# Assessment of an urban floodplain reconnection project: A case study from the Mamquam River basin, BC.

**by**

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## Declaration of Committee

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## Abstract

Dikes and culverts have limited access to off-channel rearing habitats important to juvenile coho salmon (Oncorhyncus kisutch). This study assessed the success of a floodplain reconnection project in Squamish, BC, at providing rearing habitats. Recommendations on restoration priorities within the area were also provided. A single-season, multi-scale occupancy model was used to estimate the probability of occurrence and detection of juvenile coho during the summer. Regression models were used to assess water and habitat quality and identify relationships with juvenile coho metrics. Culverts were also scored for fish passage. The results of this study indicate that the reconnection project was overall successful. Coho non-detections occurred in areas with poor dissolved oxygen and culvert passage issues. Restoration actions should focus on improving water quality in these areas, and protection of areas of high CPUE. Positive relationships between stream productivity and coho metrics indicates the importance of future studies on macroinvertebrate supply.



**Keywords**: Coho salmon; Escape cover; Rearing habitat; Floodplain reconnection; Urban channels; Mamquam River

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# Chapter 1. Introduction and study objectives

## 1.1. Introduction

Floodplains are an important component of alluvial rivers. Floodplains create diverse aquatic habitats through off-channel habitats, wetlands, and tidal sloughs. They allow for connectivity of the mainstem with to its surrounding landscape. Connected floodplains provide habitat heterogeneity that allow juvenile salmonids access to energetically beneficial environments year-round. Floodplain habitats are often areas of high productivity that can support high macroinvertebrate densities (Sommer et al. 2001; Bellmore et al. 2013). Habitat heterogeneity in connected floodplains also allow for the daily migration of salmonids to comparatively warmer temperatures that benefit food digestion (Baldock et al. 2016). Off-channel floodplain habitats are associated with lower energy habitats with greater instream complexity than main stem habitats. This is provided by escape cover structures instream and within the bank, which provide benefits such as predator protection and low velocity refugia (Allouche 2002; Orrock 2013). Coho salmon (*Oncorhyncus kisutch*) are also highly dependent on off-channel habitats for rearing (Nickelson et al. 1992; Nickelson et al. 1998).

Rivers are often channelized and disconnected from their floodplains in urbanized regions. This can occur from urban features such as roads, or through hard engineering structures such as dikes and culverts. Although engineering structures serve to protect humans from flooding, they often have significant and detrimental impacts to the availability of off-channel habitats. This has been observed in many urban Pacific Northwest watersheds (Finn et al. 2021). There are provincial and federal guidelines for culvert design to promote fish passage, however culverts in fish-bearing streams can fail to meet these guidelines resulting in barriers to fish passage (Gibson et al. 2005). When fish passage guidelines are not met or prioritized in culvert design, a passage barrier can result. This can occur from height differences between the bottom of the culvert compared to the water level, and from channel constriction resulting in an increase of water velocity to speeds which juvenile salmon cannot overcome (Davis & Davis 2011). These issues can become barriers to juvenile coho when swimming upstream. Freshwater rearing species such as coho salmon are disproportionally affected by culverts located within lowland floodplains and streams that restrict access to off-channel habitat (Sheer & Steel 2006). In BC, over 170,000 culverts have been identified as barriers to at least 1 km of upstream fish habitat (Fish Passage Technical Working Group 2014). In addition to urbanization disrupting channel connectivity and modifying channel hydrology through the construction of roads and other urban infrastructure (Walsh et al. 2005; Anim et al. 2018; Feist et al. 2017), urban channels often experience habitat simplification and reduced water quality from stormwater runoff (Chow et al. 2019). Coho salmon are also sensitive to changes in water quality (Scholz et al. 2011; Feist et al. 2017; Chow et al. 2019) and have been referred to as a sentinel species and indicator of stream health (Feist et al. 2017). Both the quality and accessibility of freshwater habitat is an important determinant of the viability within coho populations (Nickelson & Lawson 1998).

Juvenile coho salmon rear in small freshwater off-channels in the summer and through the winter. Overwintering survival of juvenile coho has been positively correlated with fish size after summer rearing (Brakensiek & Hankin 2007; Pess et al. 2011). A lack of overwintering habitat is often considered as a bottleneck in juvenile coho populations in the Pacific Northwest (Nickelson et al. 1992). Rearing habitat for both summer and overwintering are important for juvenile coho population persistence, as the conditions during the freshwater rearing stage are associated with the production of smolts and the recruitment of adult coho (Lawson et al. 2004). Habitat-based restoration, such as the construction of artificial off-channel habitat, has shown to be effective at increasing salmonid production by providing suitable rearing habitat (Solazzi et al. 2000; Ogston et al. 2015; Morley et al. 2005). However, monitoring for the effectiveness of habitat-based ecological restoration is generally limited (Katz et al. 2007). Assessments of the relationships between fish metrics (e.g., abundance, growth) and habitat parameters are important to understand factors limiting salmonid populations and effective restoration (Ebersole et al. 2009).

Presence/non-detection surveys are commonly used to monitor the distribution of species in habitats by integrating presence data into occupancy models (Mackenzie et al. 2018). These models estimate the probability of occupancy and detection of target species in a study area (Mackenzie et al. 2018). Pairing of habitat characteristics with presence/non-detection data can be used in occupancy models to analyze correlations between habitat characteristics with the probability of occupancy and detection for species of interest (Albanese et al. 2014; Mackenzie et al. 2018. This can therefore help identify the most important habitat features for juvenile coho salmon (Sethi & Benolkin 2013; Flitcroft et al. 2014).

Gee traps are a low-disturbance method to collect presence/non-detection data for juvenile coho salmon. Gee traps have a relatively high detection efficiency for this species (Bryant 2000; Sethi & Benolkin 2013). Catch per unit effort data (CPUE) can be calculated from Gee traps as a measure of relative abundance. These data be used in the analyses of salmonid-habitat associations (Schoen et al. 2021). Regression analyses (such as generalized linear models and generalized linear mixed-models) can then be used to identify important habitat parameters for juvenile coho rearing (Bradley et al. 2016).

The focus of this research was to assess the success of a multi-stage floodplain restoration project, the Mamquam River Reunion (MRR), finalized 10 years ago. The MRR was initiated in response to recommendations from the Squamish River Watershed salmon recovery plan in 2005 (Golder Associates Ltd 2005) to improve declining stocks of Georgia basin coho populations. One of the primary issues identified was the disconnection of the Mamquam River (and the Squamish River) from its historical floodplain and estuary habitat. This floodplain disconnected was the results of extensive dike construction between 1921 to the 1970s. The Squamish River Watershed Society (SRWS) led the MRR with the main objective of increasing rearing habitats for juvenile coho salmon by restoring mainstem flow to remnant historical floodplain channels. Reconnection of the floodplain was conducted via installation on water intakes and culverts, and by constructing new off-channel habitats. The project was initiated in 2005 and completed in 2012, however, until now, had not been evaluated for its effectiveness (E.Tobe 2021, personal communication.).

## 1.2. Study objectives

The main goal of this study was to assess the extent of occurrence of juvenile coho salmon, and to identify habitat features associated with their presence within the MRR study area. Three objectives were identified to achieve the goal of the study:

**Objective 1*:*** Determine the extent of juvenile coho salmon occupancy within the Mamquam River Reunion study area.

**Objective 2:** Evaluate each accessible culvert within the MRR study area for fish passage characteristics to determine the location of potential migration barriers.

**Objective 3*:*** Evaluate habitat quality and relationships with juvenile coho metrics to prioritize restoration and protection from future development.

Occurrence of juveniles was assessed by conducting presence/non-detection surveys. A null occupancy model was then used to estimate probability of occupancy of coho at two scales, and detection of coho over the summer. Habitat assessments were conducted to identify habitat relationships with juvenile coho metrics of presence, abundance, and growth. Habitat parameters focused on water quality, physical channel parameters, escape cover type and abundance. Culverts throughout the study area were also identified and assessed to identify potential barriers to juvenile coho migration.

This study was aimed at providing recommendations of priority areas to focus restoration efforts, and habitat protection, from future development within this heavily urbanized area of Squamish.

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# Chapter 2. Background

## 2.1. Introduction

The MRR study area is located on the historical floodplain of the Mamquam River, within the city of Squamish, British Columbia (BC), Canada, in the unceded and traditional territory of the Squamish First Nations (Sḵwx̱wú7mesh Úxwumixw). The Mamquam River has a drainage area of approximately 360 km$^2$ (Friele & Clague 2002), and it originates from November Lake, of the Garibaldi Ranges in the Coastal Mountains of BC (Google Earth Pro, 2022). The river flows northwest for approximately 7 km until its confluence with the Skookum River. The Skookum River drains the Mamquam Lake, and the Mamquam Icefields, and flows approximately 14 km southwest until its confluence with the Mamquam River. The Mamquam River flows west for approximately 13 km until its confluence with the Squamish River (Figure 2.1).

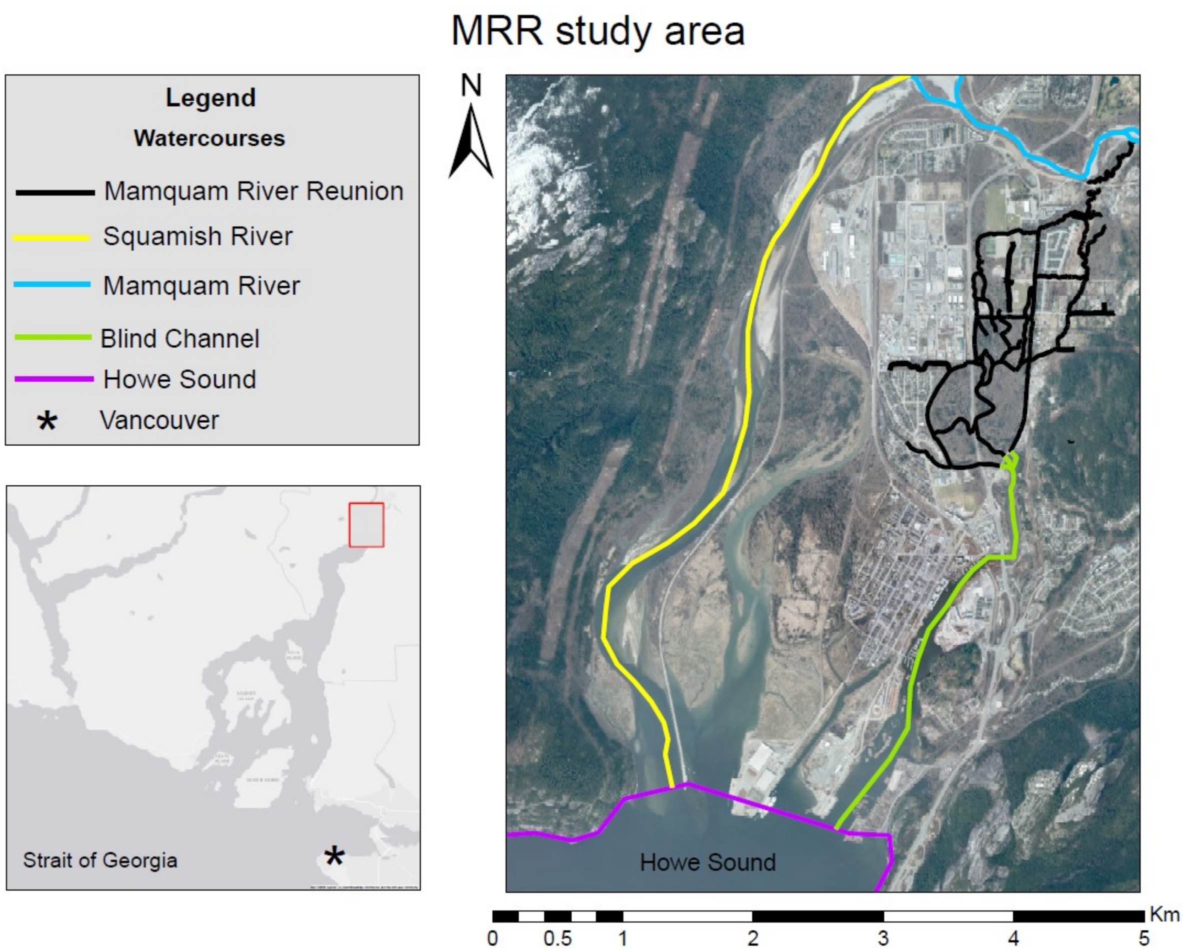


**Figure 2.1.Location of the MRR study area in relation to Howe Sound and Vancouver, BC, Canada.**

Before 1921 the Mamquam River flowed through the town of Squamish, to drain directly into Howe Sound. In 1921, an extreme flood event occurred and caused a major shift in the channel. The channel planform migrated west by approximately 2 km. Instead of flowing through Squamish, the river converged with the Squamish River. As result of the flooding and the widespread damage, the dike at Centennial Way was constructed to protect Squamish from future flooding. The construction of the dike resulted in the disconnection of the river to its historical floodplain. It also prevented the Mamquam River from accessing its historical outflow to Howe Sound (Figure 2.2).

In 2005, a water intake was installed at the confluence of the Mamquam River with the Squamish River. This location is considered the northernmost extent of the MRR. From here, the water flows into a constructed spawning channel then flows south until it reaches a second intake structure that bypasses the dike at Centennial Way. Waterflow is then distributed to the rest of the channel network via a series of culverts south to the outlet at Howe Sound (Figure 2.2).

Culverts were installed to connect channels within the network at road crossings and to maintain connectivity. There are three culverts connecting the MRR channels with the Blind Channel: (1): West Britannia slough, (2) East Britannia slough, and (3) East Logger's Lane Creek (Figure 2.3). These represent the three pathways for juvenile salmonids to access Howe Sound after freshwater rearing. The Blind Channel eventually drains into Howe Sound, where salmonids historically have entered the Mamquam River to spawn and rear alongside multiple anadromous and marine species (Hoos & Voos 1975).

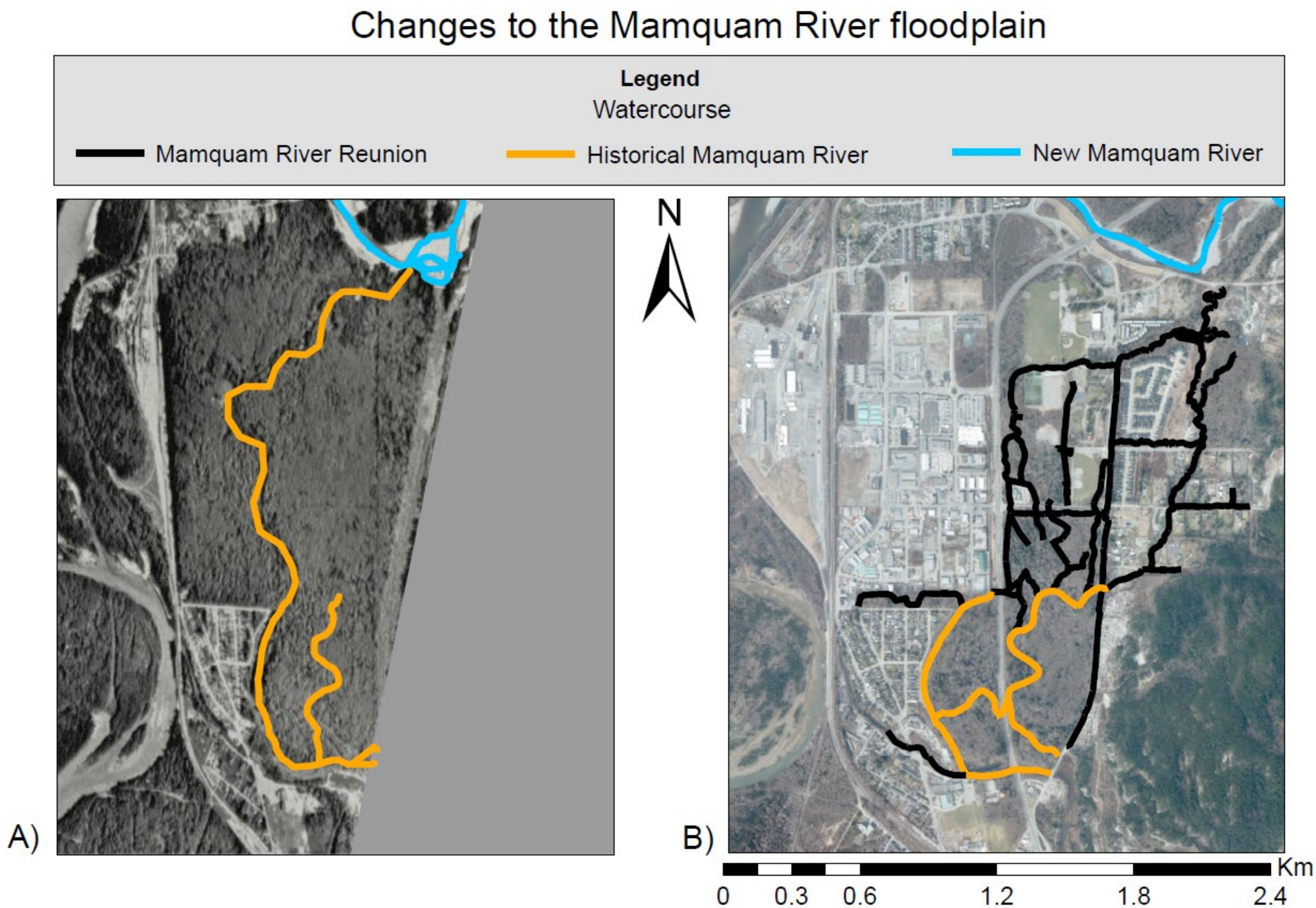


**Figure 2.2. A). A 1963 aerial photograph of the Mamquam River. Image shows the historical path of the Mamquam River and the new course due West after the construction the dike at Centennial Way. (Bell 1975). B). Image of the MMR and the historical Mamquam River channels (Squamish Open Data Portal 2021).**

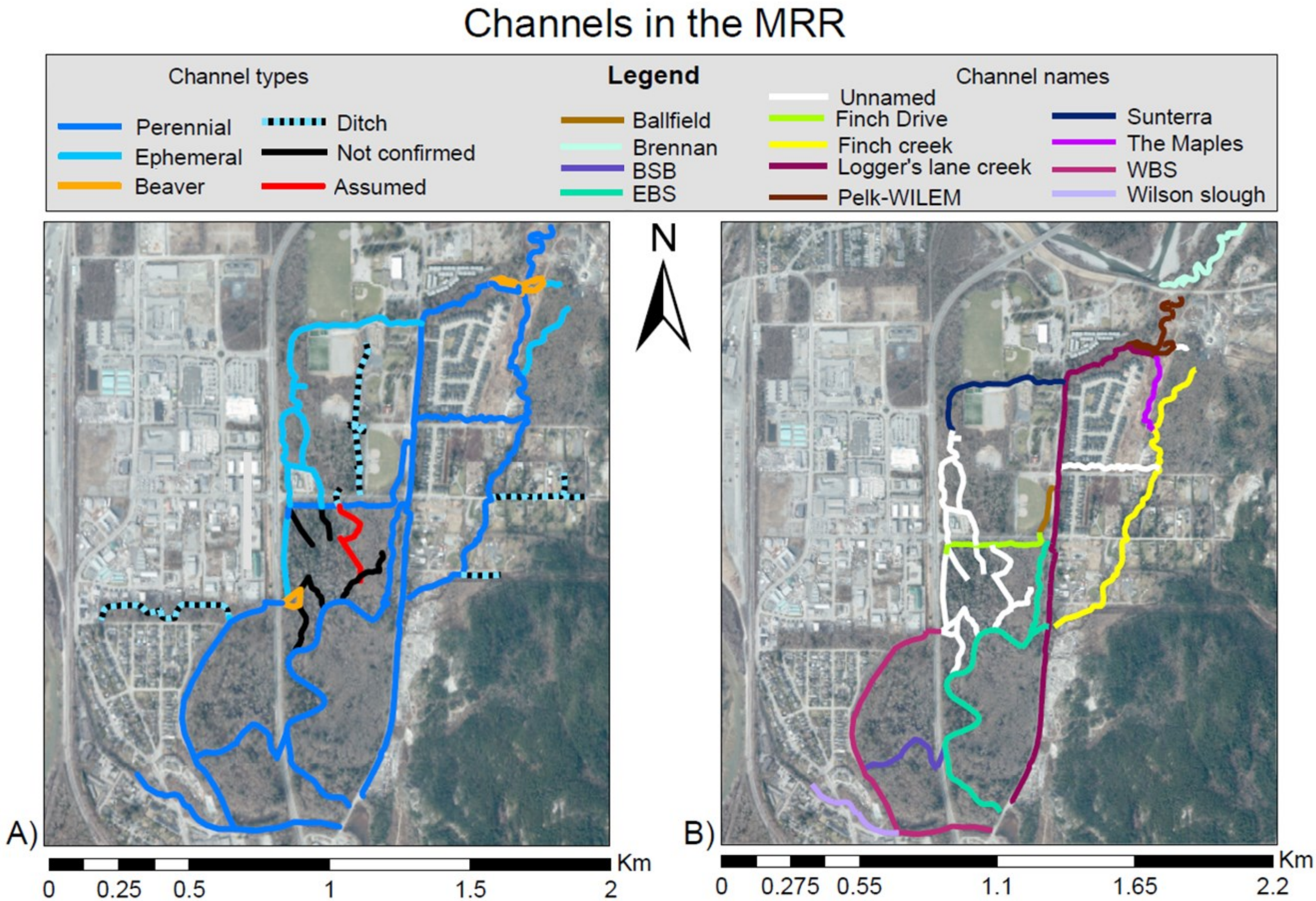


**Figure 2.3. A) Channel types represented in the MRR. B) Names of channels that were identified through available maps and reports. "BSB" stands for Britannia sough by-pass channel. "EBS" stands for East Britannia slough, and "WBS" stands for West Britannia slough.**

## 2.2. Geology and glaciation

The Mamquam River basin has a complex geologic, volcanic, and glacial history. The dominant bedrock within the Mamquam River basin is intrusive igneous rock from post-glacial volcanic activity within the Garibaldi Ranges (Holland 1976; Friele & Clague 2002a; Friele & Clague 2002b). The lower reaches of the Mamquam River, including the MRR, were under the Squamish valley glacier during the Younger Dryas glaciation period until about 11,900 cal year (calibrated years) ago (Friele & Clague 2002a; Friele & Clague 2002a). The upper reaches of the Mamquam River were in turn covered by ice originating from the Garibaldi Ranges, that created a lake once it retreated because of the presence of an ice dam (Friele & Clague 2002b). Deposition of material at the base of this ice dam formed terraces of sand, silt, and gravel after the retreat of the ice dam. Infilling of the valley below occurred from the unstable surrounding hillslopes that deposited material in the valley bottom. (Friele & Clague 2002a; Friele & Clague 2002b). The gravel and fine sediment deposited into the Mamquam River and floodplain that originates from the glacial history of the area, created optimal spawning and rearing habitat for salmonids.

## 2.3. Climate and hydrology

The city of Squamish is located within the Western Coastal Hemlock biogeoclimatic zone. Its maritime location promotes warm, dry summers, and mild, wet winters (Pojar et al. 1991; Kerr Wood Leidal 2017). The hydrological regime in the Mamquam River basin is mixed glacial and pluvial dominated (Wade et al. 2001; Kerr Wood Leidal 2017). Snow and glacial ice melt from the Mamquam Icefields during the summer, keeping water temperatures in streams cool as atmospheric temperatures increase (Lisi et al. 2015). Melting glacial ice also causes peak flow discharges to occur during the late summer months (May to August) (Wade et al. 2001). Glacier melt during the summer months is important for salmonids as it prevents thermal stress by providing cool water temperatures and mitigates low flow periods (Arismendi et al. 2013). The Coastal mountains are characterized by high periods of precipitation and streamflow during winter months (Eaton & Moore 2007). A secondary peak flow discharge occurs between October and January that is associated with heavy rainfall events common in the region (Bell 1975; Wade et al. 2001; Government of Canada 2020). These peak flows can be detrimental to freshwater rearing salmon if low velocity refugia are inaccessible or limited (Mantua et al. 2010).

Because the Mamquam River outlets to Howe Sound, which outlets to the Georgia Straight and Pacific Ocean, the lower 2 km of the watershed are influenced by daily tidal forces. Tidal forces from Howe Sound also impacts the Squamish River up to 2 km upstream the confluence with the Mamquam River (Squamish Estuary Management Plan 1982).

## 2.4. Salmonid populations

Historically, the Squamish River watershed supported all species of Pacific salmon: Pink (*Oncorhynchus gorbuscha),* chum (*Oncorhynchus keta),* coho (*Oncorhynchus kisutch),* sockeye (*Oncorhynchus tshawytscha),* and Chinook *(Oncorhynchus mykiss*). In addition, it supported over 60 other freshwater and marine species with food and cultural roles for the Squamish Nations (Golder 2005). All species of Pacific salmon were in decline in the 1990s, because of fisheries exploitation, low marine survival, and freshwater habitat degradation from urban development (DFO 2002). For instance, between the 1950s and 1990s, coho escapement decreased from 8,000 to >1,000 adults in the Mamquam River (Golder 2005).

Current data on spawning adult coho presence in the MMR study area is limited to count surveys conducted by a community stewardship group, the Squamish Streamkeepers. Since 2018, only 4 coho and 7 chum spawning adults were observed in up to two channels within the MMR. Upstream of the MRR, in the spawning channel north of the dike at Centennial Way, 228 coho, 420 chum, and 107 pink salmon have been observed since 2018.From 2007 to 2009, juvenile coho salmon presence was evaluated using Gee traps. These data showed that coho parr and fry were present throughout the entire MMR (Foy & Gidora 2008; Foy & Gidora 2008). However no other fish surveys have either been conducted, or their data have not been made available.

## 2.5. Off-channel habitats and salmonids

Rivers with connected floodplains spill over their banks and fill new or existing off-channel habitats during high discharge events. Off-channel habitats may be seasonally or permanently available. Pacific salmon species that have a freshwater residence life stage utilize off-channel habitats to rear before ocean migration. These species are coho, Chinook, and chum salmon. Coho salmon can migrate as far as 30 km downstream in search of off-channel habitats in the floodplain after emergence (Peterson 1982). Coho salmon remain in freshwater rearing habitats up to 3 years before smolting and migrating to estuaries (Sandercock 1991). Emergent and sub-yearling coho may migrate between freshwater and estuary channels as an alternative rearing strategy (Koski 2009; Craig et al 2014, Jones et al. 2014). In the Pacific northwest, smolt migration usually occurs after one year of rearing in freshwater off-channels, with migration to the ocean peaking in May (Armstrong & Argue 1977).

Natural and constructed off-channel habitats provide unique environmental conditions that promote the growth and survival of juvenile salmonids (Morley et al. 2005). Floodplain connectivity allows juvenile salmonids to exploit both the mainstem and off-channels to maximize food opportunities and prevent thermal stress (Armstrong & Schindler 2013; Hunstman & Falke 2019). Water temperatures are often higher in off-channels than in the mainstem, because of shallower depths and low velocities (Poole & Berman 2000). Primary productivity can increase in warm temperatures and provide an abundance of macroinvertebrate supply for juvenile salmonids (Sommer et al. 2001). However, accessibility to cold water temperature is important to juvenile coho abundance because of negative effects of thermal stress (Morley et al. 2005). When water temperature is too high and beyond their preferred range, juvenile coho may migrate out of warm off-channels in search of thermal refugia (Sutton & Soto, 2012). Overall, the connectivity between mainstem and off-channels within the floodplain can be directly linked to increased juvenile rearing capacity (Bond et al. 2019).

Overwintering survival of juvenile coho has been positively correlated with overall habitat quality and fish size after summer rearing (Brakensiek & Hankin 2007; Pess et al. 2011). Because of this, habitat condition during the summer is an important determinant of smolt production as it is a period of increased mortality. This occurs because juvenile salmonids can experience slower growth during summer rearing as a result of reduced food consumption associated with predator avoidance behaviors (Orrock et al. 2013). Suitable food consumption is important for the increased metabolic demands on the fish when stream temperatures increase (Armstrong &

Schindler 2013). However, metabolic demands can be offset by a high abundance and diversity of macroinvertebrates (Lusardi et al. 2020). Overall, the production of smolts and the recruitment of adult coho relate to the conditions experienced during the summer and overwintering rearing stage (Lawson et al. 2004).

Changes in stream discharge drive the re-distribution of juvenile salmonids from summer into overwintering seasonal habitats. Migration into overwintering habitats is timed with peak flows in the fall, as salmonids select deeper and more complex habitats to shelter from high flows (Peterson 1982; Nickelson 1992). Juvenile coho can be observed in stream-type off-channel habitats during both summer and overwintering periods, although larger parr can selectively occupy comparatively lower velocity off-channel ponds (Rosenfeld 2005; Rosenfeld et al. 2008; Bradley et al. 2016).

Instream cover have well-established positive correlations with juvenile salmonid survival (Smokorowski & Pratt 2007). Instream cover includes any structure instream or above-stream that: (1) Creates hydraulic conditions that are energetically favourable for resting and foraging, (2) increases hiding spaces for predator-protection, and (3) used for avoidance of intraspecific competition (Allouche 2002; Orrock 2013). During high flow conditions, instream structures such large woody debris (LWD) and instream vegetation, can create areas of low velocities. Instream cover is particularly important during low flow conditions (Penaluna et al. 2021). Deeper water provides shading and depth provides protection from avian predation (Steinmetz et al., 2003; Penaluna et al. 2016). A lack of escape cover can also increase local intraspecific competition and prompt the emigration of juvenile salmon from streams (Penaluna et al. 2021).

Indices of riparian condition and canopy cover are commonly assessed to evaluate instream habitat complexity (Sheer & Steel 2006). Canopy cover and overbank vegetation plays a role in predator protection through direct stream shading (Penaluna et al. 2016). Riparian vegetation is also associated with reducing both light penetration and stream temperature (Warren et al. 2013).

# Chapter 3. Methodology

## 3.1. Sampling unit selection

The watercourses within the study site were first identified in a desktop analysis using maps and shapefiles obtained from the SRWS and the City of Squamish (District of Squamish 2021). The location of each watercourse was then confirmed in the field between the months of April and May, 2021. Watercourses were divided into individual channels. Channels were established mainly at channel convergences, and at clear changes in channel morphology or vegetation. Channels were then categorized as continuous, seasonal, or beaver-influenced. Continuous channels were defined as channels with yearly flowing water. Seasonal channels were defined as natural or constructed intermittent tributaries but excluded ditches. Beaver-influenced channels were defined as areas with ponded or marsh-like water that had been created by the construction of a beaver dam(s).

A channel segment from every channel type within almost all identified channels were sampled. The starting location for each segment was determined by first using ArcGIS to determine the length of each segment. Then a random number between 0 and the length of the segment (from upstream to downstream) was selected using a pseudo-random number generator in R. The number represented the GPS coordinate corresponding to the distance within the length of the segment. This GPS coordinate marked the starting location of an approximate 100 m section of the segment to be sampled. Two GPS coordinates were selected at 50 m intervals downstream from the starting location. This 100 m section of each segment is referred from here on as the sampling unit. Deviations from the random starting location occurred where the location was not accessible, the channel was dry, or where discrepancies between the mapped location and the actual watercourse. A total of 22 sampling units throughout the study area were initially selected for analysis. However, only 20 sampling units were successfully sampled. This is because the remaining two sampling units were seasonal-type channels that were dry for most of the duration of this study, and thus removed from the study.

Sampling units were divided into 10 regions with similar characteristics for comparative analyses (Figure 3.1). Selection of each region was done by utilizing existing ecosystem type polygons obtained from Durand and Polly (2016) to group sampling units. Ecosystem type polygons corresponded to areas of similar vegetation characteristics and surficial geology (see Appendix A). Additional considerations for region designation included obvious changes in channel and vegetation characteristics and density of culverts. For instance, significant changes in channel width and canopy cover mark the boundaries between region 1 and 4, and culvert density differences mark the boundaries between region 7, 8 and 10.

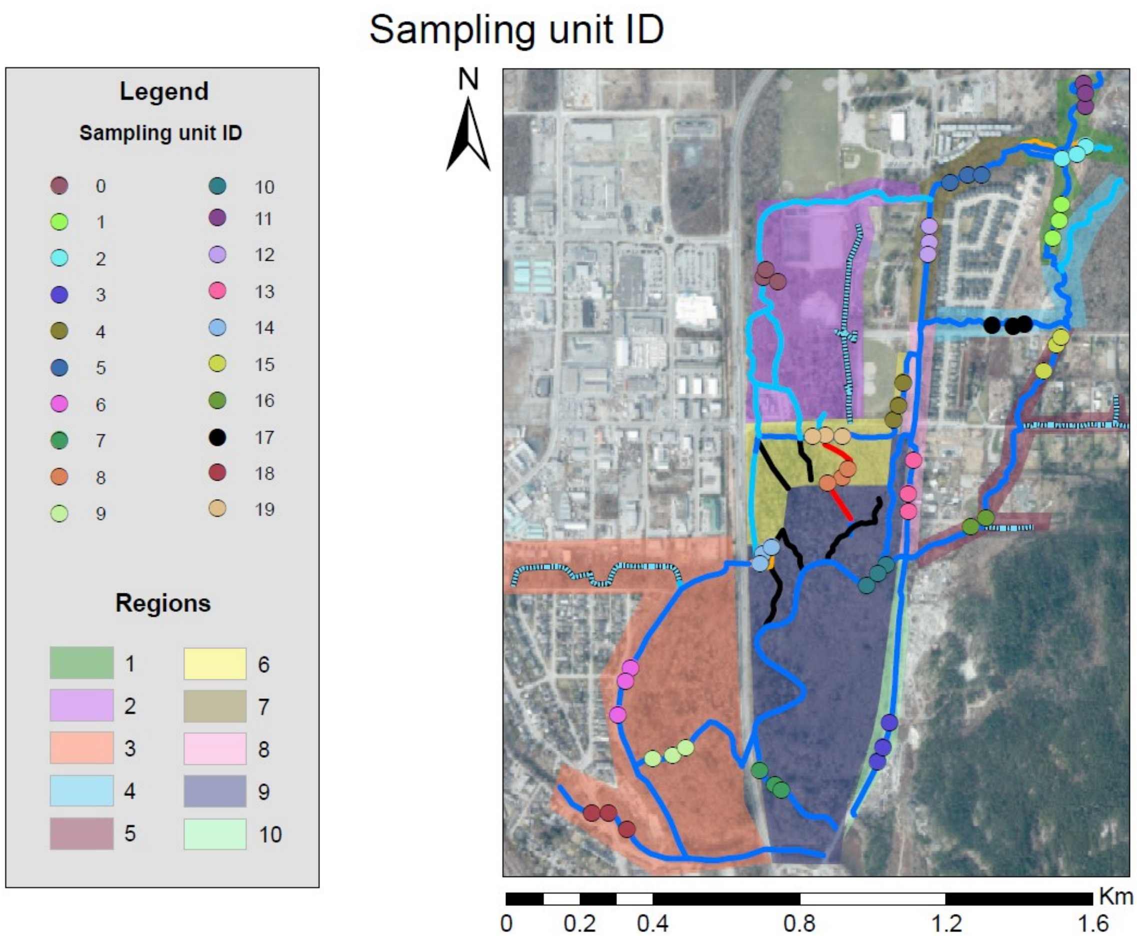


**Figure 3.1. Sampling units that were sampled and their location within the 10 regions of the MRR. Each colored circle represents trap-sites within each sampling unit.**

## 3.2. Juvenile coho trapping

Gee traps used were used for all presence/non-detection surveys. Each Gee trap was made of standard galvanized steel with 1/8” mesh, baited with salmon-flavoured cat food inserted onto a ball fashioned out of stockings. The bait was positioned to hang from the middle of the trap when placed in water.

Presence/non-detection surveys were conducted by placing three Gee traps within each sampling unit. The first trap was selected as close as possible to the selected GPS coordinate. The other traps were positioned downstream following the location of each 50 m spacing in a slow-velocity area deep enough to submerge all, or most of the trap (Sethi & Benolkin 2013). The traps were placed either upstream of downstream of the first trap when channel features were not appropriate for channel selection (e.g., shallow water depth, or safety issues), or there were deviations from the starting point. In these cases, the 50 m spacing was maintained by selecting a GPS coordinate located 50 m apart from the preceding trap on a geo-referenced map in Avenza map.

Each sampling unit was sampled three separate times (sampling periods) between June and August, 2021: June $10^{th}$ to$18^{th}$, July $12^{th}$ to $20^{th}$, and August $18^{th}$ to $27^{th}$. The order of sampling units visited was randomized to minimize detection bias. Exceptions were made for tidally influenced sites and difficult access sites, which were sampled earliest in the day. This was done to ensure trap placement was at a low, but rising tide, and for the safety of the sampling crew. Traps were set between 16:00-20:00 then removed the next morning between 07:30-10:30. This allowed for three sampling units to be sampled on most days, and an approximate 15 hours of soak time. Additional sampling occurred during October $27^{th}$ and December $15^{th}$ at select ephemeral and seasonal sampling units.

Salmonids (Pacific salmon species and trout species) were identified to the species level and non-salmonids to the family level. Counts were obtained for all species. Juvenile coho salmon CPUE was calculated at each trap-site during each of the sampling periods. In addition, the fork lengths of only juvenile coho salmon were measured to the nearest millimeter and recorded. Three juvenile coho metrics were used to assessed relationships with habitat parameters: CPUE, log-odds of presence, and fish length.

## 3.3. Habitat assessment

Habitat quality was assessed for three components: (1) water quality, (2) physical parameters (i.e., velocity and depth), and (3) estimates of canopy cover and escape cover. Visual estimates of cover for the distance between trap sites were completed at each sampling unit. This was an approximate distance of 15 m. This area extended downstream from the first (upstream-most) trap-site to the middle trap, both upstream and downstream of the middle trap, and then upstream from the last (downstream-most) trap-site. Photographs of the channel at each trap-site were taken then analyzed digitally to assess dominant escape cover type and abundance, as well as canopy cover %. This allowed in-field observations to be confirmed or corrected, when appropriate.

### 3.1.1. Water quality

Spot measurements of stream temperature (°C), dissolved oxygen (mg/L), and turbidity (FNU) were collected once at each trap site during each sampling period. Temperature and DO concentration were measured at each trap-site using a YSI meter. Turbidity was measured by analysing a water sample from each trap-site using a turbidimeter.

Water quality samples were collected at each trap-site. Measurements were collected once during each sampling period from June 10$^{th}$ to 16$^{th}$, July 12$^{th}$ to 19$^{th}$, and August 18$^{th}$ to 26$^{th}$, 2021. Water quality parameters included stream temperature (°C), dissolved oxygen concentration (mg/L), and turbidity (FNU). Mean velocity and mean depth measurements were taken from one transect (three measurements at ¼, ½, and ¾ wetted width) at each trap-site in August.

All sampling units except for three were sampled nine times throughout the summer for water quality. Sampling unit ID 1 was sampled six times because measurements during the June sampling period were not obtained. Sampling unit ID 16 was also sampled six times because only two trap-sites were sampled for each sampling period. Seasonal sampling unit ID 0 was sampled five times for water quality because a trap-site was dry in July, and all trap-sites were dry in August.

Each sampling unit was sampled three times during the summer for mean velocity and depth. Only perennial and beaver-type sampling units were evaluated for these parameters because transects were measured when all seasonal sampling units were dry. A summary of each water quality parameter and mean velocity and depth measurement data for each sampling unit are found in Appendix B and C. A summary of significant mean ranges of each parameter is found in Table 4.2.

Spot measurements of stream temperature provide limited information because it only provides temperature measurements at specific points in time. Therefore, maximum weekly average temperature (MWAT) and maximum weekly maximum temperature (MWMT) were also obtained. These are commonly used indices of temperature thresholds for salmonids (Welsh et al. 2001; Lusardi et al. 2020). MWAT and MWMT represent, respectively, the highest mean, and highest maximum temperature averaged over a period of 7-days. A limited number of data loggers were available for this study, and as such, MWAT and MMWT could not be measured at all sampling units. Therefore, spot measurements and data logger measurements were used to assess stream temperature

One Hobo Tidbit temperature logger was installed for the duration of the July and August sampling periods. Depending on the region, the data logger was placed in the same location as one of the sampling units, or another more representative location. The unit was submerged at substrate level in shallow segments (<1 m in depth), or mid-depth in deeper segments (>1 m in depth). The unit was set to log stream temperature every hour to record the average, maximum and minimum temperature every 24 hours. The MWMT and MAWT during each sampling period was calculated for each sampling unit.

Eleven data loggers were installed to record daily water temperatures in the MRR during July 15 to 18$^{th}$ and August 20 to 26$^{th}$, 2021(see Appendix D). One data logger was installed in a seasonal-type channel during only the July sampling period (data logger ID "Two"), and one data logger was installed at a perennial-type channel during the sampling period (data logger ID "Orange dam").

### 3.1.2. Physical channel measurements

Channel depth and velocity were collected at a transect across each trap site. Depth and velocity were measured at three equally distanced points along the wetted width (at ¼, ½, and ¾). Velocity was measured using a Hach portable velocity meter at a depth of 40 % from the bottom. The presence of gravel or boulders at each trap-site was also noted.

### 3.1.3. Canopy cover and escape cover

The canopy cover % and abundance of instream escape cover was assessed by type at each trap-site. Canopy cover was defined as the percent of vegetation covering the sky and estimated by taking a picture of the canopy at each trap-site and using the iOS application %Cover (Public Interest Enterprises) to calculate the cover percentage. Canopy cover was assigned into four abundance categories using Fish Habitat Assessment Protocol (FHAP): 100-71%, 70-41%, 40-21%, and ≤ 20% cover (Johnston & Slaney 1996).

The main escape cover types assessed in this study were: LWD, instream vegetation, and overbank vegetation (Johnston & Stanley 1996). LWD was defined as pieces with diameter >10 cm and longer than 1 m (Rosenfeld & Huato 2003). Rootwads or tree trunks >10 cm but shorter than 1 m in length were also identified as LWD. Instream vegetation are macrophytes located within the wetted width of the channel. Overbank vegetation was defined as vegetation overhanging <1 m above the stream channel.

The different types of escape cover present at each trap-site were identified visually and assigned into four abundance categories using the FHAP (Johnson & Stanley 1996). The abundance categories are based on the proportion between the stream section and the area occupied by the cover type, and included: None (<3 %), trace (3-5 %), moderate (5-20 %), and abundant (≥20 %). The dominant escape cover type at each trap-site was identified based on the assigned category. The average dominant escape cover type was determined for each sampling unit. Multiple dominant escape cover types were identified when two or more escape cover types were assigned the highest abundance category. A “mixed” dominant cover refers to equally dominant instream vegetation, overbank vegetation, and LWD. “Low LWD” refers to equally dominant overbank and instream vegetation. “Low vegetation” refers to equally dominant LWD and overbank vegetation. Finally, “low overbank” refers to equally dominant LWD and instream vegetation. Photographs were taken at each trap-site for post-field analysis of escape cover.

Overall escape cover complexity was also estimated. An abundance score (from 0 to 9) was calculated for all trap-sites and then averaged for each sampling unit (from a total of two or three scores depending on the sampling unit). This represented the average sum of all escape cover types present. For the purposes of this study, a score of 9 indicates that all escape cover types within a trap-site are in the “abundant” category. A score of 6 indicates that all escape cover types are in the “moderate” category. Finally, a score of 3 indicates that all escape cover types are in the “trace” category. Scores lower than 3 indicate that one or more escape cover type was not present in the trap-site. For each sampling unit, an abundance score between 0 and 4 represents that, on average, most or all of the escape cover types were present at a “trace” abundance category. A score between 4 and 7 represents that most or all of the escape cover types were present at a “moderate” abundance category. Finally, a score between 7 and 9 represents that most of the escape cover types were present at an “abundant” category.

## 3.1. Culvert assessment

An inventory of the location of culverts within fish habitat (i.e., excluding culverts within ditches) was done by utilizing available maps and a culvert GIS layer (Squamish Open Data 2021). Ground-truthing surveys were conducted in the spring between April and May. Measurements of culvert parameters outlined in BC Provincial culvert guidelines (BC Ministry of Environment 2011) were taken for each identified culvert. Culvert measurements were taken during the June and July sampling period and consisted of culvert length, the ratio of stream bankfull width to culvert width (stream with/culvert width ratio), slope (%), outfall drop height, and the ratio between the substrate embeddedness within the culvert and the width of the culvert (culvert embeddedness %). Each parameter was given a score based on measurement thresholds provided in the guidelines (see Appendix E). Culverts were designated as being likely passable, a potential barrier, or a barrier based on the sum of these scores (BC Ministry of Environment 2011).

# Chapter 4. Results

## 4.1. Statistical analyses

Generalized linear mixed-effects models (GLMM) were used in R (R Core Team 2020) with the "glmmTMB" package (Brooks et al. 2017) to characterize water quality and CPUE measurements within each sampling unit. These analyses were selected to better fit deviations from normality and to account for pseudo replication arising from the stratified sampling design of the study leading to repeated measures. Sampling unit ID was used as a fixed effect, and sampling period month was used as a random effect in the GLMM (Table 4.1). The models were run using trap-site observations per sampling unit for each water quality parameter and coho CPUE. The GLMM with random effects was compared to a null generalized linear model (GLM). The null GLM model did not include sampling period in the model. Physical parameters (mean depth and mean velocity) were characterized for each sampling unit using GLMs with the "glmmTMB" package (Brooks et al. 2017). This is because replication occurred only at the spatial scale. Comparisons between the GLMMs and GLMs was done using log-likelihood tests using the "lmtest" package (Zeileis & Hothorn, 2002). The model with the log-likelihood value closest to zero (absolute value) was determined to be the best fit model.

When fixed effect variables are categorical (i.e., sampling unit ID), regression models such as GLMMs and GLMs calculate slopes and significance (p-value <0.05) for each level. Three different reference levels were selected to compare the slopes of the remaining levels against: (1) The sampling unit ID with the highest mean of the response variable, (2) the sampling unit ID with a mid-value mean of the response variable, and (3) the sampling unit ID with the lowest mean of the response variable. The p-values and sign of the slope coefficients were then used to identify significant ranges of means between the individual sampling units. Each sampling unit ID was then assigned a range category for each response variable based on the results of the highest, mid-value, and lowest mean reference models.

**Table 4.1.Summary of competing models compared by log likelihood test to select the best fit model. Models were used to fit water quality and CPUE as response variables to characterize significantly different ranges of each response variable.**

| Response variable | Family | Log-likelihood test | Response variable~ Sampling unit ID+ (1\|Sampling period) | Response variable~ Sampling unit ID |
|---|---|---|---|---|
| Stream temperature | Normal | *df* | 22 | 21 |
| | | *logLik* | -275.6* | -317.9 |
| | | *p-value* | $2.2\text{x}10^{-16}$ | |
| Dissolved oxygen | Normal | *df* | 22 | 21 |
| | | *logLik* | -185.6* | -191.9 |
| | | *p-value* | 0.0004 | |
| Turbidity | Normal | *df* | 22 | 21 |
| | | *logLik* | -694.7* | -721.4 |
| | | *p-value* | $2.93\text{x}10^{-13}$ | |
| CPUE | Poisson | *df* | 20 | 19 |
| | | *logLik* | -312.4* | -347.3 |
| | | *p-value* | $2.2\text{x}10^{-16}$ | |

* Represents selected best fit model

## 4.2. Occupancy model

A single-season occupancy model was used to estimate juvenile coho probability of presence at sampling units and at trap-sites. This accounts for observation error (sampling error) in detection. Deviations from true occupancy at a site are due to imperfect detection, that is, when non-detections are not equivalent to absence because of sampling error (Mackenzie et al. 2017). The basic occupancy model can be explained by the statements:

$$Zi{\sim}Bernoulli\ (\psi) \tag{1}$$

$$yi|zi{\sim}Bernouli(p * zi) \tag{2}$$

Equation one explains that the true occupancy state of a species at a site ($zi$) is represented by the first statement, which describes that true species occupancy ($\psi$) is drawn from a Bernoulli distribution (there are only two possible states, present or absent). The result of presence/non-detection surveys can be represented in detection histories, where 0 represents non-detection, and 1 represents detections. Equation 2 represents that the observed occupancy state at the same site ($yi|zi$) is affected by the probability of detection ($p * zi$). Occupancy models use the detection history of sampled sites as occupancy observations and compare it to the assumed Bernoulli distribution to estimate probability of occupancy and detection (Mackenzie et al. 2017).

Simple single-season occupancy models estimate the probability of occupancy at a sampling unit ($\psi i$) and the probability of detection ($pi *$) under the assumption of imperfect detection. Multi-scale models are a modification to standard single-season models that allow estimation of occupancy at the larger and small-scale, by including estimation of a third parameter. This parameter is the probability of small-scale occupancy ($\theta$) at spatial subunits given that the species is present within a site (Nichols et al. 2008). In this case, repeated surveys are done at two levels. Primary survey occasions (e.g., spatial replication at each trap-site) serves to estimate occupancy parameters, and secondary survey occasions (i.e., temporal replication at each sampling month) to estimate detection parameters (Pavlacky et al. 2012). An estimate of small-scale occupancy accounts for a species' temporary movement between subunits that would influence detection probability at the larger site.

The effect of habitat and survey-specific covariates on both occupancy and detection probabilities can then be modelled by linear regression with "logit-link" functions (Mackenzie et al. 2017). Linear regressions are calculated to produce estimates of occupancies and detection probably. The intercept of these linear equations represents the estimate of occupancy or detection probability. The slopes represent the effect that each covariate included in the model has on the estimates of occupancy or detection probability

## 4.3. Coho extent of occurrence and passage limitations

### 4.3.1. Coho extent of occurrence

A total of 20 sampling units were evaluated between June and August 2021. Coho salmon were detected in 17 out of 20 sampling units (Figure 4.1), and within every region except region 2. A total of 394 coho salmon were captured, with fork length ranging in size from 22 to 89 mm.

Coho non-detections occurred in one wetted seasonal-type sampling unit and in two perennial-type sampling units. Coho salmon were trapped in all primary and secondary survey occasions (at every trap-site in every sampling period) in only one sampling unit (ID 16). However, only two trap-sites were set in this sampling unit instead of three trap-sites. Coho non-detection occurred in the perennial sampling unit IDs 10 and 14, and in the seasonal sampling unit ID 0. Additional sampling occurred in sampling units with coho non-detection to observe potential seasonal use. This occurred in sampling unit IDs 0 during October 27$^{th}$, and in sampling unit IDs 14 during December 15$^{th}$, 2021. Sampling in the seasonal channels in region 1 and region 4 also occurred during October 27$^{th}$, 2021. These represent sampling units originally chosen for sampling during the summer sampling period but were dried during the summer. Juvenile coho were only detected in the seasonal channel in region 4 during the additional sampling periods.

**4.3.2. Culvert assessments**

A total of 34 culverts were identified within the MRR project area. Four culverts were not assessed because of accessibility issues. Assessments (both complete and incomplete) were conducted for 30 of the 34 culverts (Figure 4.2). Three of the 30 culverts (culvert IDs 14, 15, and 30) had incomplete assessments because of visibility or accessibility issues for at least one parameter. However, these culverts were evaluated in the study.

Of the 30 culvert assessments, 19 culverts were scored as a passable, 10 as potential barriers, and one as a barrier. Every culvert assessed had a high stream with/culvert width ratio (over 1.6). A total of 24 culverts had low or zero substrate embeddedness leading to a low culvert embeddedness % (substrate embeddedness depth between 0 and <20 % of culvert width). Slope measurements for all culverts were low and ranged between 0.1 and 2.6 %. One culvert was scored as a barrier because of an outfall drop height of 0.35 m.

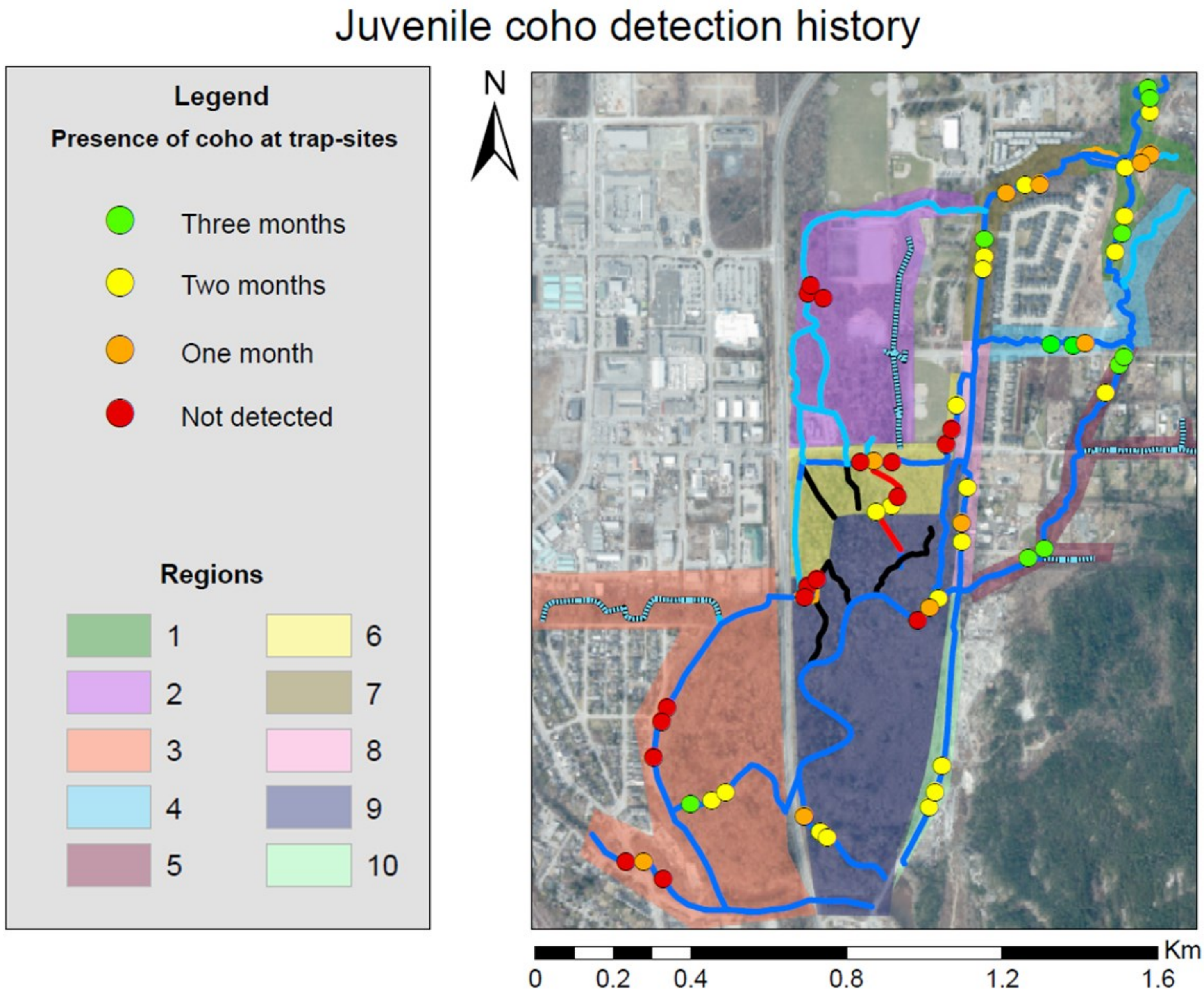


**Figure 4.1. Occupancy history of sampling units in the MRR study area during June, July, and August 2021. Each circle represents a trap-site within the sampling unit, and the color represents the detection frequency over the total sampling period.**

All culverts with incomplete assessments were scored as potential barriers except for culvert ID 15, which was scored as passable. The outflow of Culvert ID 18 was not visible, but the stream width/culvert width ratio was assumed to be high (over 1.6). In addition, the substrate embeddedness/culvert width ratio was assumed low (under 20% of culvert width). The slope of culvert ID 30 could not be measured because of the length of the culvert and its location across a highway. The culvert was scored as a potential barrier because of an accumulation of debris in the inflow side of the culvert. This creates an area of turbulence on the inflow side of the culvert, as well as potentially low flow in the culvert.

Coho salmon were not detected in sampling unit ID 0. However, there were no culverts that scored as barriers or potential barriers directly upstream or downstream of the sampling unit. Because of this, coho non-detection was not associated with culvert passage limitation. Two other sampling units with coho non-detections (sampling unit ID 6 and 14) had at least one culvert that scored as a potential barrier or barrier directly upstream or downstream the sampling unit that was. Non-detections for these sampling units were at least partially attributed to fish passage limitation. Potential fish passage barriers were identified upstream and/or downstream of 10 other sampling units (Appendix F). Coho were detected in these 10 sampling units, which suggests that only seasonal upstream fish passage limitation may occur in these sampling units.

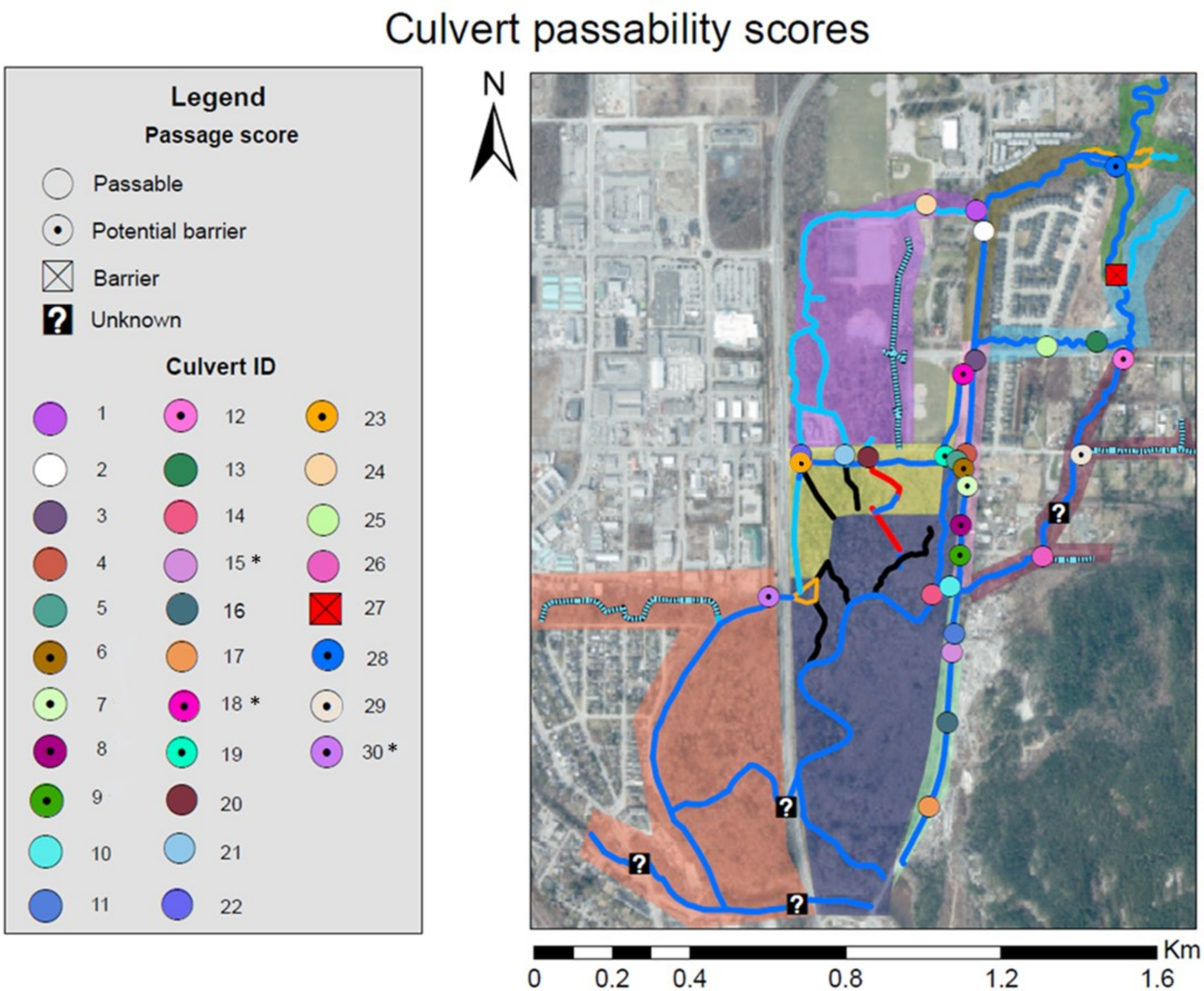


**Figure 4.2. Fish passage scores of culverts in the MRR. Scoring was based on BC Provincial guidelines (British Columbia Ministry of the Environment 2011). Culverts in ditches and stormwater culverts were not included in this study, and not shown in this map.**

*Represents culverts with incomplete assessments. Assumptions about the passage score were made.

## 4.4. Characterization of habitat quality

### 4.4.1. Water quality and channel measurements

**Stream temperature**

Spot measurements of stream temperature at trap-sites ranged between recorded values of 7.5 and 21.1 °C. Mean stream temperature values at sampling units ranged between 11.5 and 17.8 °C.

A GLMM was used to group sampling units with statistically similar mean stream temperatures. Spot measurements were used for analysis. The best fit model used sampling period as a random effect. The model results indicate that sampling unit ID had a significant effect on mean temperature, and it is a significant parameter to evaluate within the MRR. All sampling units had a significantly negative slope when the highest reference level was used (sampling unit ID 14, mean= 17.8 °C). Sampling unit IDs 5, 6, 10, 13, and 15 (mean range from 11.9 to 12.9 °C) had non-significant slopes when the lowest reference level was used (sampling unit ID 11, mean= 11.5 °C). The slopes for all other sampling units were significantly positive. Sampling unit IDs 11, 5, and 15 (mean range from 11.5 to 12.1 °C) had a significantly negative slope when the medium reference level was used (sampling unit ID 8, mean= 13.3 °C). However, sampling unit IDs 0, 14, 2, 7, 9, and 18 (mean range between 14.5 and 17.8 °C) had a significantly positive slope when the medium reference level was used (sampling unit ID 8, mean= 13.3 °C). This suggests that there are three significantly different ranges of mean stream temperatures based on spot measurements within the MRR: 17.8-14.5 °C (high), 14.4-12.2 °C (medium), and 12.1-11.5 °C (low).

Most sampling units had a low range of mean stream temperature. The lowest stream temperature range was between 11.4 and 13.2 °C. These were recorded in sampling unit IDs 3-6, and 10-15. The medium stream temperature range was between 13.3 and 15.4 °C, recorded in sampling unit IDs 1-2, 7-9, 16-17, and 19. The highest temperature range based on average spot temperatures was between 15.5 and 17.8 °C, located in sampling unit ID 0, 14, and 18.

**Table 4.2. Summary of water quality parameters and mean velocity and depth data for each sampling unit in the MRR. Stream temperature ranges are as follows: 17.8-14.5 °C (high), 14.4-12.2 °C (medium), and 12.1-11.5 °C (low). Dissolved oxygen concentration ranges are as follows: 11.3-10.4 mg/L (highest), 10.5-9.1 mg/L (high), 9.0-6.2 mg/L (medium), and 6.1-2.0 mg/L (low). Turbidity ranges are as follows: ≥56.4 FNU (high), and ≤56.3 FNU (low). Mean depth ranges are as follows: >36 cm (high), 35-27 cm (medium), <27 cm (low). Mean velocity ranges are as follows: 17.0-12.4 cm/sec (high), 12.3-6.0 cm/sec (medium), 6-0.9 cm/sec (low).**

| Sampling unit ID | Stream temperature (°C) | Dissolved oxygen (mg/L) | Turbidity (FNU) | Depth (m) | Velocity (m/sec) |
|---|---|---|---|---|---|
| 0 | High | Low | Low | High* | Low* |
| 1 | Medium | Highest | Low | High* | Low* |
| 2 | High | Highest | Low | Medium | Low |
| 3 | Medium | High | Low | High | Low |
| 4 | Medium | High | Low | High* | Low* |
| 5 | Low | Highest | Low | Low | High |
| 6 | Medium | Low | Low | High* | Low* |
| 7 | High | Medium | Low | Low | Low |
| 8 | Medium | High | Low | Low | Medium |
| 9 | High | Medium | Low | High* | Low* |
| 10 | Medium | High | Low | Medium | Medium |
| 11 | Low | Highest | High | High | Low |
| 12 | Medium | High | Low | High | Medium |
| 13 | Medium | High | Low | Low | High |
| 14 | High | Low | Low | High | Low |
| 15 | Low | High | Low | Medium | High |
| 16 | Medium | High | Low | High* | Low* |
| 17 | Medium | High | High | High* | Low* |
| 18 | High | Medium | Low | Medium | Low |
| 19 | Medium | Medium | Low | High | Low |

* Represents sampling units where data was determined using visual estimate.

The MAWT and MWMT for the summer ranged from 12.3 to 16.9 °C, and from 13.6 to 17.2 °C, respectively (Table 4.3).   Maximum daily temperature ranged from 13.9 to 17.6 °C, and mean daily temperature ranged between 11.5 and 17.5 °C. Maximum daily temperature ranged from 14.5 to 20.4 °C, and mean daily temperature ranged between 11.5 and 17.5 °C.

**Table 4.3. MAWT and MWMT for data loggers installed at representative sampling units in each region of the MRR. Data organized by weekly mean temperature (from lowest to highest).**

| Equivalent sampling unit ID | Data logger ID | Region | MAWT (°C) | SD | MWMT (°C) | SD | Both months? |
|---|---|---|---|---|---|---|---|
| 0 | Two | 2 | 16.9 | 0.6 | 17.2 | 0.4 | N |
| 1 | Five (Pink) | 1 | 13.4 | 0.7 | 14.5 | 0.9 | Y |
| 3 | Seven | 10 | 14.1 | 0.8 | 14.7 | 0.8 | Y |
| 6 | Orange | 3 | 13.5 | 0.3 | 13.7 | 0.3 | Y |
| 7 | Twelve | 9 | 15 | 0.8 | 15.8 | 0.9 | Y |
| 9 | Orangedam | 3 | 14.9 | 0.5 | 15.3 | 0.5 | N |
| 10 | Eighteen | 9 | 13.6 | 0.6 | 14.4 | 0.6 | Y |
| 12 | Twenty | 7 | 12.3 | 0.6 | 13.6 | 0.8 | Y |
| 15 | Twenty-two | 5 | 13.6 | 0.8 | 14.6 | 0.8 | Y |
| 17 | Eighty-eight | 4 | 13.8 | 0.8 | 14.9 | 1 | Y |
| 19 | Ninety | 6 | 13.8 | 0.7 | 14.5 | 0.7 | Y |

**Dissolved oxygen concentration**

Dissolved oxygen (DO) concentrations at trap-sites ranged between 1.0 and 12.7 mg/L. Mean DO concentrations at sampling units ranged between 2.0 and 11.3 mg/L.

A GLMM was used to group sampling units with statistically similar mean DO concentrations in the MRR. The best fit model used sampling period as a random effect. The model results indicate that sampling unit ID had a significant effect on mean temperature, and it is a significant parameter for evaluating the MRR. The slopes of all sampling units except sampling unit ID 11 were significantly negative when the highest reference level was used (sampling unit ID 1, mean= 11.3 mg/L). The slope of all sampling units except sampling unit ID 0 were significantly positive when the lowest reference level was used (sampling unit ID 6, mean= 2.0 mg/L). The slopes of sampling unit IDs 1, 11, 2, and 5 (mean ranging from 10.4 and

11.3 mg/L) were significantly positive when the medium reference level was used (sampling unit ID 8, mean= 9.6 mg/L). However, the slopes of sampling units IDs 6-7, 14, 0, 9, 18, and 19 (mean ranging from 2.1 and 8.8 mg/L) were significantly negative. This suggests that there are three significantly different ranges of DO concentration within the MRR: 11.3-10.4 mg/L (high), 10.3-9.1 mg/L (medium), 9.0-2.1 mg/L (low). However, because of biological significance of DO lower than 6 mg/L, another range of DO concentration is indicated. Therefore, the MRR is better represented by four different ranges of significantly different mean DO concentrations: 11.3-10.4 mg/L (highest), 10.5-9.1 mg/L (high), 9.0-6.2 mg/L (medium), and 6.1-2.0 mg/L (low).

Most sampling units had a high mean DO concentration range. The DO concentration range "low" recorded in sampling unit IDs 0, 6, and 14. The DO concentration range "medium" was recorded in sampling unit IDs 7, 9, 18, and 19. The DO concentration range "high" was recorded in sampling unit IDs 3, 4, 8, 10, 12, 13, and 15 17. The DO concentration range "highest" was recorded in sampling unit IDs 1, 2, 5, and 11.

**Turbidity**

Turbidity (FNU) at trap-sites ranged from 0.4 to 165.3 FNU. Mean turbidity at sampling units ranged from 13.3 to 68.5 FNU.

The best fit model to group sampling units with statistically similar mean turbidity in the MRR was a GLMM with sampling period as a random effect. The model results indicate that sampling month and sampling unit ID had a significant effect on mean turbidity, and it is a significant parameter for evaluating the MRR. Turbidity in the MRR was higher in both July (slope= 24.5, p-value= $1.26\text{x}10^{-9}$) and August (slope= 9.8, p-value= 0.01). The slopes for all sampling units except for sampling unit ID 17 were significantly negative when the highest reference level was used (sampling unit ID 11, mean= 68.5 FNU). The slopes of sampling unit IDs 11, 5, 12, 13, 15, and 17 (mean range from 40 to 68.5 FNU) were significantly positive when the lowest reference level was used (sampling unit ID 19, mean= 13.3 FNU). The same result was obtained when the mid-value reference level was used (sampling unit ID 7, mean= 25.9 FNU). This suggests that there are two significantly different ranges of mean turbidity within the MRR: ≥56.4 FNU (high), and ≤56.3 FNU (low).

Most sampling units had a low range of mean turbidity. The highest mean turbidity range was recorded in only sampling unit IDs 11 and 17. These sampling units are located in region 1 and 4.

**Velocity**

The mean velocity (cm/sec) at trap-sites ranged from 0.7 to 26 cm/sec during the August sampling period. Mean velocity at sampling units ranged from 0.9 to 17.0 cm/sec. Maximum velocities at sampling units ranged from 1.1 to 28.6 cm/sec. Some trap-sites and sampling units were excluded when the velocity could not be measured because of safety and accessibility issues. These trap-sites and sampling units had low velocity as they had standing water on visual inspection.

A GLM was used to group sampling units with statistically similar mean velocities in the MRR. The model indicated that sampling unit ID had a significant effect on mean velocity, and it is a significant parameter for evaluating the MRR. The slopes of all sampling units except sampling unit IDs 13 and 15 (mean ranging from 15.0 to 17.0 cm/sec) were significantly negative when the highest reference level was used (sampling unit ID 5, mean= 17.0 cm/sec). The slopes of all sampling units except sampling unit IDs 2, 18, and 19 (mean ranging from 1 to 4.0 cm/sec) were significantly positive when the lowest reference level was used (sampling unit ID 17, mean= 0.9 cm/sec). The slopes of sampling unit IDs 5, 13, and 15 (mean ranging from 15 and 17.0 cm/sec) were significantly positive when the medium reference level was used (sampling unit ID 10, mean= 10 cm/sec). However, the slopes of sampling unit IDs 14, 2, 18, and 19 (mean ranging from 1.0 to 4.0 cm/sec) were significantly negative. This suggests that there are three significantly different ranges of velocities within the MRR: 17.0-12.4 cm/sec (high), 12.3-6.0 cm/sec (medium), 6-0.9 cm/sec (low).

Most sampling units had a medium range of mean velocity. The mean velocity range “high” was found in sampling unit IDs 5, 13, and 15. The “medium” mean velocity range was found in sampling unit IDs 8, 10, 12, and 14. The “low” mean velocity range was found in all other sampling units.

**Depth**

The depth (cm) at trap-sites (maximum and minimum from three measurements per transect) ranged from 8 to 121 cm. The depths at sampling units (the mean of three transects) ranged from 23 to 59 cm. Some trap-sites and sampling units were excluded when the depth could not be measured because of safety and accessibility. These trap-sites and sampling units had depths exceeding 1 m based on visual inspection.

A GLM was used to group sampling units with statistically similar mean depths in the MRR. The model indicated that sampling unit ID had a significant effect on mean depth, and it is a significant parameter for evaluating the MRR. The model was fitted with a Gaussian distribution with a “log” link because the data was not normally distributed. The slope of all sampling units except sampling unit IDs 19, 11, 12, and 14 (mean ranging from 37 to 43 cm) was significantly negative when the sampling unit with the highest mean depth was used as the reference (sampling unit ID 3, mean= 46 cm). The slope of all sampling units except sampling unit IDs 5, 7, 10, and 13 (mean ranging from 20 to 28 cm) were significantly positive when the smallest mean depth was used as the reference (sampling unit ID 8, mean= 18 cm). The slope of sampling unit ID 3 (mean= 47 cm) was significantly positive when the mid-value mean depth was used as the reference (sampling unit ID 2, mean= 36 cm). However, the slope of sampling unit IDs 8, 5, 7, and 13 (mean ranging from 18 to 25 cm) were significantly negative. This suggests that there are three significantly different ranges of depth within the MRR: >36 cm (high), 35-27 cm (medium), <27 cm (low).

Most sampling units had high mean depth ranges. The mean depth range “high” was found in sampling unit IDs 0, 1, 3, 4, 6, 9, 11, 12, 14, 16, 17, and 19. The mean depth range “medium” was found in sampling unit IDs 2, 10, 15, and 18. The mean depth range “low” was found in sampling unit IDs 5, 7, and 8.

### 4.4.2. Escape cover and canopy cover

The most common average dominant escape cover type in sampling units was instream vegetation (Table 4.4). Instream vegetation was the dominant escape cover in sampling unit ID 4, 12, 13, 14, 15, and 18. "Low vegetation" (a mix between overbank and LWD dominant cover) was the least common average dominant escape cover type. Only sampling unit ID 17 was assigned this category. A moderate abundance category (covering between 5 and 20 % of the channel) for the dominant escape cover type was the most common abundance category at trap-sites (Appendix G). Trace abundance category (covering less than 3 % of the channel) of the dominant escape cover type was the least common abundance category at trap-sites.

**Table 4.4.Dominant escape cover types at sampling units. The proportion of sampling units with each dominant cover type is also indicated.**

| Dominant escape cover type | Proportion of sampling units |
|---|---|
| Instream vegetation | 32% |
| Low LWD* | 21% |
| Overbank | 21% |
| Mixed* | 11% |
| LWD | 11% |
| Low vegetation* | 5% |

* Represent multiple dominant escape cover types.

The mean sampling unit abundance score was 4.3, which indicates most trap-sites at sampling units had at least one non-dominant escape cover type at a moderate abundance category. Sampling unit IDs 1,9, 11, 13, 16, 18, and 19 had average abundance scores below 4, and the least variation (complexity) of escape cover types. Sampling unit ID 14 and 8 had average abundance scores above 7, and the most complexity of escape cover types in the MRR. The remaining sampling unit had a moderate complexity of escape cover types (abundance scores between 4 and 7).

Most trap-sites had a canopy cover of >70 % (42 trap-sites) (Appendix H). The least common canopy cover category was between 0 and 20 % (2 trap-sites). Sampling unit IDs 1 and 2 had the most open canopies, with a canopy cover between ≤20 %. Sampling unit IDs 2, 4, 6, 7, 8, 11, 13, 14, 18, and 19 had the most closed canopy (over ≥70 %).

### 4.4.3. Influence of habitat quality on coho occupancy

The detection history of coho over the MRR was analyzed using the program PRESENCE (Mackenzie et al. 2017), to estimate occupancy probability and detection probability while accounting for imperfect detections.

Presence of coho was determined using a null occupancy model (covariates were not modelled). The null model revealed that sampling unit occupancy and trap-site occupancy was relatively high ($\psi$= 90%, CI= 66-98 %, and $\theta$= 85%, CI= 71-93 %). Similarly, a trap-site detection probability of 0.67 (67 %) was estimated when using Gee traps in the MRR. However, the overall detection during the June sampling period was lower than during the July and August sampling periods (p(June)= 37 %, p(July)= 84 %, p(August)= 79 %).

Estimation of probabilities of occupancy at each individual sampling unit was limited without the inclusion of covariates. However, while considering imperfect detection, the model estimated that sampling units where coho salmon were present had $\psi$= 100% (CI=1-1), and those with coho non-detections had $\psi$= 3-33 % (CI= 0.003-0.33).

Issues with model convergence and parameter estimations may have occurred because of high coho detectability and small sample size. Inclusion of covariates in the occupancy model to estimate the effect of habitat associations on detection and occupancy probabilities was not successful. This was because of issues with model convergence and parameter estimations potentially because of high coho detectability and small sample size (Rota et al. 2011). Logistic regression using GLMs with a “logit-link” function can be used to assess general relationships between habitat parameters and the presence of highly detectable species, such as juvenile coho (Rota et al. 2011). Relationships between habitat and juvenile coho abundance and length can also be assessed with GLMs (Bradley et al. 2016). Therefore, relationships between habitat parameters and juvenile coho metrics were assessed using GLMs. The best fit model was chosen log-likelihood tests (Table 4.5).

**Coho presence**

The log-odds of coho presence was fit with a binomial distribution GLM with a “logit-link” function. Samples with missing data from habitat parameters were removed. Coho presence was best explained by including the following parameters in the model: Canopy cover %, mean temperature, mean DO, mean turbidity, and sampling period. Similar to the estimates from the null occupancy model, the log-odds of presence was higher in July and August compared to that in June (slope= 2.49, p= 0.003, and estimate= 2.02, p= 0.01, respectively). The log-odds of presence increases by 0.28 per unit increase of mean DO concentration (p= 0.03). On the other hand, the log-odds of presence decreases by -1.62 per unit increase of canopy cover % (p= 0.008). Mean turbidity and temperature did not have a significant effect on log-odds of presence.

**Table 4.5.Summary of comparisons between the null model and the best fit model to assess relationships between habitat parameters and juvenile coho metrics. The models were compared by log likelihood tests, and the best fit models represent the model with the lowest absolute log likelihood value (closest to zero).**

| Response variable | Family | Null model (Response variable~ Sampling period) | | Best model | | |
|---|---|---|---|---|---|---|
| | | Log-likelihood test | | Model | Log-likelihood test | |
| Log-odds of presence | Binomial, "logit" link | *df* | 3 | *Presence~ Sampling period + Canopy + Temperature + Turbidity + DO* | *df* | 7 |
| | | *logLik* | -97.7 | | *logLik* | -82.9 |
| | | *p-value* | | | *p-value* | $5.91 \times 10^{-6}$ |
| Length | Normal | *df* | 4 | *Length~ Sampling period +Abundance score + Canopy* | *df* | 16 |
| | | *logLik* | -1276.9 | | *logLik* | -1207 |
| | | *p-value* | | | *p-value* | $2.2 \times 10^{-16}$ |
| CPUE (as counts with trap soaking time as an offset) | Poisson | *df* | 3 | *Counts~ Sampling period+ Canopy + Temperature + Dominant cover type, Offset= Log (Trap soaking time)* | *df* | 11 |
| | | *logLik* | -423.1 | | *logLik* | -375.4 |
| | | *p-value* | | | *p-value* | $2.2 \times 10^{-16}$ |

**Coho length**

The relationship between habitat parameters and coho length was assessed using a GLM. Samples with missing data from habitat parameters were removed. The relationship between habitat parameters and coho length was best explained by sampling period, escape cover abundance score, and canopy cover %. Fish length was higher in July and August compared to the June sampling period (slope= 14.1, p= $2 \times 10^{-16}$, and slope= 19.7 p= $2 \times 10^{-16}$, respectively). Fish length increases with average escape cover abundance scores (slope= 2.1, p= $8.48^{-12}$). Canopy cover also increases fish length (slope= 0.07, p= $4.32 \times 10^{-5}$).

**Coho catch per unit effort (CPUE)**

A GLMM with a Poisson distribution was used to group together sampling units with statistically similar mean CPUE. The best fit model included sampling period as a fixed effect. The model was run of count data with trap soaking time as an offset to obtain CPUE values. Observations with zero counts were removed to prevent zero-inflated data. The model results indicate that sampling unit ID had a significant effect on mean CPUE, and it is a significant parameter for evaluating the MRR. All sampling units except sampling unit IDs 4, 10, and 19 (mean range from 0.56 to 0.068 CPUE) had negative slopes when the lowest reference level was used (sampling unit ID 18, mean= 0.008 CPUE). All sampling units except for sampling unit ID 16 (mean range from 0.20 to 0.008 CPUE) had a negative slope when compared to the highest reference level (sampling unit ID 15, mean= 0.56 CPUE). Sampling unit IDs 15 and 16 (mean range from 0.56 to 0.41 CPUE) had positive slopes when compared to the medium reference level (sampling unit ID 11, mean= 0.14 CPUE). However, the slopes of sampling unit IDs 18 ,4, 10, 16 and 19 (mean range from 0.063 to 0.008 CPUE) were negative. This suggests four ranges of statistically significant mean CPUE in the MRR: 0.56-0.21 CPUE (high), 0.20-0.064 CPUE (medium), 0.063-0.008 CPUE (low), and none (Table 4.6).

The relationship between habitat parameters and coho CPUE was assessed using a Poisson distribution GLM. Samples with missing data from habitat parameters were removed. The relationship between habitat parameters and coho CPUE was best explained by sampling period, canopy cover %, mean temperature, and dominant escape cover type. Results indicate that mean CPUE decreases by 3.5 % per unit increase of canopy cover (exp(slope)= -3.45, $p = 8.43x^{-11}$). Mean CPUE also decreases by 1.1 % per unit increase of stream temperature (exp(slope)= -0.52, p= 0.001). Compared to mean CPUE in June, mean CPUE increases by 3.2 % and 2.4 % in July and August, respectively (exp(slope)= 3.16, $6.97 \times 10^{-7}$, and exp(slope)= $1.71 \times 10^{-5}$). This result is in line with the increased detection probability in July and August compared to the June sampling period.

**Table 4.6. Summary of significantly different CPUE ranges in the MRR. Mean counts, trap soaking time, and CPUE represent the average respective values at each sampling unit during the total summer sampling period. CPUE ranges are as followed: 0.56-0.21 CPUE (high), 0.20-0.064 CPUE (medium), 0.063-0.008 CPUE (low), and none.**

| Sampling unit ID | Mean counts | Mean trap soaking time (hours) | Mean CPUE | CPUE group |
|---|---|---|---|---|
| 0 | 0 | 14.92 | 0 | None |
| 1 | 2 | 14.79 | 0.160 | Medium |
| 2 | 2 | 15.06 | 0.110 | Medium |
| 3 | 2 | 14.68 | 0.140 | Medium |
| 4 | 1 | 14.44 | 0.039 | Low |
| 5 | 1 | 12.75 | 0.085 | Medium |
| 6 | 0 | 14.51 | 0 | None |
| 7 | 3 | 15.18 | 0.170 | Medium |
| 8 | 1 | 14.84 | 0.068 | Medium |
| 9 | 3 | 15.25 | 0.180 | Medium |
| 10 | 1 | 15.01 | 0.063 | Low |
| 11 | 2 | 14.93 | 0.140 | Medium |
| 12 | 3 | 14.04 | 0.190 | Medium |
| 13 | 2 | 15.18 | 0.100 | Medium |
| 14 | 0 | 14.65 | 0 | None |
| 15 | 8 | 14.59 | 0.560 | High |
| 16 | 6 | 13.82 | 0.410 | High |
| 17 | 4 | 14.48 | 0.200 | Medium |
| 18 | 0 | 14.59 | 0.008 | Low |
| 19 | 1 | 14.85 | 0.040 | Low |

Dominant escape cover type also had a significant effect on mean coho CPUE. Tukey contrasts were used for pairwise comparisons between each dominant escape cover type, using the "Multcomp" package (Hothorn et al. 2008). Only the significance between the main escape cover types (i.e., instream vegetation, overbank vegetation, LWD, and mixed) was assessed. This is because of the small sample size of the other multiple escape cover categories ("low vegetation" and "low LWD"). Results indicate that mean CPUE increases by 1.9 % when the dominant escape cover type is instream vegetation compared to mixed escape cover (exp(slope)= 1.90, p= 0.003). Mean CPUE also increases by 2.4 % when the dominant escape cover type is instream vegetation compared to overbank vegetation (exp(slope)= 2.36, p= 0.001).

# Chapter 5. Discussion

## 5.1. Water quality

### 5.1.1. Water temperature

Overall, stream temperatures are within acceptable thresholds for juvenile coho salmon. BC Water Quality Guidelines (Ministry of Environment) outline the optimal temperature for rearing juvenile coho is from 9 to 16 °C. Other sources suggest an average from 12 to 15 °C as optimal temperatures for rearing coho (Hicks 2000). In the PWN, MWMT and MAWT temperatures related to juvenile coho persistence in streams occur at 16 °C MWMT and 15 °C MAWT (Welsh et al. 2001; Richter & Kolmes, 2005; Lusardi et al. 2020). The density of juvenile coho decreases at MWMT above 20 °C (Ebersole et al. 2009). Mortalities are expected to occur when temperatures exceed from 23 to 25 °C for various stocks of juvenile salmon in BC (McGreer et al. 1991). The stream temperature of the MRR ranged from 7.3 to 21.1 °C at trap-sites, and from 11.4 to 17.8 °C at sampling units. Maximum weekly maximum temperature (MWMT) and maximum average weekly temperatures (MAWT) ranged from 13.6 to 17.2 °C and from 12.3 to 16.95 °C 11.5 to 17.5 °C, and from 14.5 to 20.4 °C, respectively.

Most sampling units with low and medium spot temperature ranges had MAWT and MWMT within healthy thresholds for juvenile coho rearing. The mean temperature obtained from spot measurements in sampling unit ID 0 (high temperature range) did not exceed the optimal temperature. However, the MAWT and MWMT measured by the data logger in the same location (data logger ID "Two") exceeded thresholds (16.9 and 17.2 °C, respectively). Mean temperature in sampling unit ID 7 (medium temperature range) did not exceed optimal temperatures. However, the MAWT and MWMT were marginally within thresholds as recorded by data logger ID "Twelve" (15 and 15.8 °C, respectively). Even though optimal temperatures were exceeded in sampling unit ID 14 and 18 (high temperature range) based on spot measurements, the MWMT and MAWT in these sampling units were not assessed.

In this study, 84 % of all sampling units had stream temperature within optimal thresholds and within survivable temperatures for rearing juvenile coho. The temperature in the seasonal sampling unit ID 0 exceeded persistence thresholds based on data logger measurements. However, more data is required to conclude whether sampling unit ID 14 and 18 exceed temperature thresholds for juvenile coho survival. Coho detections occurred in sampling unit ID 7, therefore more data is required to determine if thermal stress occurs in this sampling unit. One juvenile coho was also detected in sampling unit ID 18, however no detections occurred in neither sampling unit ID 14 nor 0. Sampling unit ID 0 is a seasonal channel, and sampling unit IDs 14 and 18 are ephemeral channels. These three sampling units are ponded channels with low-velocity water flow. The channel characteristics of these sampling units promote heat absorption from sunlight, heat retention, and reduced flushing with freshwater to cool water temperatures (Poole & Berman 2000). Sampling unit ID 14 and 18 are likely to exceed temperature thresholds in the summer, although more data is required to conclude temperature thresholds exceedance. Evidence to support the statement that there are warmer temperatures in ponded channels is found in the comparison of temperatures between data logger ID 11 and 3. Data logger ID 11 was installed in a ponded channel directly upstream a beaver dam. In contrast, data logger ID 3 was installed directly downstream of the beaver dam. Both MWMT and MAWT were over 1 °C higher downstream of the ponded channel. This suggests that high stream temperature is at least partially related to coho non-detection in sampling unit 14 and 0.

#### 5.1.2. Dissolved oxygen

Coho salmon were detected in all sampling units with mean DO concentrations from 6.1 mg/L to 11.3 mg/L. Non-detections occurred in sampling unit IDs 0, 14, and 6, that all had average DO concentrations between 2.0 and 6.1 mg/L These sampling units had DO concentrations below healthy levels for juvenile salmon. Low oxygen-related impairments of salmonids begin at DO concentrations as low as 6 mg/L (Herrman et al. 1962; Davis 1975) during which avoidance behaviors from coho occur (Whitmore et al. 1960). Acute mortalities occur when salmonids are chronically exposed to DO concentrations of approximately 2.0 mg/L (Herrman et al. 1962).

Sampling units with low DO concentrations were found in the West Britannia slough (WBS) within region 3, and immediately upstream of the WBS within region 6. The WBS is characterized by thick, orange-mats (Figure 5.1). These mats are made up of a complex of iron-oxidizing bacteria, referred to as flocs, and are present in freshwater bodies rich in Fe (II) (Emerson & Weiss 2004). Flocculants can be observed in slow moving environments when anoxic waterflows, such as groundwater, mixed with oxic waterflows (Emerson and Weiss 2004). Although flocculants can be natural component of watersheds, too much build-up can lead to negative consequences to the channel. Decreased oxygenation can occur when an overabundance of flocs obstructs the interchange of oxygen at the groundwater-surface water boundary (Koski & Herricks 1999), and at the air-surface water boundary. Primary productivity can also be impaired by flocs by direct or indirect effects on algae and macroinvertebrates (Kotalik & Cadmus, 2019). These conditions are aggravated in channels with limited waterflow, as in the case of the WBS.

**Figure 5.1. A) Characteristic flocculant mats covering the channels in the West Britannia slough (WBS). B) Image facing upstream(east) of the Britannia slough by-pass channel, adjacent to the WBS but lacking flocculant mats. Images taken on August 23rd, 2021.**

The poor water quality in the WBS was recognized even prior to the MRR project initiation in 2005 (Squamish River Watershed Society 2008). However, previous Gee trapping surveys in 2009 captured juvenile coho salmon in the WBS (see Appendix I). The lack of juvenile coho detection in this study suggests that water quality in the WBS has decreased since 2009. DO measurements from previous years are not available, therefore direct comparisons of changes is not possible.

Even though the WBS is characterized by low water quality, one sampling unit in region 6 had healthy DO concentration. Sampling unit ID 9 in the Britannia slough by-pass (BSB) channel had a mean DO concentration of 7.4 mg/L and is within the “medium” DO concentration range (mean DO concentration from 9.0 to 6.2 mg/L). A higher DO concentration in this channel can be attributed to the restoration work that occurred in 2008 (Squamish River Watershed Society 2008). Restoration included deepening of the channel, addition of LWD complexes, and planting of native vegetation. Waterflow to the BSB originates from a culvert that crosses the East Britannia slough within region 9. Sampling unit ID 7 is in proximity of this culvert within region 9, and it is also characterized by a medium DO concentration range (mean DO concentration of 8.6 mg/L). DO concentration in the BSB is appropriate because of good oxygenated waterflow originating from the East Britannia slough (EBS), and because of the deepening of the channel in 2008 that likely improved groundwater seepage. In comparison, the WBS receives poorly oxygenated waterflow from sampling unit ID 14 within region 6 that is also characterized by low average DO concentration (mean DO concentration of 6.1 mg/L).

Coho salmon were detected in the BSB channel. These fish are believed to have migrated downstream from the East Britannia slough in region 9. The channels in region 3 are connected to region 9 through a culvert crossing Highway 99 (not assessed in this study). Although DO concentration (and water temperature) in the BSB channel is within survivable ranges, it can be creating an ecological trap for juvenile coho salmon causing a decrease in long-term survivability (Robertson and Hutton 2006). This would occur if fish migrate downstream into the lower reaches of the WBS during the smoltification process. There is evidence that smolts have a lower tolerance for DO concentrations than pre-smolts (Elshout et al. 2013). Therefore, smolt mortalities may occur if DO concentration remains below 6 mg/L downstream of the BSB channel. There is likely low DO concentration downstream of the BSB channel because flocculants were also observed in this region. However, DO concentration was not assessed in this portion of the WBS.

#### 5.1.3. Turbidity

The highest turbidity range (>56.4 FNU) was found in sampling unit ID 11 and 17 (68.5 FNU and 56.4 FNU, respectively). Sampling unit ID 11 is in region 1 that is the closest to the intake on the dike at Centennial Way. The Mamquam River receives glacial meltwater during the summer months that is naturally turbid from the high concentrations of fine silts and clay from glacial meltwater (i.e., glacial flour). Turbidity was highest in these sampling units in July and August because of the inputs of glacial flour and during high-water flows from glacial meltwater (see Appendix J). Turbidity can be beneficial as it provides predator protection to juvenile salmonids (Gregory and Levings 1998). However, elevated turbidity can cause respiratory issues (Servizi and Martens 1991) and decreased growth because of reduced food consumption (Martin et al. 2019). A threshold of 55 NTU was found for a PNW coastal stream before growth limitation occurs in juvenile coho salmon (Martin et al. 2019). Emigration out of some channels to avoid suspended sediments can occur at turbidity levels as low as 11 NTU (Siegler et al. 1984). However, the negative effect of turbidity depends, in part, on the shape and size of the suspended particles. Large particles with high angularity can incur more stress on juvenile salmonids at smaller concentration than rounder particles (Lake & Hinch 1999), such as silts and clay.

Coho salmon in the highest turbidity range in the MRR (>56.4 FNU) are not likely to be experiencing negative impacts, because turbidity in this region is temporal and non-anthropogenic in nature. This is supported by the results of this study, where turbidity was not negatively associated with coho presence, length, or CPUE. Coho salmon were also consistently found in the sampling units with the highest turbidity (sampling unit ID 11 and 17). However, large depositions of silt and clay were observed mid-channel near the dike at Centennial Way, which may have potential indirect and negative consequences to coho salmon (see Appendix J). Suspended sediments that deposit in the channel bed can fill in the interstitial spaces of a gravel bed stream. If there is too much infilling, it can inhibit egg incubation (Greig and Carling 2005). One spawning chum salmon was observed in in sampling unit ID 1 during the August sampling period, however it is unclear whether the salmon was able to spawn. This observation suggest that spawning adults can access the intake on the dike at Centennial Way. However, spawning salmon accessing the MRR through the intake may either abandoned sampling unit ID 1 because of a lack of gravel, or spawn unsuccessfully. Therefore, more research is required to determine potential effects of turbidity on spawning and egg incubation success.

## 5.2. Connectivity and fish passage

Using the BC provincial guidelines (British Columbia Ministry of the Environment 2011) to assess culvert passage for fish, 63.3 % of culverts in the MRR scored as passable, 33.3 % scored as potential barriers and 3.3 % scored as impassable. The parameters with the greatest impact on fish passability in the MRR are culvert width and substrate embeddedness. This is a similar result to other culvert assessments completed throughout the province (Chestnut 2001). A small culvert width relative to the width of the channel will constrict flow passing through the culvert. This creates high-water velocities impassable for small fish (Baker & Votapka 1990). Low or zero substrate embeddedness can also increase water velocity through a culvert. Substrate found in embedded culverts creates substrate roughness that allow slow water velocity, creating refuges inside the culvert that benefit fish migrating upstream (Baker & Votapka 1992). Therefore, culverts with low or zero embeddedness lack these velocity refuges. However, measurements of water velocity within culverts and comparison with juvenile coho swimming speed thresholds should occur to confirm fish passage is restricted (Davis & Davis 2011).

Non-detections of juvenile coho salmon occurred both upstream and downstream of two culverts that were scored as potential barriers. Culvert ID 23 and 30 are located directly upstream of sampling unit ID 14 and sampling unit ID 6, respectively. Sampling unit ID 6 is located within the WBS while sampling unit ID 14 is located upstream in region 6. These sampling units are characterized by low DO concentrations. Non-detections may be occurring because of avoidance behaviors or mortalities of juvenile coho unable to escape the poor water quality in these sampling units.

Culvert ID 23 may also be creating another ecological trap ecological trap population sink, which would be occurring if upstream passage to better quality water is restricted. Juvenile coho salmon were detected upstream of sampling unit ID 14 and culvert ID 23, where DO concentrations were above avoidance and mortality thresholds. Because culvert ID 14 was scored as a potential barrier, it is possible that juvenile coho salmon that migrate downstream into sampling unit ID 14 are unable to return to the higher water quality conditions upstream of culvert ID 23. Therefore, juvenile coho salmon may migrate downstream of sampling unit ID 14 into the WBS through culvert ID 30 without the ability to relocate. Mortalities of juvenile coho salmon are likely to occur when fish are unable to escape poorly oxygenated waters, which may be the case in sampling unit ID 14 and the WBS.

## 5.3. Juvenile coho-habitat relationships

### 5.3.1. Escape cover

Three cover types were assessed qualitatively in the channels of the MRR: (1) Instream vegetation, (2) LWD, and (3) overbank vegetation. Instream vegetation was the escape cover type that was the most common and abundant in the sampling units (i.e., the dominant escape cover). Instream vegetation as the dominant escape cover type had a positive effect of coho CPUE compared to an all-mixed dominant cover and dominant overbank vegetation cover. This result suggests rearing juvenile coho salmon benefit from abundant instream vegetation in the summer months.

The most common instream vegetation type observed was the emergent macrophyte Skunk cabbage (*Lysichiton americanus*), and to a lesser extent unidentified species of Phragmites (species of grasses, and reeds). Macrophytes provide instream cover that can be used by fish to hide from predators (Vowles & Kemp 2019). In addition, macrophytes create instream conditions that support higher densities of macroinvertebrate taxa that supports fast growth of juvenile salmonids (Lusardi et al. 2018; Marsh et al. 2022). Emergent and submerged macrophytes can therefore be preferentially used for summer rearing by juvenile salmon than other instream cover types, such as LWD (Lusardi et al. 2018). However, macrophytes die off during the fall and winter months, which decrease the instream vegetation and the dietary benefits provided during the summer (Vowles and Kempt 2019; Marsh et al. 2022) (Figure 5.2). Because of the temporal nature of vegetation, channels where instream vegetation is the dominant escape cover provide limited escape cover for fish during fall and winter months. Sites where overbank vegetation is the dominant escape cover can similarly provide limited escape cover during the fall and winter months. Therefore, channels with more permanent escape cover instead of solely instream or overbank vegetation, such as LWD, provide better escape cover to overwintering juvenile salmon.

In this study, there was little to no association between rearing juvenile coho salmon and LWD. However, LWD is commonly associated with benefits to juvenile salmonids for escape cover (Hafs et al. 2014; Bair et al. 2019) and food (Eggert & Wallace 2007; Hasselquist et al. 2018). Positive relationships between LWD and salmonids have been related to the impacts of LWD on the size and frequency of pool formation (Gonzales et al. 2017). An assessment of pools in the MRR revealed a lack of pool habitat within the MRR. Detailed assessment of pool location and characteristics was limited because of water turbidity and fine, organic sediments in the channel bed. Therefore, there was no determined association between pools and LWD within the MRR. The effect of LWD on pool formation may be limited in small, low gradient channels like the floodplain channels of the MRR (Beechie & Sibley 1997; Rosenfeld & Huato 2003). Therefore, the associated benefits of LWD on salmonids because of their impact on pools are also limited in these channels (Bradley et al. 2016). The escape cover that LWD provides for predator protection is most important during the fall and winter, when temperatures drop and fish movement and speed decreases (Roni & Quinn, 2001; Giannico & Hinch 2003). Therefore, LWD is likely more significant as escape cover in colder water temperatures and in the fall and winter than in the warmer summer temperatures.

The overall abundance and complexity of escape cover had a positive relationship with fish length. A higher escape cover abundance score in this study represents an overall higher abundance of all main escape cover types. Sampling units with the highest abundance scores represent more complex habitats than those with lower abundance scores. In this study, sampling unit ID 8 and 14 had the most complex habitat in terms of escape cover availability (escape cover abundance scores of 8 and 7, respectively). Increased fish length in more complex habitats may be a result of juvenile coho competition. Larger individuals can outcompete smaller individuals out of more productive habitats that may promote faster growth (Van Leeuwen et al. 2011). Dominant juvenile coho may face higher mortalities than less dominant fish when predator threat is high, because of exposure risks of territorial behaviors (Martel 1996). Therefore, a higher abundance of escape cover may be counteracting this where predator presence in high in the MRR. However, the abundance and complexity of escape cover in the MRR was moderate to low, like other engineered off-channels (Morley et al. 2005).


A)


B)

**Figure 5.2. A) Instream and overbank vegetation observed during the summer. B) Instream and overbank vegetation decrease during the fall and winter. Picture taken on August 20th and December 12th, 2021, respectively.**

### 5.3.2. Canopy cover

Canopy cover in the MRR had statistical significance with juvenile coho metrics. A negative association was found between higher canopy cover and log-odds of juvenile coho presence and CPUE. This result suggests that juvenile coho either avoid areas with closed canopies, or that these areas support lower densities of juveniles than those with open canopies. Positive associations between juvenile salmonid abundance and a lower canopy cover have occurred in previous studies (Bradley et al. 2016). An open canopy provides increased light availability for photosynthesis, which can increase the abundance of macrophytes and macroinvertebrates and provide energetic benefits to juvenile coho salmon (Riley et al. 2009; McCormick & Harrison 2011). Primary productivity can be reduced from lower light penetration into the stream. However, canopy cover can provide the stream with shelter from thermal stress (Malcolm et al. 2004; Warren et al. 2013). A negative relationship between CPUE and higher stream temperature as well as canopy cover %, suggests that canopy cover does not have a significant effect on stream temperatures in the MRR. However, canopy cover % had a positive relationship with fish length in this study, which suggests potential growth benefits of canopy cover. Therefore, future research should be done to understand the effect of canopy cover on stream productivity and juvenile coho salmon growth.

### 5.3.3. Channel velocity and depth

Mean velocities of all sampling units are within acceptable velocities for juvenile coho salmon. Habitat suitability curves for juvenile coho indicate that velocities ranging from 1.5 cm/sec to 10.7 cm/sec are the most suitable (Suchanek et al. 1984). Similarly, in PNW streams habitats with velocities ranging from 3.0 to 12.0 cm/sec are preferentially selected based on average suitability curves (Beecher et al. 2002). Velocities above 24.6 cm/sec have significantly lower suitability for juvenile coho rearing (Suchanek et al. 1984). However unsuitable velocities have been found above a range from 42.7 cm/sec to 55.0 cm/sec (Suchanek et al. 1984; Beecher et al. 2002). The highest velocity recorded in the MRR at trap-sites was 26.0 cm/sec, and the highest mean velocity range recorded was between 12.3 and 17.0 cm/sec at sampling unit ID 5, 13, and 15.

Mean depths in the MRR are also within acceptable depths for juvenile coho salmon. Preferred depths by juvenile coho in the PNW have been shown to be between 46 and 120 cm (Beecher et al. 2002). The highest ranges of mean depth were found in sampling unit IDs 0, 1, 3, 4, 6, 8, 11, 12, 14, 16, and 17. Sampling unit IDs 0, 1, 4, 6, 9, 16, and 17, could not be measure in this study. However, these sampling units potentially reach depths higher than 46 cm in the summer. In contrast, sampling unit ID 5,7,8, and 13 have the lowest range of mean depths (<27 cm). Although juvenile coho can use a large range of depths (Shirvell 1994), preference decreases significantly at depths lower than 15 cm (Beecher et al. 2002).

Relationships between velocity and depth and coho metric were not assessed in this study because of missing measurements from multiple sampling units. However, there is evidence of benefits to high stream velocities to rearing juvenile coho. Increased access to higher velocity habitats becomes more accessible with fish size, because of increased swimming speed (Del Signore et al. 2016). Habitats with velocities above 10.0 cm/sec provide a source of drift macroinvertebrates that promote higher growth compared to lower velocity habitats (Nielsen 1992; Rosenfeld et al. 2005). When macroinvertebrate supply is highly available, juvenile coho also utilize higher velocity habitats than those accessed when food supply is low (Rosenfeld et al. 2005). Therefore, a high macroinvertebrate availability creates a positive feedback loop for juvenile salmonid growth. Possible interactions between stream velocity, depth, and fish length exists because larger, and older, juveniles can selectively occupy slow and deep channels (Rosenfeld et al. 2008; Bradley et al. 2016). Therefore, the connectivity between channels to maintain velocity and depth complexities is important and should be maintained (Rosenfeld et al. 2008).

# Chapter 6. Recommendations and Conclusions

The main objective of the MRR project was to increase rearing habitat quality and connectivity for juvenile coho salmon. This study confirmed that juvenile coho are indeed utilizing most regions of the MRR. Juvenile coho salmon were not detected in sampling units where water quality was poor and where connectivity issues were identified. These issues within the MRR can be addressed to improve rearing habitat for juvenile coho.

## 6.1. Recommendations

### 6.1.1. Fish passage

The study of fish passage barriers through culverts was limited in this study because of accessibility issues at some culverts. Therefore, complete assessment could not be completed for all culverts. In addition, velocity measurements were not obtained as part of the assessment which limits the conclusions about actual fish passage limitations. Culvert ID 23 and 30 are listed as potential barriers. Velocity data should be collected and paired up with additional presence/non-detection surveys to determine if juvenile coho salmon are able to pass through culvert IDs 23 and 30. If these culverts are deemed a barrier, they should be replaced with wider culverts that are embedded within the substrate.

There are three culverts (culvert ID 30, and two unnamed culverts) that are considered potential barriers, and where debris-clearing is recommended (Appendix K). Debris clearing will improve fish passage connectivity between regions, improving habitat accessibility. Culvert ID 30 should be cleared of woody debris on the inflow side of the culvert to reduce turbulence, passage obstacles, and improve waterflow downstream. An unnamed culvert in sampling unit ID 18, within the Wilson slough, was not assessed in this study. However, this culvert is also blocked by debris on the inflow side and is likely limiting fish passage and flushing of the Wilson Slough. Therefore, clearing of this culvert is also recommended. An additional unnamed culvert requiring clearing exists in region 1, which was also not assessed in this study. This culvert connects an overflow/seasonal channel next to sampling unit ID 2 in the PELK-WILEM channel. This culvert is blocked by debris from beaver activity. Although juvenile coho were not observed in the overflow/seasonal channel during the summer season or during the additional sampling in

October, three-spined sticklebacks (*Gasterosteus aculeatus*) were captured in this channel. This suggests that fish may be entering this channel through seasonal flooding and overflow of the PELK-WILEM channel into the overflow channel. Clearing of this culvert would provide access to additional overwintering habitat for juvenile coho. This would be beneficial as the overflow channel contains LWD and provides pond-like low velocity refuge. This action would also alleviate flooding of the PELK-WILEM channel onto the local walking trail.

In addition to culvert passability, beaver dams were identified as potential barriers as they were found to limit connectivity between streams within the MRR. A beaver dam within sampling unit ID 14 was determined to be a significant barrier to waterflow into a pre-existing channel (Figure 6.1). Although it was not possible to visually confirm, it is believed this channel flows into the higher water quality channels within region 9. Removal of the beaver dam would provide the following: (1) provide an escape route for stranded juvenile coho in sampling unit ID 14 into better habitat conditions, and (2) improve water quality within sampling unit ID 14 through increased flushing of streamflow. A field survey is recommended to confirm the connectivity of this channel to region 9.

Only one culvert out of the 30 culverts evaluated was scored as a barrier to fish passage based on culvert measurements. The only culvert scored as a barrier to fish passage was culvert ID 28 located in region 1. This culvert is a slide gate culvert that is used to control the amount of water flowing downstream from the PELK-WILEM channel. Although this culvert was score as a barrier to upstream passage, it may not be a limiting factor to juvenile coho. Upstream fish passage into region 1 is not likely a limiting factor to juvenile coho salmon because this region is characterized by low escape cover availability, particularly LWD that provides overwintering rearing habitats. Therefore, it is unlikely that seasonal upstream fish migration into region 1 occurs. Because of this, culvert replacement is not recommended. However, a large boulder is placed directly underneath the outflow of the culvert. This boulder should be removed to prevent physical damage to juvenile coho moving downstream through this culvert.

A) 

**Figure 6.1. A) Disconnected channel downstream of the beaver dam that likely connects to the channels within region 9. Images taken on August 25th, 2021.**

**6.1.2. Rearing habitat**

A water quality assessment of the lower reaches of the WBS is required to determine survival of smolts out-migrating from the BSB channel. The summer occupancy and survival of juvenile coho salmon in the downstream reach of the WBS should be assessed with additional presence/non-detection surveys. The results of these assessments should inform prioritization of restoration actions to decrease flocculants downstream of the BSB channel.

These could include increased waterflow, physical removal of flocculants, and planting of vegetation that uptakes iron from the water (Koski & Herricks 1999). Increased waterflow could be achieved by deepening of the channel to promote increased groundwater seepage and by promoting waterflow inputs from existing tributaries. The construction of new channels to divert water from BSB or from region 9 could also be an alternative to improve waterflow. Planting of certain plant species with rhizomes, such as Cattails and Bulrushes, has potential applications to decrease flocculants as they can retain Fe II within their roots (Karathanasis & Johnson 2003).

Although flocculants were not observed in sampling unit ID 0, flocculants were observed upstream in region 2 within the Sunterra channel. Waterflow in these channels was low throughout the summer, which may explain the drying up of sampling unit ID 0. Coho salmon were not detected here during the summer, nor during additional sampling in October and December. This additional sampling was an attempt to detect seasonal usage of juvenile coho in this channel. Overall, waterflow should be improved in this channel if increasing juvenile salmon habitat in this region is prioritized.

This study was able to establish general relationships between juvenile coho and habitat parameters. These relationships can be used to inform instream restoration to benefit juvenile salmonid rearing habitat. Amongst these is the significant positive relationship found between instream vegetation and juvenile coho CPUE, which suggests that this escape cover type is either preferred or can support greater densities of juvenile coho. However, because of the temporal nature of vegetation, the addition of LWD should be considered in sampling units dominated by instream vegetation or overbank vegetation. Research on macroinvertebrate supply in the MRR, particularly as it related to the effect of instream vegetation and canopy cover %, be considered to better understand juvenile coho habitat associations. This is particularly important under climate change considerations because the survivability of juvenile coho under increased temperatures can be offset by macroinvertebrate abundance (Lusardi et al. 2020).

Additional climate change considerations are needed in the MRR. Glacial retreat in BC decreases streamflow during the summer months (Stahl & Moore 2006). Therefore, sampling units with both low depths and high temperatures ranges should be monitored particularly during low flow periods.

#### 6.1.3. Protection from developments

Most active developments within the MRR are within region 1,4, and 5 (Figure 6.2). These regions also provide important rearing habitat for juvenile coho salmon as determined by CPUE. Both sampling units in region 5 (sampling unit IDs 15 and 16) had the highest CPUE range in the MRR. Sampling unit ID 17 is located directly upstream of region 5 and it had the next highest CPUE range. Protection from developments should also consider habitat connectivity between summer rearing and overwintering habitats. This is because the distance between seasonal habitats influences migration between them and utilization by juvenile salmon (Flitcroft et al. 2012). Sampling unit ID 17 provides overwintering habitat because of dominant LWD escape cover. On the other hand, the adjacent sampling unit IDs 15 and 16 provide summer rearing habitats because of dominant instream vegetation escape cover. Therefore, region 4 and 5, and the connectivity between them, should be protected from developments. In addition, protection of region 1 is important because juvenile coho salmon likely migrate from the dike at Centennial Way and continue a downstream migration into the channels in the MRR.

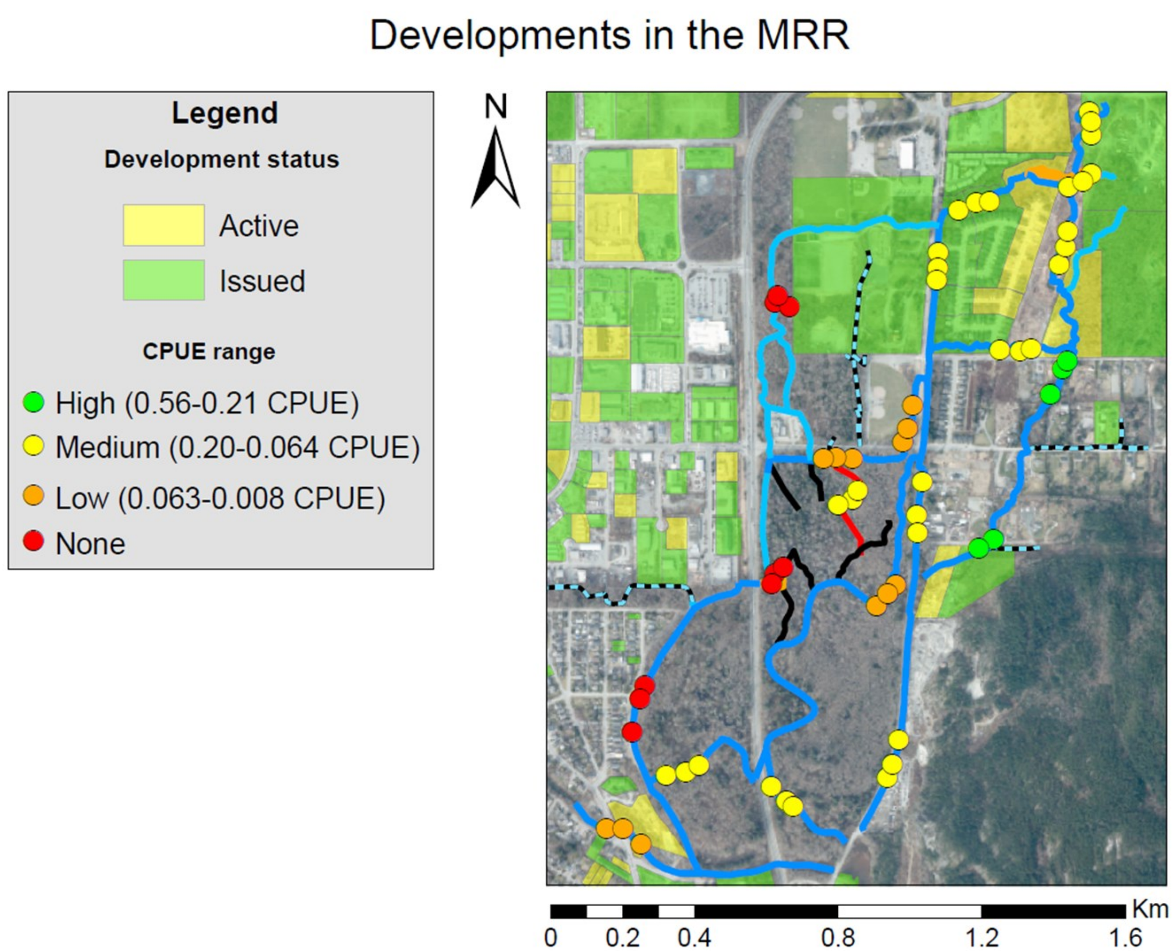


**Figure 6.2. Comparison of the mean CPUE ranges at sampling units and currently active and issued development permits in the MRR.**

## 6.2. Conclusion

This study assesses the success of the MRR project at providing rearing habitats for juvenile coho salmon. Results indicate that the MRR was successful at providing rearing habitats, and juvenile coho occupancy and detection probability were relatively high. Non-detections of juvenile coho were restricted to sampling units with low dissolved oxygen concentration, and where culverts are a potential fish passage limitation. These areas represent opportunities where restoration can be aimed to improve fish habitat. The relationships between juvenile coho metrics and habitat parameters can also be informative for future studies, and for the protection of important rearing areas for juvenile coho.

Key associations between juvenile coho and habitat parameters highlight the importance of food availability for rearing. Instream vegetation as the dominant escape cover was identified as positively associated with coho CPUE compared to other escape cover types. On the other hand, canopy cover was negatively associated with both CPUE and log-odds of coho presence. This suggest that factors influencing macroinvertebrate supply are important to habitat selection by rearing juvenile coho in the MRR. Overall abundance of escape cover was also associated with increased fish length, suggesting benefits of habitat complexity for fish growth and survival.

This study identified areas where restoration activities and research should be prioritized to maximize rearing habitat available for juvenile coho. A key priority is an assessment of the BSB. There is evidence that this area is a sink for out-migrating juvenile coho salmon because of poor water quality in the WBS. Presence/non-detection surveys and water quality measurements should be conducted downstream of the BSB to assess the survivability of coho smolts. A similar assessment should occur in the channel between culvert ID 23 and culvert ID 30. Restoration action in this channel and the downstream reach of the WBS should be considered to improve water flow and water quality. Additions of LWD structures in regions dominated by instream or overbank vegetation are also recommended to support overwintering rearing.

Protection of important fish habitat from current and future developments can also be guided by the results of this study. Sampling units and regions with higher mean CPUE should be protected from urban developments. In addition, the connectivity between channels dominated by instream vegetation and where LWD is available should be protected. This would ensure the proximity of escape cover suitable for both summer rearing and overwintering that can maximize juvenile coho survivability.

## Chapter 7. Appendices

### Appendix A. Terrestrial Ecosystem Mapping polygons.

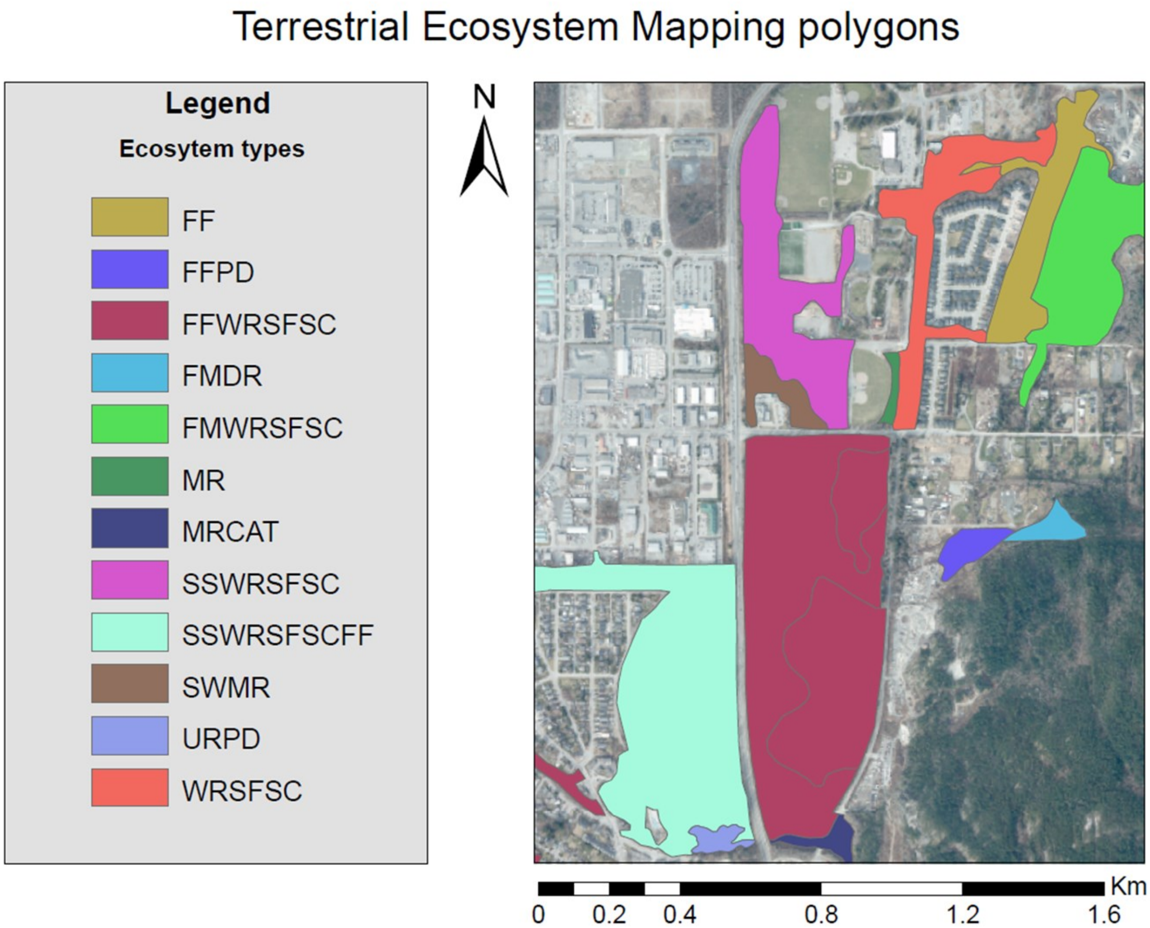


**Figure A.1. Polygons outlining ecosystem types within the study area from a Terrestrial Ecosystem Mapping (TEM) study for the District of Squamish (Durand Ecological LTD and POLAR Geoscience Ltd, 2016). Polygons represent areas with similar topology and soils (bioterrain polygons), and similar ecosystem types that are characteristic of the Western Coastal Hemlock biogeoclimatic zone. Ecosystem type categories were designated by compiling ecosystem type at three sites for each polygon (see Table A.1). TEM GIS layer and associated ecosystem type description obtained from the District of Squamish open data portal.**

**Table A.1. Ecosystem types associated with each polygon from the TEM GIS layer (Durand Ecological LTD and POLAR Goescience Ltd, 2016; District of Squamish 2021). Map codes representing each ecosystem type categories were designated by compiling ecosystem type at three sites for each polygon.**

| Ecosystem type | | | Map code |
|---|---|---|---|
| **Site 1** | **Site 2** | **Site 3** | |
| Flat moss | Western redcedar - sword fern - skunk cabbage swamp | | FMWRSFSC |
| Flat moss | HwCw - Deer fern | | FMDR |
| CW-Foamflower | PD | | FFPD |
| CW-Foamflower | Swamp-unclassified | | FF |
| SS-Salmonberry | Western redcedar - sword fern - skunk cabbage swamp | CW-Foamflower | SSWRSFSCFF |
| SS-Salmonberry | Western redcedar - sword fern - skunk cabbage swamp | | SSWRSFSC |
| Pond | Urban | | URPD |
| Marsh-unclassified | Cattail | CW-Foamflower | MRCAT |
| Marsh-unclassified | | | MR |
| Swamp-unclassified | Marsh-unclassified | | SWMR |
| Western redcedar - sword fern - skunk cabbage swamp | | CW-Foamflower | FFWRSFSC |
| Western redcedar - sword fern - skunk cabbage swamp | | | WRSFSC |

## Appendix B. Summary of mean water quality parameters.

**Table B.1.Summary of mean water quality parameters at sampling units in the MRR.**

| Sampling unit ID | Temperature (°C) | SD | DO (mg/L) | SD | Turbidity (FNU) | SD |
|---|---|---|---|---|---|---|
| 0 | 15.5 | 1.5 | 2.3 | 0.7 | 15.0 | 6.9 |
| 1 | 13.3 | 4.5 | 11.3 | 0.8 | 19.2 | 14.2 |
| 2 | 15.2 | 2.9 | 10.4 | 1.4 | 23.4 | 16.1 |
| 3 | 12.9 | 1.9 | 9.5 | 0.2 | 18.7 | 8.5 |
| 4 | 13.0 | 2.8 | 9.3 | 0.6 | 26.5 | 19.1 |
| 5 | 12.1 | 3.3 | 10.4 | 0.4 | 40.4 | 7.3 |
| 6 | 12.9 | 1.2 | 2.1 | 0.6 | 34.2 | 39.4 |
| 7 | 14.5 | 1.6 | 8.6 | 0.6 | 25.9 | 16.9 |
| 8 | 13.3 | 1.9 | 9.6 | 0.4 | 27.4 | 24.4 |
| 9 | 14.8 | 2.2 | 7.4 | 0.5 | 15.1 | 10.3 |
| 10 | 12.3 | 1.9 | 9.6 | 0.9 | 19.0 | 10.5 |
| 11 | 11.5 | 3.0 | 11.0 | 0.6 | 68.5 | 62.1 |
| 12 | 12.6 | 2.6 | 10.1 | 0.4 | 41.3 | 35.7 |
| 13 | 12.6 | 2.7 | 10.0 | 0.6 | 34.0 | 24.8 |
| 14 | 17.8 | 2.2 | 6.1 | 1.0 | 17.5 | 8.0 |
| 15 | 11.9 | 2.7 | 10.1 | 0.9 | 34.8 | 17.8 |
| 16 | 13.3 | 2.3 | 9.1 | 0.5 | 16.4 | 12.1 |
| 17 | 13.8 | 2.2 | 9.8 | 1.1 | 56.4 | 43.1 |
| 18 | 16.3 | 2.0 | 7.1 | 1.1 | 18.4 | 11.0 |
| 19 | 13.9 | 1.2 | 8.8 | 0.2 | 13.3 | 12.4 |

## Appendix C. Summary of mean velocity and depth.

**Table C.1. Summary of velocity and depth at sampling units in the MRR.**

| Sampling unit ID | Mean depth (cm) | SD | Maximum depth (cm) | SD | Mean velocity (cm/sec) | SD | Maximum velocity (cm/sec) | SD |
|---|---|---|---|---|---|---|---|---|
| 0 | - | - | - | - | - | - | - | - |
| 1 | - | - | - | - | - | - | - | - |
| 2 | 35.7 | 19.3 | 37 | 13 | 1 | 0.3 | 2 | 1 |
| 3 | 46.6 | 12.1 | 51 | 15 | 6 | 1.6 | 11 | 3 |
| 4 | - | - | - | - | - | - | - | - |
| 5 | 25.2 | 9.9 | 30 | 16 | 17 | 7 | 29 | 9 |
| 6 | - | - | - | - | - | - | - | - |
| 7 | 19.6 | 6.7 | 23 | 9 | 9 | 4 | 13 | 3 |
| 8 | 18.3 | 3 | 23 | 7 | 12 | 7 | 17 | 6 |
| 9 | - | - | - | - | - | - | - | - |
| 10 | 27.8 | 4.7 | 35 | 7 | 10 | 4.8 | 16 | 11 |
| 11 | 42.2 | 12.2 | 55 | 15 | 6 | 2.3 | 12 | 7 |
| 12 | 43.2 | 14.9 | 59 | 17 | 11 | 3.9 | 20 | 13 |
| 13 | 22.3 | 4.1 | 27 | 6 | 17 | 3.5 | 25 | 2 |
| 14 | 40.8 | 9.4 | 52 | 16 | 1 | 0.2 | 1 | 0 |
| 15 | 30.1 | 6.6 | 34 | 8 | 15 | 5 | 26 | 15 |
| 16 | - | - | - | - | - | - | - | - |
| 17 | - | - | - | - | - | - | - | - |
| 18 | 34.8 | 6 | 44 | 12 | 3 | 0.4 | 5 | 2 |
| 19 | 37.3 | 12 | 44 | 14 | 4 | 3 | 3 | 1 |

-Represents sampling units where measurements where not obtained from at least one trap-site

## Appendix D. Location of data loggers installed.

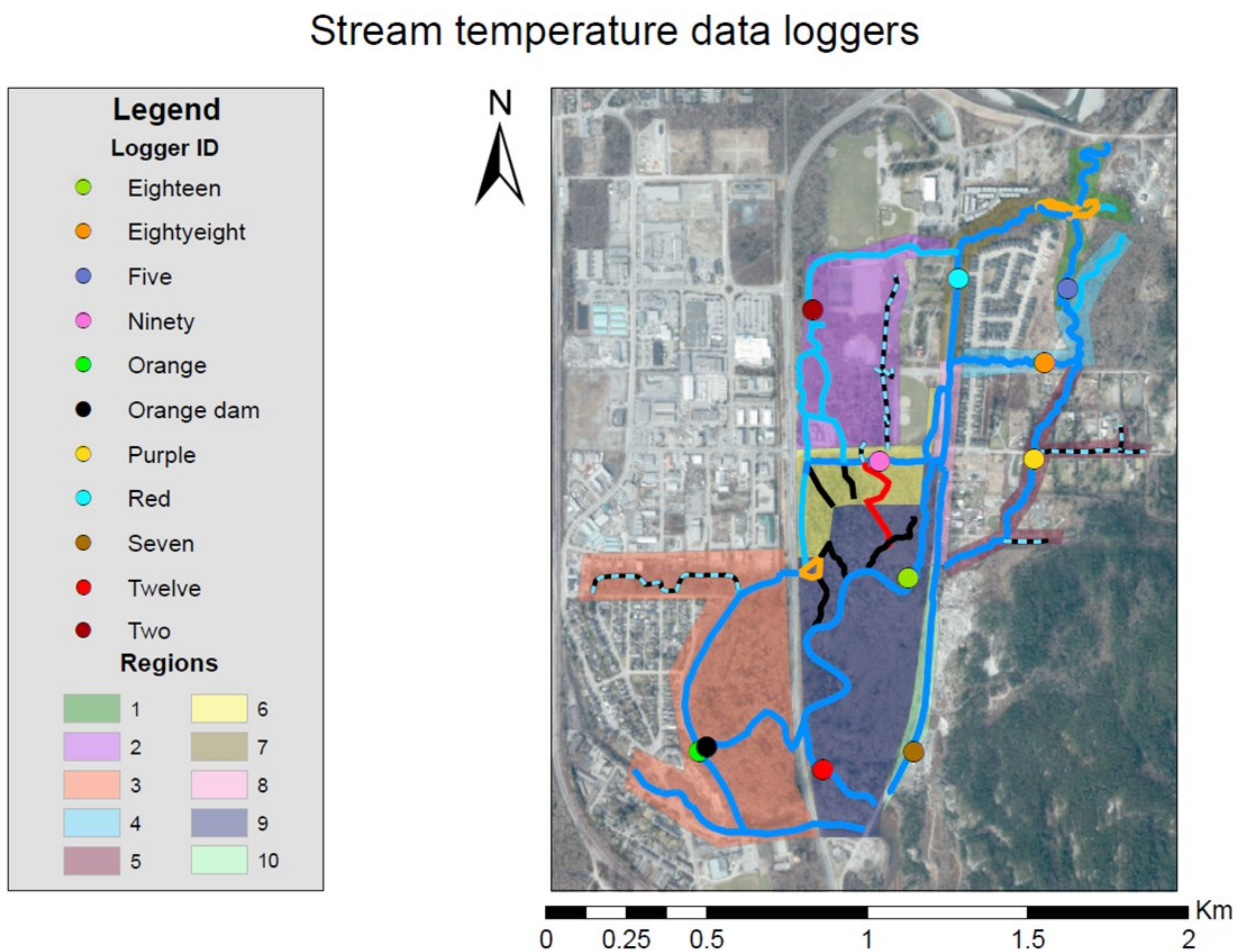


**Figure D.1. Location of data loggers in the MRR project area.**

**Appendix E. Culvert scoring guidelines.**

**Culvert Length Score:** Fish Barrier Scoring. Culvert Length scores are <15m = 0, 15-30m = 3, >30m = 6

**Embeded Score** : If there is not continuous embeddedment, then score = 10.
If continuous embeddedment is present but less than 20% of pipe diameter or less than 30cm deep, then score = 5.
If continuous embeddedment is present and greater than 20% of pipe diameter or greater than 30cm deep, then score = 0.

**Outlet Drop Score:** Fish Barrier Scoring. Outlet Drop scores are <15cm = 0, 15-30cm = 5, >30cm = 10

**Culvert Slope Score:** Fish Barrier Scoring. Culvert Slope scores are <1% = 0, 1-3% = 5, >3% = 10

**Stream Width Ratio:** The Stream Width Ratio is simply the ratio determined by stream channel width over the culvert width shown as:

$$\frac{\text{Channel Width}}{\text{Culvert Width}}$$

**SWR Score** : Fish Barrier Scoring. Stream Width Ratio scores are <1.0 = 0, 1.0-1.3 = 3, >1.3 = 6

**Final Score :** Fish Barrier Scoring. Sum of Fish Barrier Scoring values. Scores of 0-14 indicate passable, 15-19 are potential barriers, and >20 is a barrier

**Barrier Result:** Evaluation of the crossing as a barrier to fish passage, based on FINAL_SCORE. (Passable/Potential barrier/Barrier)

**Figure E.1. Guidelines for scoring culverts for fish passability. Obtained from BC Ministry of Environment (2011).**

## Appendix F. Presence/non detections in sampling units.

**Table F.1. Summary of coho presence/non-detection in sampling units assessed in the MRR. Potential passage limitations exist at sampling units where culverts scored as barrier or potential barrier are located directly upstream or downstream.**

| Sampling unit ID | Coho present? | Potential barrier immediately upstream? | Potential barrier immediately downstream? |
|---|---|---|---|
| 0 | N | N | N |
| 1 | Y | N | Y |
| 2 | Y | N | Y |
| 3 | Y | N | Y |
| 4 | Y | N | Y |
| 5 | Y | Y | Y |
| 6 | N | Y | - |
| 7 | Y | N | N |
| 8 | Y | Y | N |
| 9 | Y | - | - |
| 10 | Y | N | N |
| 11 | Y | N | Y |
| 12 | Y | Y | N |
| 13 | Y | N | N |
| 14 | N | Y | N |
| 15 | Y | Y | Y |
| 16 | Y | N | N |
| 17 | Y | N | N |
| 18 | Y | - | - |
| 19 | Y | - | - |

- Represents sampling units with culverts upstream or downstream that were not assessed.

## Appendix G. Dominant escape cover types and average abundance scores.

**Table G.1. Dominant escape cover and average abundance scores in the MRR. The average dominant escape cover per sampling unit ID, and abundance category of the dominant escape cover type at each trap-site is described. An increasing abundance score indicates higher abundance categories for all escape cover types at each trap-site within a sampling unit.**

| Sampling unit ID | Trap-site 1 | Trap-site 2 | Trap-site 3 | Average dominant cover | Average abundance score |
|---|---|---|---|---|---|
| 1 | Trace | Trace | Trace | Mixed* | 2.3 |
| 2 | Abundant | Moderate | Moderate | Low LWD* | 5 |
| 3 | Abundant | Abundant | Abundant | Low LWD* | 6 |
| 4 | Moderate | Moderate | Abundant | Instream | 4.7 |
| 5 | Abundant | Moderate | Moderate | LWD | 4.7 |
| 6 | Moderate | Trace | Abundant | Mixed* | 4 |
| 7 | Moderate | Moderate | Moderate | Overbank | 4 |
| 8 | Abundant | Abundant | Abundant | Low LWD* | 8 |
| 9 | Moderate | Trace | Moderate | Overbank | 2.7 |
| 10 | Abundant | Abundant | Moderate | LWD | 5.3 |
| 11 | Trace | Moderate | Moderate | Overbank | 3 |
| 12 | Moderate | Moderate | Moderate | Instream | 4.3 |
| 13 | Moderate | Moderate | Moderate | Instream | 3.3 |
| 14 | Abundant | Abundant | Abundant | Instream | 7 |
| 15 | Abundant | Abundant | Abundant | Instream | 4.3 |
| 16 | Trace | Moderate | - | Low LWD* | 2.5 |
| 17 | Moderate | Trace | Abundant | Low vegetation* | 4 |
| 18 | Trace | Moderate | Abundant | Instream | 2.7 |
| 19 | Moderate | Moderate | Moderate | Overbank | 3 |

* Represent multiple dominant escape cover types.

## Appendix H. Summary of average canopy cover %

**Table H.1. Average canopy cover % in sampling units within the MRR.**

| Sampling unit ID | Average canopy percent cover | Canopy cover category % |
|---|---|---|
| 1 | 0% | 0-20% |
| 2 | 41% | 40-70% |
| 3 | 87% | >70% |
| 4 | 77% | >70% |
| 5 | 25% | 20-40% |
| 6 | 86% | >70% |
| 7 | 85% | >70% |
| 8 | 81% | >70% |
| 9 | 58% | 40-70% |
| 10 | 30% | 20-40% |
| 11 | 72% | >70% |
| 12 | 63% | 40-70% |
| 13 | 84% | >70% |
| 14 | 88% | >70% |
| 15 | 0% | 0-20% |
| 16 | 70% | 40-70% |
| 17 | 21% | 20-40% |
| 18 | 71% | >70% |
| 19 | 82% | >70% |

## Appendix I. Juvenile coho captures in 2008-2009.

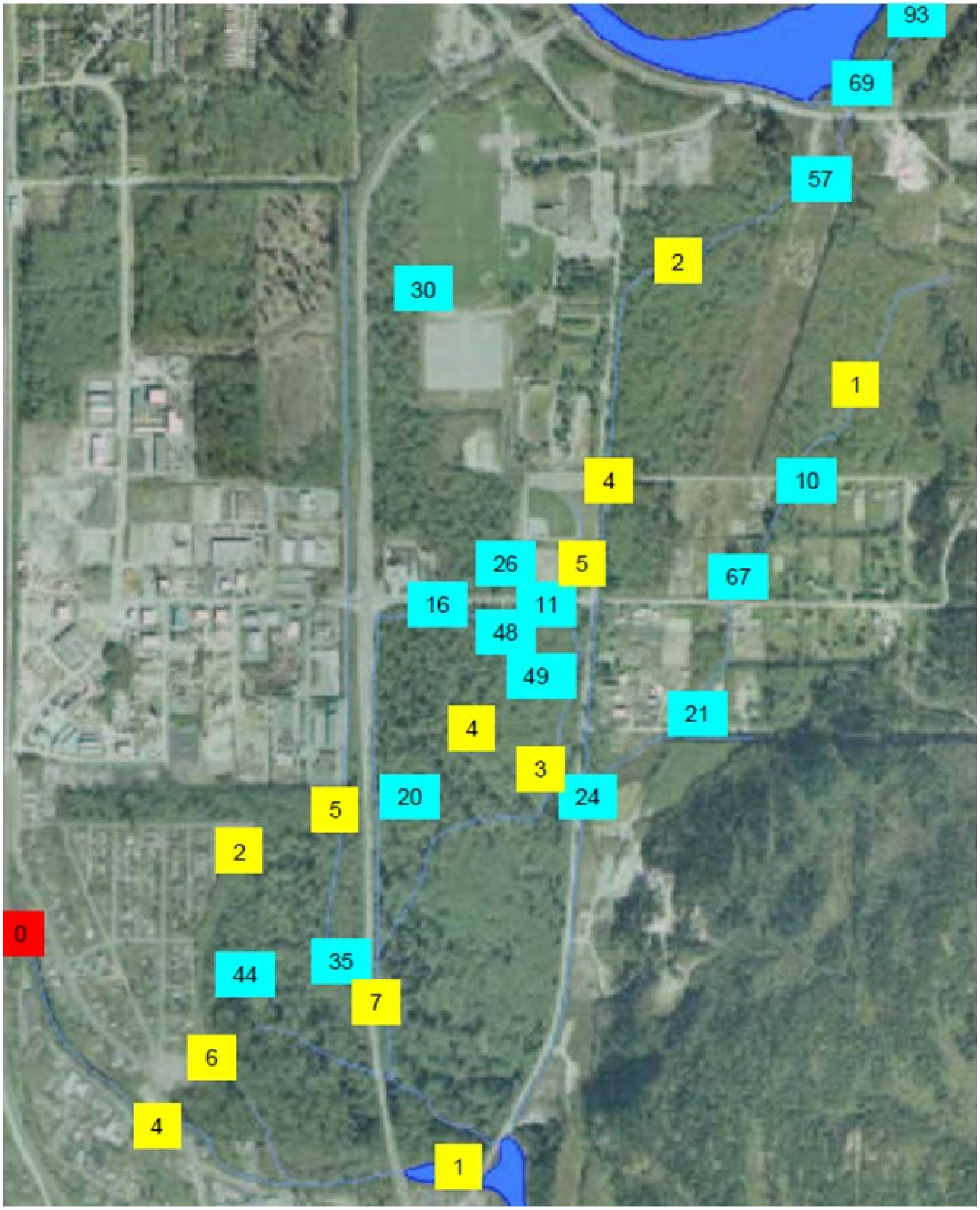


**Figure I.1. Juvenile coho salmon captures during Gee trap sampling in between 2008 and 2009 in the MRR study area (Foy & Gidora 2008; Foy & Gidora 2009).**

## Appendix J. Turbidity from glacial flour.

**Figure J.1. Image shows high turbidity from glacial flour and a mid-channel bar. Image taken August 27th, 2021, looking upstream (north) from the mid-channel of sampling unit ID 11, looking upstream (north).**

## Appendix K. Beaver dam blocking culverts.

**Figure K.2. A) Culvert in Wilson slough blocked by debris. B) Culvert blocked by beaver activity that disconnects sampling unit ID 2 with an ephemeral channel. Images taken on June 17th, and August 18th, 2021, respectively.**

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