*
falsy lily of the valley	*Maianthemum dilatatum*
foam flower	*Tiarella trifoliata*
large-leaved avens	*Geum macrophyllum*
yellow jewelweed	*Impatiens pallida*
skunk cabbage	*Lysichiton americanus*
slough sedge	*Carex obnupta*
field horsetail	*Equisetum arvense*
Invasive and Exotic Species	
Common Name	**Scientific Name**
English ivy	*Hedera helix*
Japanese knotweed	Fallopia japonica
yellow archangel	*Lamiastrum galeobdolon*
policeman's helmet	*Impatiens glandulifera*
cherry laurel	*Prunus laurocerasus*
herb Robert	*Geranium robertianum*
wall lettuce	*Lactuca muralis*
common nipplewort	*Lapsana communis*

Table A3. Mosquito Creek upper site vegetation July 2020

Native Vegetation Species	
Common Name	**Scientific Name**
western red cedar	*Thuja plicata*
western hemlock	*Tsuga heterophylla*
bigleaf maple	*Acer macrophyllum*
red alder	*Alnus rubra*
vine maple	*Acer circinatum*
black cottonwood	*Populus trichocarpa*
salmonberry	*Rubus spectabilis*
western sword fern	*Polystichum munitum*
dull oregon grape	*Mahonia nervosa*
dwarf rose	*Rosa gymnocarpa*
Fireweed	*Epilobium angustifolium*
goat's beard	*Aruncus dioicus*
field horsetail	*Equisetum arvense*
falsy lily of the valley	*Maianthemum dilatatum*
common self-heal	*Prunella vulgaris*
Invasive and Exotic Species	
Common Name	**Scientific Name**
English ivy	*Hedera helix*
English ivy 'Needlepoint'	*Hedera helix 'needlepoint'*
Himalayan blackberry	*Rubus armeniacus*
creeping buttercup	*Ranunculus repens*
broad-leaved helleborine	*Epipactis helleborine*
common daisy	*Bellis perennis*
common plantain	*Plantago major*
wall lettuce	*Lactuca muralis*
herb Robert	*Geranium robertianum*
common nipplewort	*Lapsana communis*
common dandelion	*Taraxacum officinale*

Table A4. Mosquito Creek lower site vegetation July 2020

Native Vegetation Species	
Common Name	**Scientific Name**
western hemlock	*Tsuga heterophylla*
western red cedar	*Thuja plicata*
big leaf maple	*Acer macrophyllum*

red alder	*Alnus rubra*
bitter cherry	*Prunus emarginata*
black cottonwood	*Populus trichocarpa*
vine maple	*Acer circinatum*
salmonberry	*Rubus spectabilis*
western sword fern	*Polystichum munitum*
lady fern	*Athyrium filix-femina*
dull Oregon grape	*Mahonia nervosa*
goats beard	*Aruncus dioicus*
false lily of the valley	*Maianthemum dilatatum*
large leaved avens	*Geum macrophyllum*
stream violet	*viola glabella*
field horsetail	*Equisetum arvense*
common selfheal	*Prunella vulgaris*
yellow jewelweed	*Impatiens pallida*
palmate coltsfoot	*Petasites frigidus* var. *palmatus*
Invasive and Exotic Species	
Common Name	**Scientific Name**
Himalayan blackberry	*Rubus armeniacus*
English ivy	*Hedera helix*
yellow archangel	*Lamiastrum galeobdolon*
creeping buttercup	*Ranunculus repens*
welsh poppy	*Papaver cambricum*
ground elder	*Aegopodium podagraria*
herb Robert	*Geranium robertianum*
wall lettuce	*Lactuca muralis*
black locust	*Robinia pseudoacacia*
foxglove	*Digitalis purpurea*
common nipplewort	*Lapsana communis*
spatula-leaf loosestrife	*Lythrum portula*
sweet cicely	*Myrrhis odorata*
Kenilworth ivy	*Cymbalaria muralis*

Table A5. Mission Creek upper site vegetation July 2020

Native Vegetation Species	
Common Name	**Scientific Name**
western hemlock	*Tsuga heterophylla*
western red cedar	*Thuja plicata*

big leaf maple	*Acer macrophyllum*
red alder	*Alnus rubra*
vine maple	*Acer circinatum*
beaked hazelnut	*Corylus cornuta*
European mountain ash	*Sorbus aucuparia*
salmonberry	*Rubus spectabilis*
red huckleberry	*Vaccinium parvifolium*
salal	*Gaultheria shallon*
western sword fern	*Polystichum munitum*
western bracken fern	*Pteridium aquilinum*
lady fern	*Athyrium filix-femina*
deer fern	*Blechnum spicant*
dull Oregon grape	*Mahonia nervosa*
goat's beard	*Aruncus dioicus*
trailing blackberry	*Rubus ursinus*
Invasive and Exotic Species	
Common Name	**Scientific Name**
English Ivy	*Hedera helix*
English holly	*Ilex aquifolium*
Japanese knotweed	*Fallopia japonica*
yellow archangel	*Lamiastrum galeobdolon*
common periwinkle	*Vinca minor*
Himalayan blackberry	*Rubus armeniacus*
cherry laurel	*Prunus laurocerasus*
wall lettuce	*Lactuca muralis*
herb Robert	*Geranium robertianum*
spurge laurel	*Daphne laureola*
lady's thumb	*Persicaria maculosa*
ground elder	*Aegopodium podagraria*
foxglove	*Digitalis purpurea*

Table A6. Mission Creek lower site vegetation July 2020

Native Vegetation Species	
Common Name	**Scientific Name**
western redcedar	*Thuja plicata*
western hemlock	*Tsuga heterophylla*
big leaf maple	*Acer macrophyllum*
black cottonwood	*Populus trichocarpa*

red alder	*Alnus rubra*
vine maple	*Acer circinatum*
beaked hazelnut	*Corylus cornuta*
salmonberry	*Rubus spectabilis*
red huckleberry	*Vaccinium parvifolium*
western sword fern	*Polystichum munitum*
lady fern	*Athyrium filix-femina*
western brackenfern	*Pteridium aquilinum*
deer fern	*Blechnum spicant*
false lily of the valley	*Maianthemum dilatatum*
field horsetail	*Equisetum arvense*
goat's beard	*Aruncus dioicus*
three-leaved foamflower	*Tiarella trifoliata*
Invasive and Exotic Species	
Common Name	**Scientific Name**
English Ivy	*Hedera helix*
Japanese knotweed	*Fallopia japonica*
English holly	*Ilex aquifolium*

Appendix B. Site Photos

Figure B1. Mackay Creek upper site photos. A. upstream. B downstream. C. Across D. Canopy cover. E. Substrate. F. Underwater.

Figure B2. **Mackay Creek lower site photos. A. upstream. B downstream. C. Across D. Canopy cover. E. Substrate. F. Underwater.**

Figure B3. **Mosquito Creek upper site photos. A. upstream. B downstream. C. Across D. Canopy cover. E. Substrate. F. Underwater.**

Figure B4. **Mosquito Creek lower site photos. A. upstream. B downstream. C. Across D. Canopy cover. E. Substrate. F. Underwater.**

Figure B5. **Mission Creek upper site photos. A. upstream. B downstream. C. Across D. Canopy cover. E. Substrate. F. Underwater.**

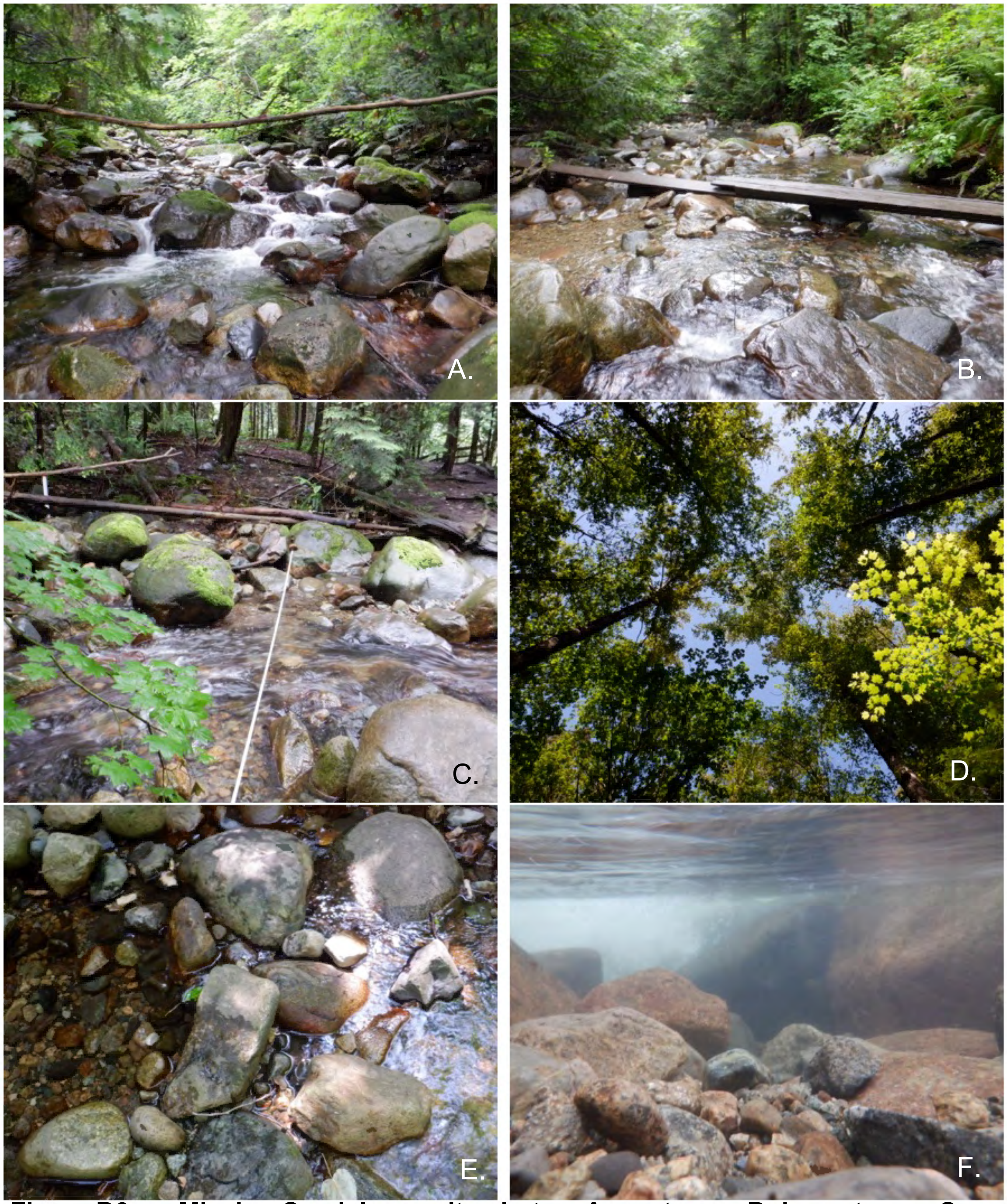

Figure B6. **Mission Creek lower site photos. A. upstream. B downstream. C. Across D. Canopy cover. E. Substrate. F. Underwater.**

Appendix C. Recreational Use Survey

Recreational survey data was collected by recording hikers and dogs visiting a site within a one hour period. Surveys were completed opportunistically during fieldwork times with an effort to visit sites at different times of the day with some surveys from morning, noon, and evening times and at least 5 to 6 surveys for each site. Survey data was collected from June to September representing the summer months. Results of the survey are summarized in Table C1. As shown in Figure C1, the highest recreational use was observed at the Mosquito Creek trail center which corresponds with the Mosquito Creek lower site. No recreational visits were recorded after surveying began at the upper Mission Creek site. Recreational use of parks can help to inform restoration planning.

Table C1. Recreation survey data at Mosquito Creek and Mackay Creek

Trail/park	Average Hikers (per hour)	Average Dogs (per hour)
Mosquito Creek trail center	64	23
Mosquito Creek trail Delbrooke	33	14
Mackay Heywood	16	4
Mission Creek trail	7	4
Mackay Murdo Fraser	2	1
Mission Creek Upper	0	0

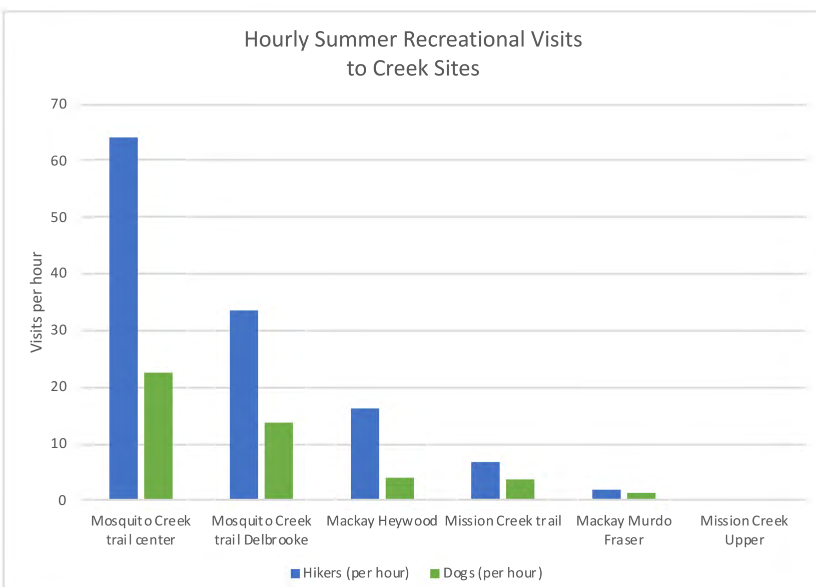

Figure C1. Recreation survey results for Mosquito, Mackay and Mission Creek showing average visits per hour during summer months.

Appendix D. Benthic Invertebrate Data

Table D1. Spring benthic invertebrate dataset

Invertebrates		Sample Jar			
Order	**Family**	**MOSUP-2**	**MOSLW-2**	**MACUP-2**	**MACLW-3**
Insects					
Ephemeroptera					
	Heptageniidae	5	10	3	2
	Ameletidae	7	4	34	7
	Leptophlebiidae		8	25	14
	Baetidae	1	1		14
	Ephemerellidae				2
	Unknown Ephemoptera	7	5	23	54
Odonata Anisoptera					
	Anisoptera				
Odonata Zygoptera					
	Zygoptera				
Plecoptera					
	Capniidae				
	Chloroperlidae		2	56	23
	Leuctridae				
	Nemouridae				
	Peltoperlidae				
	Perlidae				1
	Perlodidae				
	Pteronarcyidae				
	Taeniopterygidae				
	Unknown Plecoptera	3	3	2	1
Hemiptera					
	Corixidae				
	Gerridae	1			
	Unknown Hemiptera				
Megaloptera					
	Sialidae				
	Corydalidae (fishfly)				
	Corydalidae (Dobsonfly)				
Neuroptera					
	Sisyridae				
Trichoptera					

Invertebrates		Sample Jar			
Order	Family	MOSUP-2	MOSLW-2	MACUP-2	MACLW-3
	Brachycentridae			1	
	Limnephilidae			3	1
	Lepidostomatidae		1		2
	Polycentropodidae			1	
	Unknown Trichoptera	1	1	2	1
Lepidoptera					
	Crambidae				
Coleoptera					
	Elmidae (Microcylloepus)	1		3	6
	Elmidae (Lara)				1
	Psephenidae				
	Hydrophilidae	1			
	Unknown Coleoptera				1
Diptera					
	Athercidae	2			
	Culicidae				
	Empididae	1			1
	Tipulidae		1	1	1
	Simuliidae				
	Chironomidae	32	53	119	82
	Stratiomyidae		1		
	Ceratopogonidae	1		1	1
	Dixidae				4
	Unknown Diptera	5	1		4
Other Arthropods					
Hydracarina	Hydracarina	3		3	16
Collembola	Collembola				
Decapoda (crayfish)	Decapoda				
Amphipoda (scud)	Amphipoda	8	1	2	11
Isopoda (sowbug)	Asellidae	1			
Copepoda	Copepoda	47		7	179
Other Invertebrates					
Oligochaeta	Oligochaeta	8	1	1	13
Oligochaeta	Naididae	54	12	26	77
Bivalva (clams)	Sphaeriidae				5
Gastropoda	Prosobranchia				
Gastropoda (snails)	Pulmonata other				

Invertebrates		Sample Jar			
Order	**Family**	**MOSUP-2**	**MOSLW-2**	**MACUP-2**	**MACLW-3**
Gastropoda	Pulmonata, planorbidae			1	1
Hirudinea (leeches)	Hirudinea				
Nematoda	Nematoda				
Nematomorpha	Nematomorpha		1		1
UNKNOWN	Unknown	1	1		1
Terrestrial Invertebrates					
Hymenoptera (adults)	Apoidea			1	
	Formicidae			1	
	Ceraphronidae	1			
Diptera (adults)	Unknown Diptera	1		2	5
	Ceratopogonidae	1			
	Phoridae	1			
	Mycetophilidae	1			
	Sciaridae			1	
	Chironomidae				2
Total Specimen Abundance		195	107	319	534
Total Benthic Abundance		143	107	307	348
Total Family Richness		17	13	16	23

Table D2. Fall benthic invertebrate dataset

Invertebrates		Sample Jar							
Order	**Family**	**MOSUP-3**	**MOSUP-5**	**MOSLW-2**	**MOSLW-5**	**MACUP-3**	**MACUP-5**	**MACLW-1**	**MACLW-3**
Insects									
Ephemeroptera									
	Heptageniidae	6	14	1	2	1	2	1	
	Ameletidae				1				
	Leptophlebiidae	32	47	7	32	103	114	52	24
	Baetidae	10	22	6	12	140	46	138	38
	Ephemerellidae	4	13		9	12	7	8	8
	Unknown Ephemoptera		18	1	6	69	3	3	24
Odonata Anisoptera									
	Libellulidae								1
	Unknown Anisoptera							1	
Odonata Zygoptera									
	Unknown Zygoptera			1					
Plecoptera									
	Capniidae								
	Chloroperlidae	21	1		4	6	4	2	
	Leuctridae								
	Nemouridae	3	12		5		3	1	1
	Perlidae				1				
	Perlodidae								
	Pteronarcyidae							1	
	Taeniopterygidae								
	Unknown Plecoptera	1	9	1		2	1		3

Invertebrates		Sample Jar							
Order	**Family**	**MOSUP-3**	**MOSUP-5**	**MOSLW-2**	**MOSLW-5**	**MACUP-3**	**MACUP-5**	**MACLW-1**	**MACLW-3**
Hemiptera									
	Corixidae								
	Gerridae				1				
	Hebroidea	22	2	7	15	9	9	11	
	Unknown Hemiptera						1		
Megaloptera									
	Sialidae								
	Corydalidae (fishfly)								
	Corydalidae (Dobsonfly)								
Neuroptera									
	Sisyridae								
Trichoptera									
	Glossosomatidae	7	17	1	9	190	59	13	13
	Brachycentridae								
	Limnephilidae							1	
	Lepidostomatidae	6	7	1	5	2	1		3
	Leptoceridae								
	Psychomyiidae			3					
	Polycentropodidae		10		1	3	2	6	1
	Hydropsychidae	1	8		1	9	3	7	
	Rhyacophilidae								
	Phryganeidae						1		
	Unknown Integripalpia	12	44		6	8	23	10	33
	Unknown Annulipalpia								

Invertebrates		Sample Jar							
Order	**Family**	**MOSUP-3**	**MOSUP-5**	**MOSLW-2**	**MOSLW-5**	**MACUP-3**	**MACUP-5**	**MACLW-1**	**MACLW-3**
	Unknown Trichoptera		1	1	3			2	
Lepidoptera									
	Crambidae							1	
Coleoptera									
	Elmidae (Microcylloepus)	3	1			4	1	3	10
	Elmidae (Lara)				2			1	1
	Psephenidae								
	Hydrophilidae	3	1		1				
	Gyrinidae	1							
	Carabidae	12							
	Dryopidae								1
	Melyridae				1				
	Unknown Coleoptera	1						1	
Diptera									
	Athercidae								
	Culicidae								
	Empididae								
	Tipulidae	7				1		1	
	Simuliidae	3	5		1	1	4	11	1
	Chironomidae	147	104	13	50	66	67	46	52
	Stratiomyidae								
	Ceratopogonidae	2	1		1	2			
	Dixidae				3	2	2		
	Psychodidae	290	2		3	4	2		2

Invertebrates		Sample Jar							
Order	**Family**	**MOSUP-3**	**MOSUP-5**	**MOSLW-2**	**MOSLW-5**	**MACUP-3**	**MACUP-5**	**MACLW-1**	**MACLW-3**
	Ephydridae	2	3		3	3	4	4	3
	Muscidae	3							
	Tabanidae	39							
	Phoridae		1						
	Unknown Diptera	16	10	2	6	7	6	7	11
Other Arthropods									
Hydracarina (mites)	Hydracarina	44	21	2	35	29	27	22	16
Collembola (springtail)	Collembola	29	1	3	23	4	5	1	4
Decapoda (crayfish)	Decapoda								
Amphipoda (scud)	Amphipoda	6	12	12	11	9	12	4	17
Isopoda (sowbug)	Asellidae	1					1		1
Other Invertebrates									
Oligochaeta	Oligochaeta	8	47	24	7	2	7	2	27
Oligochaeta	Naididae	34	56	22	27	15	12	20	24
Bivalva (clams)	Sphaeriidae		14	1			4		12
Gastropoda (snails)	Prosobranchia								
Gastropoda	Pulmonata, planorbidae		1	2		1			2
Gastropoda	ancylidae		3	2			5		1
Gastropoda	physidae		1			1			
Hirudinea (leeches)	Hirudinea								
Nematoda	Nematoda	1	1				1		
Nematomorpha	Nematomorpha			1					
UNKNOWN	Unknown	15	13	2	13	6	11	6	8

Invertebrates		Sample Jar							
Order	**Family**	**MOSUP-3**	**MOSUP-5**	**MOSLW-2**	**MOSLW-5**	**MACUP-3**	**MACUP-5**	**MACLW-1**	**MACLW-3**
Terrestrial Invertebrates									
Hymenoptera (adults)	Apoidea								
	Formicidae	2			2	1			
	Ceraphronidae								
	Mymaridae				1				
	Unknown Hymenoptera	1			2			1	
Coleoptera (adults)	Unknown Coleoptera		1	1				2	1
	Ptiliidae		1						
Thysanoptera (adults)	Thysanoptera		2			3	2	2	
Psocoptera (adults)	Psocoptera	3	1		6	1	5	5	1
Neuroptera (adults)	Unknown Neuroptera		1						
Diptera (adults)	Unknown Diptera	9	23	3	21	24	9	7	6
	Ceratopogonidae				1				
	Phoridae	1							
	Mycetophilidae				2				
	Sciaridae		1			1	2	1	1
	Chironomidae				20		2	1	
	Dixidae	1			1				
	Anisopodidae						1		
	Psychodidae	2							
	Sphaeroceridae	1							
	Drosophilidae (pupa)	9							
Lepidoptera (adult)	Geometridae						1		
	Unknown Lepidoptera							2	

Invertebrates		Sample Jar							
Order	**Family**	**MOSUP-3**	**MOSUP-5**	**MOSLW-2**	**MOSLW-5**	**MACUP-3**	**MACUP-5**	**MACLW-1**	**MACLW-3**
Hemiptera (adult)	Pentatomoidea	1							
Arachnid	Arachnid	2	1		1	1	3	1	1
Other									
Chironomidae case	Chironomidae case	62	32	26	21	13	67	43	57
Zooplankton									
Copepoda (copepods)	Copepoda	present	present	present	present	present	present	present	present
Cladocera (water fleas)	Cladocera			present	present		present		
Total Specimen Abundance		886	586	146	378	755	542	452	409
Total Benthic Abundance		792	523	116	300	711	450	387	342
Total Family Richness		29	29	19	28	25	27	25	24