Campbell River Project Water Use Plan

Quinsam River Smolt and Spawner Abundance Assessment

Implementation Year 8

Reference: JHTMON-8

Year 8 Annual Monitoring Report

Study Period: April 1, 2021 to March 31, 2022

Laich-Kwil-Tach Environmental Assessment Ltd. Partnership and Ecofish Research Ltd.

October 3, 2022

JHTMON-8: Quinsam River Smolt and Spawner Abundance Assessment

Year 8 Annual Monitoring Report

Prepared for:

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Title page photographs – top left: looking downstream on the Quinsam River (August 16, 2021); top right: drift nets deployed in the Quinsam River (June 10, 2021); bottom left: collecting water quality samples in the Quinsam River (October 07, 2021); bottom right: measuring *in situ* water quality variables in the Quinsam River (July 08, 2021).

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EXECUTIVE SUMMARY

Background and Objectives

Water Use Plans (WUPs) were developed for BC Hydro's hydroelectric facilities through a consultative process. As the Campbell River WUP process reached completion, uncertainties remained with respect to the effects of BC Hydro operations on aquatic resources. To address these uncertainties, several monitoring studies were initiated, including the *Quinsam River Smolt and Spawner Abundance Assessment* (JHTMON-8).

The main objective of the program is to understand whether BC Hydro operations, through changes to streamflow, were the primary cause of changes in fish abundance in the Quinsam River. JHTMON-8 involves monitoring fish abundance and multiple environmental factors (Table i). Final data analysis will involve examining links between fish abundance and environmental factors to better understand what factors limit fish production.

The JHTMON-8 management questions, hypotheses and current status are presented in Table ii. The JHTMON-8 monitoring program was initially developed to focus on the Salmon and Quinsam rivers; however, the Salmon River Diversion Dam was decommissioned in 2017, and the terms of reference for JHTMON-8 were revised in 2018 to solely focus on the Quinsam River watershed. The Quinsam River watershed has high fisheries values and includes the Quinsam River diversion facility, which diverts a portion of the total annual flow to Lower Campbell Reservoir for hydroelectric power generation.

Table i. Summary of JHTMON-8 data collection methods.

Sampling Program	Lead Organization[1]	Method	Timing
Quinsam River Hatchery juvenile downstream migration	DFO/LKT	Fish fence	March - June
Salmon escapement surveys	DFO	Various	September - November
Water quality sampling	LKT	*In situ* and laboratory analysis	May - October
Invertebrate sampling	LKT	Drift sampling	May - October

[1] LKT = Laich-Kwil-Tach Environmental Assessment Ltd. Partnership;
DFO = Fisheries and Oceans Canada

JHTMON-8 commenced in 2014 (Year 1) and eight years of data collection (Table i) have now been completed. In Year 10 (2023), the three management questions in Table ii will be addressed by testing six null hypotheses that are designed to test whether juvenile fish abundance varies among years (*H_01*) and, if so, whether abundance is related to:

- Habitat availability (*H_02*);

- Water quality (H_03);
- Floods (H_04);
- Food abundance (H_05); or
- The abundance of returning adult fish (H_06).

Species of primary interest are Chinook Salmon (*Oncorhynchus tshawytscha*), Coho Salmon (*O. kisutch*) and Steelhead (*O. mykiss*), although the study involves compiling adult escapement and juvenile abundance data for additional Pacific salmon species.

Juvenile Fish Abundance (H_01)

Annual outmigration abundance data provided by DFO for Years 1–8 vary the most for wild Chinook Salmon (~600 to ~360,000 fry) and are lower for wild Coho Salmon (~22,000 to ~57,000 smolts) and Steelhead (~3,000 to ~13,000 smolts; Figure 5). A key result from Year 8 was the particularly high abundance of outmigrating juvenile Chinook Salmon recorded at the Quinsam Hatchery fence (~269,000), which was the second highest value recorded during the eight years of JHTMON-8, and the third highest value recorded overall in the period of record (Figure 5, Figure 6). Historical data compiled to date show considerable inter-annual variability in juvenile fish abundance, with JHTMON-8 priority species varying by at least a factor of four throughout the period of record (Figure 6).

Habitat Availability (H_02)

Regarding H_02 (habitat availability), we initially quantified the Weighted Usable Area (WUA; in m^2) for different life stages of priority species in Year 5. Variability in annual average spawning habitat WUA was similar among Chinook Salmon, Coho Salmon, and Pink Salmon, with maximum differences among years of approximately 100% (i.e., approximately two-fold differences). Annual average rearing and spawning habitat WUA for Steelhead life stages varied throughout the dataset, with variability highest for Steelhead spawning WUA. Flow-habitat relationships have not been previously developed for Pacific salmon rearing habitat. This issue is only potentially applicable to Coho Salmon because the other two species spend limited time rearing in the river. Accordingly, we plan to use Steelhead fry rearing habitat WUA estimates as a proxy for juvenile Coho Salmon rearing habitat. Further analysis of WUA was not undertaken in Year 8, although the WUA calculations will be updated in Year 10 by updating the habitat time series using the latest flow data.

Water Quality (H_03)

Water quality data (relevant to H_03) collected at an index site on the Quinsam River show that the river is typical of streams in coastal BC watersheds with low nutrient concentrations (oligotrophic), near-neutral pH, and low turbidity during baseflow. Measurements of some water quality variables were, at times, outside of the biological optimum ranges for fish species present in the watershed. Specifically, the mean weekly maximum water temperature values observed in Year 8 exceeded the upper limit of the optimum temperature ranges by >1°C at times for the rearing life stage of juvenile

Coho Salmon (31% of the period), Chinook Salmon (23% of the period), and Rainbow Trout (23% of the period). These exceedances of the upper limits of the optimum temperature ranges by >1°C for the rearing life stage were consistent with results from Years 1 to 7 (Table 15 of Appendix A); however, the temporary occurrence of undesirably warm water temperatures from a biological perspective was most pronounced in Year 8, when the highest water temperatures measured during the study to date were recorded during early summer (Figure 10). The maximum daily mean temperature of 25.0°C and the maximum instantaneous water temperature of 26.1°C were measured in late June, coinciding with a prolonged period of unusually high pressure associated with an exceptionally pronounced heat wave.

Furthermore, as in previous years, concentrations of dissolved oxygen less than the provincial guideline for the protection of buried embryos/alevins of some species were recorded in Year 8; however, these values were only marginally less than the guideline (~0.07 mg/L below the guideline minimum during the incubation period).

A background water quality review undertaken in Year 2 and a screening analysis undertaken in Year 4 showed that interannual variability in many of the water quality variables was low. This feature may limit the power of the final analysis to quantify potential effects of water quality on fish abundance (if present), based on analysis of relationships between annual metrics of water quality and fish recruitment. It will therefore be important to continue to evaluate water quality results in the context of water quality guidelines to support qualitative conclusions regarding $H_0 3$. The water quality measurements generally exhibit low variability through time, with measurements of variables such as nutrient concentrations close to method detection limits. This relative consistency in the data indicates that the measurements provide an accurate approximation of the water quality conditions experienced by fish throughout the growing season, which supports an approach of comparing the measurements to water quality guidelines to draw inferences about habitat suitability for fish.

Floods ($H_0 4$)

Preliminary analysis was undertaken in Year 8 to provide initial insights into potential links between hydrologic variability and juvenile fish abundance. Analysis involved analyzing relationships between hydrological metrics and either the shape of stock-recruitment curves (Pacific salmon species), or smolt abundance (Steelhead). The analysis provided insight regarding $H_0 4$ (effect of floods), but also provided proof of concept for analysis of the effects of environmental variables in general that will be completed during Year 10. Two hydrological metrics were considered in Year 8: maximum discharge during the incubation period (which varies by species), and the 7-day minimum discharge during the growing season. The second metric was included to extend consideration of $H_0 4$ beyond only flood events to consider hydrologic variability more generally.

Analysis of stock-recruitment curves suggested that productivity of Pink Salmon (odd year spawning stock) was negatively affected by high discharge during the incubation period, whereas no link was identified for other stocks, including all JHTMON-8 priority species and the even year stock of Pink Salmon (Table 16, Figure 8). A possible mechanism to explain this result is that high flows during the incubation period resulted in reduced incubation success due to redd scour and associated mortality of embryos. Aspects of the ecology of Pink Salmon (e.g., shallow redd construction) likely make this species more sensitive to the effect of high flows during the incubation period than other species. The observation that the effect was significant for the odd year stock but not for even year stock presumably reflects that only the odd year stock was exposed to sufficiently adverse high flow conditions for the effect to be detected.

The relationship was analyzed between low flows in the growing season and the recruitment of Coho Salmon and Steelhead, i.e., the two priority species that rear in the stream for an extended period. No statistically significant link was identified between recruitment of these species and minimum 7-day average discharge. The effects of flow on rearing habitat availability will be considered more directly in Year 10 by considering WUA.

Food Availability (H_05)

Food availability for juvenile salmonids is quantified using drift net sampling undertaken nine times throughout the growing season. Invertebrate drift biomass in the Quinsam River is often highest in the spring, although seasonal trends have been weak, including in Year 8 (2021). Total invertebrate biomass in Year 8 was within the range of previous years (0.05 – 0.59 mg/m^3); however, maximum invertebrate density in Year 8 was higher than previous years, with the maximum value observed in September, and high density also observed in August (Figure 15). High densities were largely attributed to high abundances of Ostracoda (small crustaceans), which have typically been present in low abundance in previous years. In Year 10, we will examine the relationship between invertebrate biomass (i.e., fish food) and juvenile fish abundance to test H_05, although we plan to trial invertebrate density as a secondary measure of food abundance. Interannual variability in invertebrate biomass has so far been generally low, despite seasonal patterns (Figure 16).

Adult Escapement (H_06)

Pacific salmon escapement data collected by DFO have been compiled and analyzed each year to test H_06 (adult returns). In Year 8 (2021), data were available for the period to 2020 when, consistent with previous years, Pink, Coho and Chinook salmon were the most abundant returning species, in that order. Escapement of Chinook Salmon in the Quinsam River in 2020 (8,236) was above the mean value for the period of historical record (1957–2019), whereas estimated escapement of Coho Salmon in 2020 (12,721) was close to the mean value of the dataset (Table 14, Figure 4). Pink Salmon escapement in the Quinsam River in 2020 (513,567) was higher than the mean value for the dataset (136,840; Table 14). The estimated Chum Salmon escapement in 2020 (20) was particularly low as it was the fourth lowest count recorded in the 61-year dataset, with the 1993 count (6) the lowest (Figure 4). The Chum Salmon estimate is likely biased low as the sampling period does not usually capture the full duration of the migration period.

Based on analysis initiated in Year 7 and updated in Year 8, stock-recruitment relationships generally show a positive relationship between adult abundance and juvenile outmigration. However, consistent with general relationships observed elsewhere, there is evidence of density dependence in Pacific salmon species, except for Chinook Salmon, for which spawner abundance is the lowest of the four species analyzed. Stock-recruitment curves will be further updated in Year 10 to formally test H_06. Such relationships will also allow for variability in spawner abundance to be accounted for when analyzing links between juvenile fish abundance and environmental factors.

Table ii. Status of JHTMON-8 objectives, management questions and hypotheses after Year 8.

Study Objective	Management Questions	Management Hypotheses	Year 8 (2021/2022) Status
The objective is to address the management questions by collecting data necessary to test the impact hypotheses. Analysis is designed to understand whether BC Hydro operations, through changes to flow, are the primary cause of historical changes in fish abundance. This study will reduce uncertainty about factors that limit fish abundance in the Quinsam River.	1. What are the primary factors that limit fish abundance in the Campbell River System and how are these factors influenced by BC Hydro operations? The stream of interest in this monitor is the Quinsam River. 2. Have WUP-based operations changed the influence of these primary factors on fish abundance, allowing carrying capacity to increase? 3. If the expected gains in fish abundance have not been fully realized, what factors if any are masking the response and are they influenced by BC Hydro operations?	*H_01: Annual population abundance does not vary with time (i.e., years) over the course of the Monitor*	-Juvenile fish have been sampled annually at the Quinsam Hatchery salmon counting fence to derive total outmigration estimates -Inter-annual variability has been observed in the abundance of priority species so we expect to reject this hypothesis in Year 10
		H_02: Annual population abundance is not correlated with annual habitat availability as measured by Weighted Usable Area (WUA)	-In Year 5, we used existing flow-habitat relationships to estimate WUA of habitat for priority species for 1975-2017 -Additional work relating to this hypothesis was not undertaken in Year 8; relationships will be updated in Year 10 for the final analysis to test this hypothesis
		H_03: Annual population abundance is not correlated with water quality	-Water quality has been measured each year through the growing season at a single index site -Water quality is generally within ranges to support healthy salmonid populations, although there are some exceptions -In Year 8, high water temperatures occurred in early summer associated with a heat wave -Analysis will be undertaken to test this hypothesis in Year 10. Low variability in independent variables is expected to limit the statistical power of this analysis; comparisons with water quality guidelines will be an important line of evidence
		H_04: Annual population abundance is not correlated with the occurrence of flood events	-Flow data collected by the Water Survey of Canada have been used to calculate flow metrics that will be used in the final analysis

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Study Objective	Management Questions	Management Hypotheses	Year 8 (2021/2022) Status
			-Flow metrics have been variable throughout the monitoring period, affected by background hydrological factors and BC Hydro operations -Floods have occurred during the JHTMON-8 monitoring period during sensitive life history periods (notably Pacific salmon incubation) -Preliminary analysis in Year 8 showed a potential negative effect of high flows during the incubation period on recruitment of Pink Salmon (odd year stock only); effects were not identified for other stocks -Analysis will be undertaken to test this hypothesis in Year 10
		$H_0$5: Annual population abundance is not correlated with food availability as measured by aquatic invertebrate sampling	-Aquatic invertebrate biomass has been measured each year through the growing season at a single index site -Seasonal patterns have been observed although they are inconsistent among years -Analysis will be undertaken to test this hypothesis in Year 10, although low inter-annual variability in invertebrate biomass may limit the statistical power of this analysis. Supplementary lines of evidence such as comparisons with data from other watersheds may be required in Year 10
		$H_0$6: Annual smolt abundance is not correlated with the number of adult returns	-Adult salmon escapement data have been compiled annually from DFO records -Initial stock-recruitment curves have been developed, which will be updated to test this hypothesis in Year 10

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1. INTRODUCTION

1.1. Background

Water use planning exemplifies sustainable work in practice at BC Hydro. The goal is to provide a balance between the competing uses of water that include fish and wildlife, recreation, and power generation. Water Use Plans (WUPs) were developed for all BC Hydro's hydroelectric facilities through a consultative process involving local stakeholders, government agencies and First Nations. The framework for water use planning requires that a WUP be reviewed on a periodic basis and there is expected to be monitoring to address outstanding management questions in the years following the implementation of a WUP.

As the Campbell River WUP process reached completion, several uncertainties remained with respect to the effects of BC Hydro operations on aquatic resources. A key question throughout the WUP process was "what limits fish abundance?" For example, are fish abundance and biomass limited by available habitat, food, hydrological perturbations, or other ecological interactions? Answering this question is an important step to better understand how BC Hydro operations in the watershed affect fisheries, and to effectively manage water uses to protect and enhance aquatic resources. To address this uncertainty, monitoring programs were designed to assess whether fish benefits are being achieved under the WUP operating regime, and to evaluate whether limits to fish production could be improved by modifying operations in the future. The *Quinsam River Smolt and Spawner Abundance Assessment* (JHTMON-8) is one of several monitoring studies associated with the Campbell River WUP. JHTMON-8 focuses on monitoring fish populations and environmental factors that may influence fish abundance in the Quinsam River. Prior to Year 5, JHTMON-8 also focused on the Salmon River; however, this component of the program was removed following a revision to the terms of reference (BC Hydro 2018a) after the Salmon River Diversion Dam was decommissioned in 2017, meaning that there is no longer any mechanism for BC Hydro operations to affect fish populations in the Salmon River. Accordingly, the Salmon River is not considered further in this report.

This report describes field work and analysis undertaken in Year 8 of JHTMON-8, which commenced on April 1, 2021. Detailed analysis that addresses the management questions based on data collected throughout all years of the study will be undertaken in Year 10.

1.2. The Quinsam River and Diversion

The Quinsam River is located to the west of the city of Campbell River on the east coast of Vancouver Island, British Columbia. The Quinsam River diversion facility has historically diverted a portion of water from the river mainstem to Lower Campbell Reservoir to generate hydroelectricity downstream at Ladore and John Hart generation stations (Map 1). Details of the diversion infrastructure and operations are summarized below based on the Campbell River System WUP (BC Hydro 2012).

The Quinsam River is the only major tributary of the lower Campbell River, entering the Campbell River approximately 3.5 km upstream of the mouth. The Quinsam flows through a series of lakes and has a mainstem length of 45 km (excluding lakes), a watershed area of 283 km^2, and a mean annual discharge near the mouth of 8.5 m^3/s. The river has high fisheries values, supporting an assemblage of native salmonid species (Burt 2003; see Table 1 for periodicity information). The Quinsam River Hatchery was constructed in 1957 and is located 3.3 km upstream from the confluence with the Campbell River. The hatchery has been active in the watershed, augmenting populations of Chinook Salmon, Pink Salmon, Coho Salmon and Cutthroat Trout since 2014 (Year 1), with Chum Salmon and Steelhead also released in previous years (DFO 2017). Smolt and fry life stages that are ready for downstream migration to the ocean are released from the hatchery during the spring. In addition, juvenile Coho Salmon, Steelhead and Chinook Salmon have been out-planted to the upper watershed since 1978 to promote adult returns upstream of the hatchery (Burt 2003).

The Quinsam River Diversion comprises a small concrete gravity storage dam, a concrete gravity diversion dam, a concrete flume and the natural waterways that convey water to Lower Campbell Reservoir. Non-diverted water is conveyed to the Quinsam River via an undersluice gate or the free crest weir. The dams were both constructed in 1957.

A total of 100 million m^3 is licensed to be diverted annually and the design capacity of the Quinsam River Diversion is 8.50 m^3/s. The WUP stipulates maximum down ramping rates (Table 2) and minimum flows (when naturally available) in the Quinsam River downstream of the diversion dam (Table 3).

Overview of the Quinsam River watershed

Legend

- Dam
- Stream

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 1 2 3 4 5 6 7 km

Scale: 1:180,000

NO.	DATE	REVISION	BY
1	2021-12-10	1230_BCH_CRFacilities_3603_20201210	CGA
2			
3			
4			
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Date Saved: 2021-12-10
Coordinate System: NAD 1983 UTM Zone 10N

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Map 1

Path: M:\Projects-Active\1230_JHTMON\MXD\Overview\1230_BCH_CRFacilities_3603_20201210.mxd

Table 1. Periodicity of important fish species in the Quinsam River system (from BC Hydro files for Campbell River Water Use Plan, dated 2001).

Species	Life History Stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Chinook Salmon	Adult migration												
	Spawning										P	P	
	Incubation[1]												
	Emergence begins												
	Rearing												
	Juvenile migration				F	S							
Chum Salmon	Adult migration												
	Spawning											PP	
	Incubation												
	Emergence begins												
	Juvenile migration			F									
Coho Salmon	Adult migration												
	Spawning											PPP	
	Incubation												
	Emergence begins												
	Rearing												
	Juvenile migration			F	S								
Pink Salmon	Adult migration												
	Spawning									P			
	Incubation												
	Emergence begins												
	Juvenile migration			F									
Rainbow Trout	Adult migration												
	Spawning												
	Incubation												
	Rearing												
	Juvenile migration												
Sockeye Salmon	Adult migration												
	Spawning												
	Incubation												
	Emergence begins												
	Rearing												
	Juvenile migration			F									
Steelhead (winter run; summer-run steelhead not present)	Adult migration												
	Spawning			PP									
	Incubation												
	Emergence begins												
	Rearing												
	Juvenile migration				S								

Critical times

F = fry migration begins, S = smolt migration begins, P = peak spawning

[1] The start of incubation (previously mid-Oct) was amended to align with the start of spawning, as based on Burt (2003), which states that "Spawning takes place from the beginning of October…"

Table 2. Quinsam River maximum permitted down ramping rates (BC Hydro 2012).

Stream	Discharge (m^3/s)	Maximum Down Ramping Rate ($m^3/s/h$)
Quinsam River	>4.0	8.5
	≤4.0	1.0
Quinsam Diversion	>2.0	N/A
	≤2.0	1.0

Table 3. Minimum permitted discharge in the Quinsam River (BC Hydro 2012).

Date	Minimum Discharge in Quinsam River (m^3/s)
Jan 1 to Apr 30	2.0
May 1 to Oct 31	1.0
Nov 1 to Dec 31	0.6

1.3. Background to Water Use Decision

The operating conditions (minimum flow requirements) prescribed in the WUP for the Quinsam Diversion (Table 3) match those of the "MinRisk 2c" option that was recommended by a Consultative Committee because it represented "the best trade off of all gains and losses" (Campbell River WUP CC 2004). This recommendation was based on evaluating a power/financial performance measure alongside the following four biological performance measures (Campbell River WUP CC 2004):

- Fish habitat risk: the average annual probability that Rainbow Trout and Chinook Salmon usable habitat will decline below 60% of the maximum available, calculated using a meta-analysis method;
- Fish passage (considered in JHTMON-6);
- Fish overwintering success; and
- Drawdown in Upper Quinsam Lake/Wokas Lake, with the assumption that drawdown has a negative effect on fish and wildlife resources.

The first two biological performance measures listed above were evaluated based on scores that were standardized to a scale from 0–1, whereas the second two measures were evaluated qualitatively by considering the direction of predicted change (Table 7-6 in Campbell River WUP CC 2004). The

Quinsam Diversion operating conditions prescribed in the WUP are those that were evaluated to provide the best biological outcomes of the options consider that involved flow diversion.

1.4. Management Questions and Hypotheses

The JHTMON-8 monitoring program aims to address the following three management questions, with reference to the Quinsam River:

1. What are the primary factors that limit fish abundance in the Campbell River system and how are these factors influenced by BC Hydro operations?
2. Have WUP-based operations changed the influence of these primary factors on fish abundance, allowing carrying capacity to increase?
3. If the expected gains in fish abundance have not been fully realized, what factors if any are masking the response and are they influenced by BC Hydro operations?

In addressing the questions, the monitoring program is designed to test the following five null hypotheses:

H_01: Annual population abundance does not vary with time (i.e., years) over the course of the Monitor.

H_02: Annual population abundance is not correlated with annual habitat availability as measured by Weighted Usable Area.

H_03: Annual population abundance is not correlated with water quality.

H_04: Annual population abundance is not correlated with the occurrence of flood events.

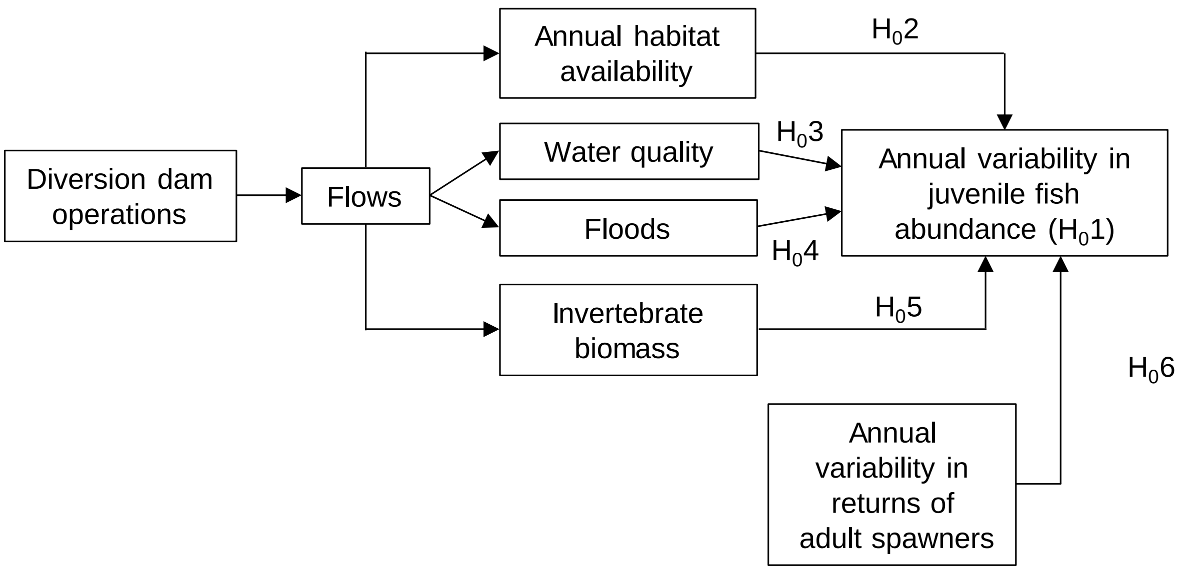
H_05: Annual population abundance is not correlated with food availability as measured by aquatic invertebrate sampling.

H_06: Annual smolt abundance is not correlated with the number of adult returns.

The basis of JHTMON-8 is outlined conceptually in Figure 1. The monitoring program is designed to first establish whether there is among-year variability in fish abundance (H_01). The program is then designed to collect data to examine whether inter-annual variability in fish abundance is related to important environmental factors that could be influenced by BC Hydro operations, specifically: Weighted Usable Area (WUA) of habitat (H_02); water quality (H_03); an accumulated flood risk index during the spawning and incubation periods (H_04), or; invertebrate abundance (food availability; H_05). The study will also investigate whether annual variability in juvenile fish abundance is affected by annual variability in salmon spawner escapement (H_06) – a factor that is not directly influenced by diversion dam operations.

The final step in the analysis will involve evaluating whether BC Hydro operations, via changes to flow, are the primary cause of any changes to environmental factors that are shown to be drivers of fish production. This step may require a mixture of quantitative and qualitative analysis as it will be easier to distinguish changes due to BC Hydro operations from those due to background variability for some factors (e.g., WUA) than others (e.g., invertebrate drift). To address Management Question 2, it will be necessary to compare pre-and post-WUP conditions, although this will not be possible for some components that lack pre-WUP data (e.g., invertebrate drift biomass). Such pre- and post-WUP comparisons will therefore focus on analyzing Quinsam River fish abundance, WUA, and flow data. We do not plan to compare changes in variables with targets that have been defined *a priori*, because we are not aware that these were developed[1]. Instead, conclusions about the biological significance of changes will be made based on multiple lines of evidence such as the effect size and, potentially, trends in other watersheds. Such conclusions may then inform decisions about whether changes to the WUP or alternative mitigation are necessary to achieve desired outcomes for fish.

Figure 1. Effect-pathway diagram showing the context of the six hypotheses that the JHTMON-8 monitoring program sets out to address.



1.5. Scope of the JHTMON-8 Study

1.5.1. Overview

The JHTMON-8 study has been designed to build upon monitoring that is already occurring in the Quinsam River watershed. This allows the study to integrate established work programs and provides an opportunity to incorporate historical data into the analyses.

Priority species for JHTMON-8 in the Quinsam River are Chinook Salmon, Coho Salmon, and Steelhead, although Pink Salmon is also of interest. Juvenile fisheries data for the Quinsam River are obtained via operation of a salmon counting fence at Quinsam River Hatchery to enumerate

[1] We recognize this is implied in Management Question 3 ("the expected gains"); however, we assume this relates to a general expectation that the WUP will qualitatively improve fish productivity in the Quinsam River.

downstream juvenile migration of a range of species. In addition to these juvenile abundance datasets, adult escapement data obtained by Fisheries and Oceans Canada (DFO) for a range of Pacific salmon species during routine monitoring are also considered as part of JHTMON-8. Water quality and invertebrates are sampled at a single index site and flow data are obtained from gauges maintained by Water Survey of Canada.

Further information about the scope and objectives of specific sampling programs is provided in the sub-sections below, which also includes an overview of how impact hypotheses will be tested for the Quinsam River in Year 10.

1.5.2. Fish Population Assessments

The JHTMON-8 juvenile fish sampling program is designed to ensure that the error associated with fish sampling methods is sufficiently small to assess among-year variability in fish abundance. The fish abundance data will first be used to test $H_0 1$: '*annual population abundance does not vary with time (i.e., years) over the course of the Monitor*' (Section 1.4).

The program was designed to enumerate both adult and juvenile life stages to allow relationships between the numbers of adult spawning fish and juvenile recruitment to be examined. This enables testing of $H_0 6$ *'annual smolt abundance is not correlated with the number of adult returns'* for the Quinsam River, which will help to tease apart the extent to which variations in abundance reflect either variations in adult returns (dependent on marine conditions and harvest) or variations in juvenile survival (dependent on freshwater conditions). In Year 5, historical data collected at the Quinsam Hatchery salmon counting fence since the 1970s were collated, increasing the duration of the dataset available for this analysis. In Year 7 (2020), we developed initial stock (spawner)-recruitment relationships to describe relationships between adult spawner abundance and associated smolt abundance. These relationships will be further updated in Year 10.

For at least some species and life stages, we anticipate that biologically significant interannual variability in juvenile fish abundance will be detected, i.e., after accounting for sampling error, we will be confident that variability among years in juvenile abundance occurred at the watershed scale. It will then be necessary to use these data to test the remaining hypotheses to determine whether there are relationships between the observed variability in fish abundance, and variations in key environmental factors, namely habitat ($H_0 2$), water quality ($H_0 3$), floods ($H_0 4$) and food availability ($H_0 5$). Analysis has been undertaken in Year 8 to provide a proof of concept for the general approach that we propose to take in Year 10 to test the study hypotheses (Section 2.1.3).

1.5.3. Weighted Usable Area (WUA) of Habitat

Changes to flow affect the width, depth, and velocity of a stream, which in turn affect the extent and suitability of fish habitat. Changes to these factors have the potential to limit juvenile fish production by either changing spawning habitat or, for stream-rearing species, changing instream rearing habitat conditions. As part of JHTMON-8, annual WUA metrics will be calculated for the Quinsam River to quantify how habitat varies among years for individual life stages of priority fish species. WUA will be calculated using existing flow–habitat relationships that were developed based on field work that was undertaken by D. Burt and Associates to inform WUP development[2], as described in Solander *et al.* (2004). Analysis will then be undertaken in Year 10 to examine whether variation in juvenile fish abundance is related to variation in applicable WUA metrics that are specific to individual species and life stages. Results of this analysis will be used to test H_02: *annual population abundance is not correlated with annual habitat availability as measured by Weighted Usable Area*.

In Year 5, we reviewed flow–habitat relationships, compiled flow data, and completed analysis to estimate a range of WUA metrics for the period since 1974, which matches the period for which juvenile fish abundance data have been compiled for the Quinsam River (Abell *et al.* 2019). To test H_02, this WUA dataset will be updated in Year 10 using the existing flow–habitat relationships and the most recent flow data.

1.5.4. Water Quality

Healthy fish populations require water quality variables to be within confined ranges. This range of suitable conditions varies depending on the individual variable, fish species, and life stage. The objective of the JHTMON-8 water quality monitoring is to measure biologically important water quality variables to provide data to test H_03: '*annual population abundance is not correlated with water quality*' (Section 1.4). Approaches to incorporate water quality data into the final analysis were evaluated in the Year 4 Annual Report (Sharron *et al.* 2018) and complete analysis will be undertaken at the end of the ten-year monitor to examine whether water quality is expected to limit fish abundance. If a relationship is detected (i.e., the null hypothesis is rejected), then we will evaluate whether BC Hydro operations are likely to have adversely affected water quality. This will be done as part of this study to help address Management Question 1 and 2. If required, we expect this analysis to be predominantly qualitative and it will involve considering the pathways of effect by which BC Hydro operations may affect water quality.

Thus, a key assumption of this aspect of the study is that the water quality data collected suitably reflect variability of water quality in time and space and are representative of the conditions experienced by fish communities (discussed further in Dinn *et al.* 2016). We recognize that grab sampling provides an instantaneous "snapshot" of water quality and therefore it will be necessary to critically evaluate whether the data are suitably representative of conditions at the site during the growing season. This evaluation will require considering the possible influence of biogeochemical processes (e.g., that drive diurnal variability in dissolved oxygen), in addition to assessment of temporal variability among measurements, e.g., by comparing measurements collected during the same month

[2] Note that, contrary to the revised TOR (BC Hydro 2018a), it is unnecessary to use information from JHTMON-6 as these relationships for the Quinsam River were developed prior to initiation of JHTMON-8. Developing flow-habitat relationships for the Salmon River was previously part of the scope of JHTMON-6; however, this is no longer applicable following decommissioning of the Salmon River Diversion. The current scope of JHTMON-6 includes quantifying flow-habitat relationships for the Quinsam River diversion route via Miller Creek, but not the Quinsam River mainstem (BC Hydro 2018b).

but during different years. A single mainstem index site was selected in the Quinsam River that was assumed to be representative of water quality in the wider watershed.

1.5.5. Floods

High flows have potential to adversely affect fish populations due to a variety of mechanisms that include redd scour, delayed redd construction, redd desiccation due to spawning occurring along channel margins during high flows, sediment intrusion, physical shock, or reduced holding opportunities shortly after emergence (reviewed in Gibbins *et al.* 2008). Discharge data are collected at numerous sites in the Quinsam River by the Water Survey of Canada. These data will be used to quantify the occurrence of high flow events during individual years to test H_04: '*annual population abundance is not correlated with the occurrence of flood events*' (Section 1.4).

During Year 3, we evaluated suitable hydrological metrics to quantify key flow characteristics that have potential to influence fish productivity (Abell *et al.* 2017). Based on that evaluation, we quantified the maximum daily mean discharge each year that occurs during the spawning and incubation periods of key species. In Year 8, we undertook preliminary analysis of the effect of high flows during the spawning and incubation period (Section 2.1.3), recognizing that these life stages have been shown to be particularly sensitive to the effects of high flows (e.g., Cattanéo *et al.* 2002). We also extended the analysis to consider hydrological variability more generally by analyzing the potential effect of low flows in the summer (Section 2.1.3), recognizing that such flow conditions can limit the abundance of juvenile fish species that rear in freshwater throughout the summer, e.g., Coho Salmon (Matthews and Olson 1980).

1.5.6. Invertebrate Drift

Invertebrates typically form the bulk of the diet of both juvenile and resident adult salmonids in rivers (Quinn 2005). Invertebrate populations can vary due to a range of factors and therefore variability in the abundance and biomass of invertebrates can limit the growth of salmonids in rivers. The objective of the JHTMON-8 invertebrate sampling is to provide data to test H_05 *annual population abundance is not correlated with food availability as measured by aquatic invertebrate sampling* (Section 1.4). Analysis will be undertaken in Year 10 to examine whether there are any relationships between fish abundance and food availability, as inferred from invertebrate biomass. If a relationship is detected (i.e., the null hypothesis is rejected), then we will evaluate whether BC Hydro operations are likely to have adversely affected invertebrate drift biomass. This will be done as part of this study to address Management Question 1 and 2. If required, we expect this analysis to be predominantly qualitative and it will involve considering the pathways of effect by which BC Hydro operations may affect invertebrate drift. These pathways relate to changes in flow and include changes to invertebrate habitat availability, in addition to changes to habitat suitability due to changes in flow velocity or sedimentation. These changes can affect total invertebrate biomass and thus food availability for fish. Furthermore, effects may vary among invertebrate taxa, creating the potential for changes to invertebrate community structure and diversity, which can potentially influence the quality of food available for fish.

A key objective is therefore to collect invertebrate data that reflect variability in time and space of watershed invertebrate communities that are representative of the food available to salmonids. Invertebrate drift includes dislodged benthic invertebrates, terrestrial invertebrates entrained in the stream, and invertebrates originating from riparian areas. Johnson and Ringler (1980) studied the diets of Coho Salmon fry and Steelhead fry and found that Coho Salmon fry fed more on terrestrial invertebrates than on aquatic invertebrates. The major terrestrial invertebrate groups that contributed to Coho Salmon fry diets were hymenopterans, coleopterans, homopterans, dipterans, and lepidopteran larvae. The main benthic groups were ephemeropterans, plecopterans, and trichopterans (EPT), as well as chironomids, and tipulids (both Diptera). Steelhead fry mainly fed on aquatic invertebrates, which were ephemeropterans, chironomids, trichopterans and tipulids. Based on Johnson and Ringler (1980), salmonids feed on a wide diversity of invertebrate taxa, including EPT taxa (indicative of good water quality) and other taxa such as dipterans that are more tolerant of disturbed environments. Other studies have also shown that a wide range of invertebrate taxa are present in drift and they provide an important food resource for salmonids, with all macroinvertebrates generally assumed to provide potential food for rearing salmonids once they are present in drift (e.g., Rader 1997). Based on these studies, we expect that total invertebrate drift biomass provides a suitable metric of food availability to rearing salmonids in the Quinsam River.

A single mainstem index site was selected that was assumed to be representative of the invertebrate communities present in the wider watershed. Invertebrate drift biomass is measured as a proxy for food availability, although invertebrate community composition is also examined to provide information on food quality. Drift sampling is undertaken during the growing season when rearing juvenile salmonid are actively feeding. In addition, a single kick net sample is collected in September. Kick net sampling targets benthic invertebrates and is therefore less representative of the total abundance of food available to fish. However, kick net sampling based on the Canadian Aquatic Biomonitoring Network (CABIN) protocol (Environment Canada 2012) has been used more widely to characterize stream invertebrate communities throughout Canada. Data collected using this method can be used to evaluate the wider ecological integrity of the streams, based on comparisons with the Environment Canada database of Georgia Basin reference sites (e.g., see Strachan *et al.* 2009).

2. METHODS

2.1. Fish Population Assessments

2.1.1. Quinsam River Salmon Escapement

Annual salmon spawner escapement estimates have been derived for the Quinsam River since the 1950s by DFO and its predecessors. These estimates are collected as part of wider salmon stock assessment work and provide important data to support JHTMON-8. The results of summer and fall 2020 surveys were finalized during Year 8 (2021). Escapement estimates were obtained from DFO's New Salmon Escapement Database (nuSEDS) and are reported here alongside results from previous years. Data for the Quinsam River will support analysis scheduled for later during JHTMON-8 to examine relationships between abundance of adult spawning fish and corresponding counts of juvenile fish in successive years.

Methods used in the 2020 surveys are summarized in Table 4 for the Quinsam River, based on information provided in the nuSEDS database (DFO 2021). Methods undertaken in previous years of JHTMON-8 are summarized in previous annual reports. Surveys of individual species conducted by DFO conform to one of six estimate classification types, ranging from Type-1 (most rigorous, almost every fish counted individually) to Type-6 (least rigorous, determination of presence/absence only). The estimate classification types are reported in Table 4, with further general details about survey types provided in Table 5.

Table 4. Methods used for 2020 salmon spawner escapement counts on the Quinsam River (DFO 2021). See Table 5 for descriptions of estimate classification types.

	Salmon Species				
	Chinook	**Chum**	**Coho**	**Pink**	**Sockeye**
Estimate classification	2	3	2	2	3
Number of surveys	Unknown	Unknown	Unknown	Unknown	Unknown
Date of first inspection	14-Jul	14-Jul	14-Jul	14-Jul	14-Jul
Date of last inspection	19-Nov	18-Nov	18-Nov	19-Nov	18-Nov
Estimation method	Mark and recap. (Petersen)	Fixed site census	Fixed site census	Fixed site census	Fixed site census

Table 5. Summary of definitions of salmon spawner escapement estimate classification types reported in Table 4 (DFO 2021).

Estimate Classification Type	Abundance Estimate Type	Resolution	Analytical Methods	Reliability (Within Stock Comparisons)	Units	Accuracy	Precision
1	True	High resolution survey method(s): total, seasonal counts through fence or fishway with virtually no bypass	Simple	Reliable resolution of between year differences >10% (in absolute units)	Absolute abundance	Actual or assigned estimate and high	± 0%
2	True	High resolution survey method(s): high effort (5 or more trips), standard methods (e.g., equal effort surveys executed by walk, swim, overflight, etc.)	Simple to complex multi-step, but always rigorous	Reliable resolution of between year differences >25% (in absolute units)	Absolute abundance	Actual or assigned estimate and high	Actual estimate, high to moderate
3	Relative	Medium resolution survey method(s): high effort (5 or more trips), standard methods (e.g., mark-recapture, serial counts for area under curve, etc.)	Simple to complex multi-step, but always rigorous	Reliable resolution of between year differences >25% (in absolute units)	Relative abundance linked to method	Assigned range and medium to high	Assigned estimate, medium to high
4	Relative	Medium resolution survey method(s): low to moderate effort (1-4 trips), known survey method	Simple analysis by known methods	Reliable resolution of between year differences >200% (in relative units)	Relative abundance linked to method	Unknown assumed fairly constant	Unknown assumed fairly constant
5	Relative	Low resolution survey method(s): low effort (e.g., 1 trip), use of vaguely defined, inconsistent or poorly executed methods.	Unknown to ill defined inconsistent or poorly executed	Uncertain numeric comparisons, but high reliability for presence or absence	Relative abundance, but vague or no ID on method	Unknown assumed highly variable	Unknown assumed highly variable
6	Presence or absence	Any of above	N/A	Moderate to high reliability for presence/absence	Present or absent	Medium to high	Unknown

2.1.2. Quinsam River Hatchery Salmon Counting Fence Operations

The age of juvenile fish captured at the fence varies by species, reflecting differences in life histories. Coho Salmon, Cutthroat Trout, and Steelhead are captured at the fence at the smolt stage (aged 1+ or older) and Chinook Salmon, Pink Salmon, and Chum Salmon at the fry stage (aged 0+). Pink Salmon and Chum Salmon emigrate from the river immediately or soon after emergence (Burt 2003). In the Quinsam River, Chinook Salmon migration from the river occurs either soon after emergence or a few months later. Those Chinook Salmon that rear for a full summer and winter are believed to do so in the estuary (Burt 2003). The life history strategies adopted by Steelhead, Cutthroat Trout, and Coho Salmon are more variable, and the timing of emigration from the river varies from the first spring to three years after emergence.

In Year 8, sampling was undertaken from March 9 to June 18, 2021. Fish were caught using inclined plane traps (Wolf traps) that capture a proportion of the fish that migrate downstream through the fence, with the aim to capture salmonid fry and smolts as they outmigrate to the ocean. Traps were deployed continuously during the sampling period. Three traps are consistently used, but the number of openings varied during the sampling period. During the period of Pink Salmon fry migration, 16 openings are typically fished, while during the period of smolt migration five openings are typically fished (Forktamp, pers. comm. 2019). Pink Salmon fry typically migrate at night and therefore traps were set overnight from approximately 15:00 to 09:00 during sampling from March 9 to April 23, 2021. For the remainder of the sampling period, traps were set constantly during the times when fish were not being processed. Target species during this time were Steelhead (kelts and smolts), Coho Salmon (smolts), Chinook Salmon (fry), Chum Salmon (fry), Sockeye Salmon (fry), Cutthroat Trout (kelts and smolts) and Dolly Varden (smolts).

Total downstream migration estimates for individual species and life stages were calculated by dividing fish capture numbers by life-stage-specific (i.e., fry and smolt) capture efficiency coefficients. The capture efficiency estimates reflect inherent differences in catchability between life stages, differences in catchability due to variability in environmental conditions (e.g., flow) at the time of sampling, and the differences due to the way the traps are operated during the fry and smolt migration periods. The capture efficiency coefficients were derived from mark-recapture studies in the Quinsam River. For Pink Salmon fry, capture efficiency was estimated based on the results of releases of wild fish marked with Bismarck brown dye. The fish were captured in the trap, marked with the dye, and released approximately 350 m upstream of the fence. A total of five releases were undertaken on March 24, March 30, April 6, April 13, and April 19; a total of 20,695 fish were released (3,412–4,542 per experiment). Separate catch efficiency estimates were derived for Coho Salmon smolts based on three releases of wild Coho Salmon smolts marked with pelvic fin clips (alternating between right and left between experiments). As for fry, smolts were captured in the traps and released upstream of the traps. Releases were undertaken on May 5 (317 fish), May 11-12 (354 fish) and May 18-20 (316 fish), with a total of 987 fish released. Capture efficiency was calculated as k/K (where k is the number of marked fish recaptured and K is the total number of fish marked in the study). The capture efficiency coefficients are then applied in chronological order, matching the date of observed counts to the date of the last mark-recapture experiment. The capture efficiency coefficients were used to estimate the abundance of fry and smolts of all salmonids that emigrate during the respective fry or smolt trapping periods (Pink Salmon, Sockeye Salmon, Chum Salmon, Chinook Salmon, Coho Salmon, Steelhead, Cutthroat Trout, undefined trout species), as well as lamprey and sculpin. Further details about the mark-recapture methods are provided in Ewart and Kerr (2014).

For Coho Salmon, separate counts were recorded for wild and 'colonized' smolts. Colonized refers to fish that were incubated at the hatchery and transplanted to the upper Quinsam River watershed as fry. All transplanted Coho Salmon were marked with an adipose fin clip. The abundance of colonized Coho Salmon was estimated with the assumption that they have equal catchabilities as wild fish. Counts of wild and colonized Chinook Salmon were recorded in 2021.

Quinsam Hatchery staff have out-planted salmon fry during each year of JHTMON-8 (in addition to previous years; Table 6). During 2014-2020 approximately 181,524 Coho Salmon fry were released in the Upper Quinsam Lake (note that releases also occurred in years prior to 2014). Chinook Salmon fry were released in the Lower Quinsam Lake in 2015 for the first time in 10 years; during 2015, 2017, 2018, and 2019 approximately 200,000 fry were released, while ~150,000 Chinook Salmon fry were released in 2016. No Chinook Salmon fry were successfully released in 2020 due to COVID-19 restrictions. In 2021, approximately ~224,130 colonized Chinook Salmon fry were released from the hatchery into Lower Quinsam Lake on May 3–4 (Table 6).

Table 6. Number released and dates of release of Coho and Chinook Salmon fry in the Quinsam watershed.

Species	Life Stage	Waterbody	Year[1]	Number Released[2]
Coho Salmon	Fry	Upper Quinsam River	2020	139,570
			2019	181,524
			2018	159,336
			2017	139,570
			2016	146,547
			2015	167,030
			2014	157,661
Chinook Salmon	Fry	Lower Quinsam Lake	2021	224,130
			2020	0
			2019	207,736
			2018	215,952
			2017	207,319
			2016	147,549
			2015	217,603

[1] DFO annually reports the number of outplanted Chinook Salmon that same year, and the number of outplanted Coho Salmon outplanted the previous year, reflecting differences in the duration of the freshwater rearing period

[2] Coho Salmon are released between early April to early June; Chinook Salmon are typically released in May

2.1.3. Effects of Flow on Production of Juvenile Fish

As a supplementary task in Year 8 (2021), we carried out initial analyses to explore the potential effects of environmental variables on the production of juvenile fish to provide proof of concept for the general approach that we propose to take in Year 10 to test the study hypotheses (Section 1.4). For this preliminary analysis, we assessed if two explanatory variables of interest related to flow (maximum discharge during the incubation period and minimum discharge during summer) affect the functional relationship between stock size and resulting recruitment, thereby providing insight into H_04 (regarding floods; Section 1.5.5), but with the potential influence of hydrological variability considered more widely by also considering a low flow metric. These analyses will be updated and extended in Year 10 to support analysis to test the JHTMON-8 hypotheses. Using the analysis to examine effects on stock-recruitment relationships (e.g., rather than solely juvenile fish abundance), allows for interannual variability in spawner abundance to be accounted for, thereby isolating the potential effects of the environmental factors that are the subject of the study hypotheses, from potential effects due to the abundance of adult spawners (which will be considered explicitly when H_06 is tested).

The null hypotheses assessed here are that the production of juvenile salmonids is not affected by water discharge during specific life history periods. Specifically, the two key potential mechanisms through which discharge may affect the production of juveniles that were considered in the analysis were: i) high discharge during the incubation period may reduce incubation success due to redd scour or effects such as delayed redd construction and changes to spawning behaviour (Gibbins *et al.* 2008), and ii) low discharge during the summer period may reduce the availability and suitability of rearing habitats for juvenile fish, recognizing that lack of rearing habitat during the critical stream flow period in summer and early fall is a key limiting factor on Vancouver Island for stream-rearing species such as Steelhead (Ptolemy and Lewis 2002).

To calculate independent variables for the analysis, we analyzed mean daily discharge data collected at gauges maintained by Water Survey of Canada (Table 7) to calculate hydrological metrics (Table 8) using the using the Indicators of Hydrologic Alteration (Richter *et al.* 1996) package developed for R (R Core Team 2019). To evaluate the effect of high flows during the incubation periods (Table 1), we calculated the maximum discharge during the incubation periods of the study species (Table 8). To evaluate the effect of low flows during the rearing period, we calculated three low flow metrics (Table 8) that we evaluated, before selecting the 7-day minimum discharge as a suitable indicator of low flow conditions that we hypothesize could reduce the availability and suitability of rearing habitats for juvenile fish. We chose 7-days as the averaging period to represent a period that is of sufficient length to be relevant to rearing success, while sufficiently short for the metric to be responsive to short-term variability in flows among years, recognizing that the three low flow metrics are highly correlated.

The potential effect of high flows during the incubation period was examined for the three priority species (Section 1.5.1), and Pink Salmon and Chum Salmon. The abundance of outmigrating juvenile fish was related to incubation conditions assuming that Chinook Salmon, Pink Salmon, and Chum Salmon outmigrated aged 0+, whereas Coho Salmon outmigrated aged 2+ (assumptions regarding ages of Pacific salmon species are described in the Year 7 report; Suzanne *et al.* 2021). Steelhead smolts were assumed to be aged 2+ and confirmed by DFO (Fortkamp, pers. comm. 2022), consistent with the evaluation by Burt (2003) who concluded that this age class is most abundant, based on review of age data collected by Lirette *et al.* (1985). Steelhead labelled as "fingerlings" in the data provided by DFO were not included in the analysis as these fish were recorded separately from smolts and are assumed to represent resident trout or younger juvenile age classes that have yet to undergo smoltification.

The potential effect of low flows during the rearing period was examined for Coho Salmon and Steelhead, which are both stream rearing species that spend over one year rearing in freshwater before outmigrating (Burt 2003). The abundance of outmigrating juvenile fish of these two species was related to growing season flow conditions during the previous year, which was the final full year that the fish spent rearing in freshwater, when requirements for space and food are most likely to be limiting for a cohort. The analyses presented in this report build on the stock-recruitment analyses presented in the Year 7 report (Suzanne *et al.* 2021). Therein we assessed the fit of different formulations of the stock-recruitment relationship for two of the target species (Chinook Salmon and Coho Salmon), and two species of interest (Pink Salmon and Chum Salmon), whereas stock-recruitment curves were not fitted for Steelhead, as estimates of adult escapement are lacking, and we hypothesize that the expected low abundance of adult Steelhead limits the potential for the abundance of recruits to be limited by high density of spawners. The curves trialled in Year 7 can be broadly categorized as density independent and density dependent, whereby we fit the two most commonly used forms of density dependent curves: Ricker and Beverton-Holt. We found that the model that best described the stock recruitment relationship for Chinook Salmon was the density independent formulation, whereas one of the density dependent formulations best described the relationship for the other species analyzed. Readers should consult the Year 7 report (Suzanne *et al.* 2021) for further details of the stock-recruitment curves, including assumptions regarding the ages of outmigrating juvenile fish.

To assess how external factors affect the functional relationship between stock size and the resulting recruitment for the species with a density dependent stock recruitment function (Coho Salmon, Pink Salmon, and Chum Salmon), we followed Malick *et al.* (2017) and modeled salmon stock productivity as a function of spawner abundance using the standard Ricker model (Ricker 1954):

$$y_t = \alpha + \beta S_{t-j} + \epsilon_t$$

where y_t is the $\log_e$ of recruits per spawner, $\log_e(R_t/S_{t-j})$, where j varies depending on assumptions regarding the age that juvenile fish of each species out migrate, as described in Suzanne *et al.* (2021), α is the density independent stock productivity at low spawning stock sizes, β is the coefficient representing the strength of density dependence, and ε_t is the residual error term assumed to be normally distributed with mean 0 and variance σ_ϵ^2. Given the linear form of this equation, it is straightforward to estimate the effects of variables of interest on productivity by simply adding a term:

$$y_t = \alpha + \beta S_t + \gamma_{var1} var_1 + \epsilon_t$$

where var_1 is the environmental variable of interest, in this case either maximum discharge during the incubation period or 7-day minimum discharge during the growing season. Note that the discharge data were lagged according to the age of the outmigrating salmonids to ensure the data corresponded to the life history stage being examined.

In the case of Chinook Salmon, given that the stock recruitment function that best described the data was the density independent formulation, we fit the following linear regression:

$$R_t = \delta S_{t-1} + \gamma_{var1} var_1 + \epsilon_t$$

where δ is the average number of recruits produced per unit of stock.

We also assessed the effects of the environmental variables of interest on the number of Steelhead recruits:

$$R_t = \alpha + \gamma_{var1} var_1 + \epsilon_t$$

where α is the expected number of outmigrating smolts when the independent variable considered is naught.

All models were fit as general linear models within the R Statistical Language (R Core Team 2019), and statistical significance of the variables of interest were assessed at $\alpha = 5\%$. Regression residuals were visually inspected when evaluating model performance.

Table 7. Hydrometric gauges maintained by Water Survey of Canada on the Quinsam River. See Map 2 for site locations.

Site Name	Site Code	Period of Record		Position Relative to Diversion
		Start	End	
Quinsam R. at Argonaut Bridge	08HD021	1993	Ongoing	Downstream
Quinsam R. below Lower Quinsam Lake	08HD027	1997	Ongoing	Downstream
Quinsam R. near Campbell R.	08HD005	1956	Ongoing	Downstream

Table 8. Hydrological metrics calculated for the Quinsam River.

Hydrological Metric	Data Period
Max. discharge during Chinook Salmon incubation	15 Oct - 30 Apr
Max. discharge during Coho Salmon incubation	15 Oct - 22 Apr
Max. discharge during steelhead incubation	15 Feb - 15 Jun
Max. discharge during Pink Salmon incubation	15 Sep - 08 Apr
1-day minimum discharge	Calendar year
7-day minimum discharge	Calendar year
30-day minimum discharge	Calendar year

2.2. Water Quality

2.2.1. Water Chemistry

2.2.1.1. Quinsam River Water Chemistry Monitoring

One water quality site was established in the Quinsam River (QUN-WQ) in 2014 (Year 1) at 327433 E 5534757 N (UTM; Zone 10) and elevation 193 masl (Map 2). This site was selected based on guidance in the British Columbia Field Sampling Manual (Clark 2013) and the Ambient Fresh Water and Effluent Sampling Manual (RISC 2003), which require sites to be established in mid-stream locations that can be safely accessed and are located away from eddies where suspended particulate material can accumulate, potentially biasing results. QUN-WQ (Figure 2) is located ~950 m downstream of the confluence with the Iron River, and downstream of the Quinsam Coal Mine and the salmon carcass nutrient enhancement site. Water quality was sampled *in situ* using a YSI Pro Plus meter and by collecting samples that were shipped for laboratory analysis by ALS Environmental; sampling dates are provided in Table 9.

Water quality has been monitored during Year 1 through Year 8 at QUN-WQ, with monitoring scheduled to continue for the remainder of JHTMON-8. Each year, water quality has been monitored six times on a monthly basis from May through October. During all years, standard methods according to the procedures set out in the Guidelines for Designing and Implementing a Water Quality Monitoring Program in British Columbia (RISC 1997a) were employed to collect samples and measure *in situ* water quality parameters. Water chemistry variables were chosen based on provincial standards (Lewis *et al.* 2004).

The variables measured in Year 8 (2021) are presented in Table 10 (*in situ*) and Table 11 (laboratory). Laboratory method detection limits (MDL) for each analyte occasionally differed (Table 11) due to matrix effects in the sample, or variations in laboratory analytical instruments.

Table 9. Quinsam River water quality index site (QUN-WQ) sampling dates, Years 1 to 8.

Study Year	Sampling Dates
1	3-May-14; 18-Jun-14; 22-Jul-14; 19-Aug-14; 24-Sep-14; 04-Nov-14
2	12-May-15; 17-Jun-15; 23-Jul-15; 13-Aug-15; 16-Sep-15; 14-Oct-15
3	18-May-16, 15-Jun-16, 13-Jul-16; 17-Aug-16, 14-Sep-16; 12-Oct-16
4	10-May-17; 14-Jun-17; 12-Jul-17; 9-Aug-17; 13-Sep-17; 11-Oct-17
5	10-May-18; 05-Jun-18; 04-Jul-18; 09-Aug-18; 12-Sep-18; 05-Oct-18
6	13-May-19; 12-Jun-19; 11-Jul-19; 12-Aug-19; 12-Sep-19; 09-Oct-19
7	11-May-20; 08-Jun-20; 07-Jul-20; 10-Aug-20; 10-Sep-20; 08-Oct-20
8	13-May-21; 10-Jun-21; 08-Jul-21; 16-Aug-21; 16-Sep-21; 07-Oct-21

Figure 2. Looking downstream to QUN-WQ on September 16, 2021.

Table 10. Water quality variables measured *in situ* in Year 8.

Parameter	Unit
Water temperature	°C
pH	pH units
Salinity	ppt
Conductivity	µS/cm
Specific conductivity	µS/cm
Oxidation reduction potential	mV
Dissolved oxygen	mg/L
Dissolved oxygen	% Saturation

Table 11. Variables analyzed in the laboratory by ALS Environmental and corresponding units and method detection limit (MDL) in Year 8.

Parameter	Unit	MDL
General Water Quality		
Specific conductivity	µS/cm	2
pH	pH	0.1
Total suspended solids	mg/L	1
Total dissolved solids	mg/L	13 to 20
Turbidity	NTU	0.1
Alkalinity, Total (as $CaCO_3$)	mg/L	1
Nutrients		
Ammonia (as N)	µg/L	5
Nitrate (as N)	µg/L	5
Nitrite (as N)	µg/L	1
Total phosphorus	µg/L	2
Orthophosphate	µg/L	1

2.2.1.2. Quality Assurance/Quality Control

In situ water quality meters were maintained and operated following manufacturer recommendations. Maintenance included calibration, cleaning, periodic replacement of components, and proper storage. Triplicate *in situ* readings were recorded at each site on each sampling date.

For samples collected for laboratory analysis, sampling procedures and assignment of detection limits were determined following the guidelines of the BC Field Sampling Manual (Clark 2013) and the Ambient Fresh Water and Effluent Sampling Manual (RISC 2003). Duplicate samples were collected on each sampling date at the site.

Samples for laboratory analysis were collected in clean 500 mL plastic bottles provided by a certified laboratory (ALS Environmental). Samples were packaged in clean coolers that were filled with ice packs and couriered to the laboratory in Burnaby within 24 to 48 hours of collection. Standard Chain of Custody procedure was strictly followed. ALS Environmental performed in-house quality control checks including analysis of replicate aliquots, measurement of standard reference materials, and method blanks. A summary of the QA/QC laboratory results is provided in Section 4 of Appendix A.

In Year 8 (2021), one field blank and one trip blank were collected on May 13, 2021. Values for all parameters for both blanks were below the respective MDLs. Overall, for the JHTMON-8 sampling program on the Quinsam River, the total number of quality assurance/quality control (QA/QC) samples collected over eight years (26 out of 96 samples, or 27%) met or exceeded recommendations; the BC field sampling manual recommends that 20% to 30% of samples consist of QA/QC samples

(Clark 2013), whereas the RISC (1997a) manual recommends a minimum of 10% of samples consist of QA/QC samples.

In Vancouver Island streams, concentrations of several variables (notably nutrients) are commonly less than, or near to, the MDL. When this occurs, there are several different methods to analyze these values. In this report, any values that were less than the MDL were assigned the MDL values and averaged with the results of the other replicates. In these cases, the "real" average is less than the average reported.

2.2.1.3. Comparison with Guidelines for the Protection of Aquatic Life

Water quality guidelines for the protection of aquatic life (WQG-AL) and typical ranges of water quality variables in BC waters that were considered for this report are provided in Appendix A. Any results for water chemistry variables that approximated or exceeded WQG-AL, or ranges typical for BC, are noted in Section 3.1.3 of the Results.

For most water quality variables measured in this study, there are provincial WQG-AL. For total phosphorus, there are no provincial WQG-AL; however, there are federal guidelines (CCME 2004). For the remaining variables without provincial WQG-AL (i.e., orthophosphate, alkalinity, and specific conductivity) there are no federal guidelines either.

2.2.2. Water and Air Temperature

2.2.2.1. Quinsam River Temperature Monitoring

Water and air temperature monitoring was completed in Year 8 (2021) for the Quinsam River. Water temperature data have now been collected at the Quinsam River water quality index site for the period May 2014 to October 2021. Air temperature has also been measured near-continuously throughout this period.

Water temperature was recorded at intervals of 15 minutes using self-contained TidbiT v2 loggers (Onset, MA, USA). These TidbiT loggers had an operating range of -20°C to +70°C with an accuracy of ±0.2°C and a resolution of 0.02°C. Water temperature at the monitoring station was logged using duplicate TidbiT loggers installed on separate anchors. This redundancy is intended to prevent gaps in the data if one of the loggers malfunctions or is lost.

Air temperature was measured using one HOBO Air Temperature U23 Data Logger (range of -40°C to 70°C, accuracy of ±0.21°C) at the water quality index site (QUN-AT). The temperature logger recorded air temperature at a regular interval of 15 minutes. The logger was placed on a tree that was close (< 100 m) to the site.

2.2.2.2. Data Analysis

Water temperature data were analyzed as follows. First, erroneous data were identified and removed. Sources of erroneous data include occasional drops in water level that can expose the sensors to the atmosphere, and high flows which can move sediment and bury the sensors. Second, the records from duplicate loggers (when available) were averaged and records from different download dates were combined into a single time-series for the monitoring station. The time series for the station was then interpolated to a regular interval of 15 minutes, starting at the full hour.

Time series of water and air temperature data were plotted at 15-minute intervals; the hourly rates of change in water temperature were also plotted. Analysis of the water temperature data involved computing a range of summary statistics (Table 12) that were chosen based on the provincial WQG-AL (Oliver and Fidler 2001; Table 5 of Appendix A). The following statistics were computed: mean, minimum, and maximum water temperatures for each month of the record; hourly rate of change of temperature; days with mean daily temperature >18°C, >20°C, and <1°C; the length of the growing season, and; the accumulated degree days in the growing season. Statistics were based on the data collected at, or interpolated to, intervals of 15 minutes. Mean weekly maximum temperatures (MWMxT) were calculated and compared to optimum temperature ranges for different fish species and their life stages as outlined in the provincial WQG-AL (Oliver and Fidler 2001).

Table 12. Parameters calculated based on water and air temperature data.

Parameter	Description	Method of Calculation
Monthly water- and air-temperature statistics	Mean, minimum, and maximum on a monthly basis	Calculated from temperatures observed at or interpolated to 15-min intervals
Rate of water temperature change	Hourly rate of change in water temperature	Calculated observed or interpolated to 15-min intervals. The hourly rate of change is set to the difference between temperature data points that are separated over one hour.
Degree days in growing season	The beginning of the growing season is defined as the beginning of the first week that mean stream temperatures exceed and remain above 7°C; the end of the growing season was defined as the last day of the first week that mean stream temperature dropped below 7°C (modified from Coleman and Fausch 2007).	Daily mean water temperatures were summed over this period (i.e., from the first day of the first week when weekly mean temperatures reached and remained above 7°C until the last day of the first week when weekly mean temperature dropped below 7°C).
Number of Days of Extreme Daily Temperature	Daily temperature extremes for all streams	Total number of days with daily mean water temperature >18°C, >20°C, and <1°C
MWMxT (Mean Weekly Maximum or Minimum Temperature)	Mean, minimum, and maximum on a running centered weekly (7 day) basis	Mean of the warmest daily maximum or coldest daily minimum water temperature based on hourly data for 7 consecutive days; e.g., if MWMxT = 15°C on August 1, 2018, this is the mean of the daily maximum water temperatures from July 29 to August 4, 2018; this is calculated for every day of the year.

2.3. Invertebrate Drift

2.3.1. Sample Collection

One invertebrate drift sampling site was established on the Quinsam River (Map 2, Figure 3), located close (<150 m) to the water quality index site. The site location has been consistent among years; UTM coordinates (Zone 10) were: 327,361 E and 5,534,796 N. The site was located in riffle or run habitats (depending on flow), upstream of any obvious source of debris that could clog the nets or areas that seemed subject to frequent erosion. Invertebrate sampling was conducted monthly from May to October, with weekly sampling conducted during May in Year 8 (the month that is sampled weekly is rotated among study years to quantify the variance of monthly data). In total, sampling occurred on nine dates in the Quinsam River in Year 8 (2021) (Table 13).

Invertebrate drift sampling followed methods recommended in Hatfield *et al.* (2007) and Lewis *et al.* (2013). Upon arrival at site, local areas with velocities of approximately 0.2 m/s to 0.4 m/s were identified using a model 2100 Swoffer meter with a 7.5 cm propeller and a 1.4 m top-set rod. This range of velocities is ideal for sampling invertebrate drift as velocities are low enough to prevent clogging of the nets. Due to flow conditions at the time of sampling, it was not always possible to deploy the nets in areas with velocities of 0.2 m/s to 0.4 m/s (as per Hatfield *et al.* 2007), and nets sampled higher or lower water velocities at times.

Five drift nets were deployed simultaneously across the channel (Figure 3). The mouth of each drift net was positioned perpendicular to the direction of stream flow, and nets were spaced apart to ensure that each individual net did not obstruct flow into an adjacent net. The drift net mouth dimensions were 0.3 × 0.3 m and the nets (250 μm mesh) extended 1 m behind the mouth. Nets were anchored such that there was no sediment disturbance upstream of the net before and during deployment. All nets were deployed so that the top edge of the net was above the water surface so that invertebrate drift in the water column and on the water surface could be sampled.

At the start of sampling, measurements were made of water depth in each net and the water velocity by each net at the midpoint of the water column that was being sampled. These measurements were repeated hourly so that the volume of water sampled with each net could be calculated. Large debris (e.g., leaves) that entered the nets was periodically removed from the nets (after it had been washed of any invertebrates, which were returned to the nets). Nets were deployed for approximately four hours on each sample date (Table 13). Once the nets were removed, the contents of all five nets were transferred into sample jars (500 mL plastic jars with screw top lids) for processing as a single sample in Years 2–8. This detail differed from Year 1 (2014), when contents of each net were processed separately. Samples were preserved in the field with a 10% solution of formalin (formalin = 37–40% formaldehyde).

In Year 8, kick net sampling was also undertaken on September 16, 2021 at QUN-IV. The CABIN standardized sampling method was followed (Environment Canada 2012), with a single drift net (described above) used as a kick net. This required one crew member to hold the net flush with the stream bed immediately downstream of a second crew member undertaking the sampling. Sampling proceeded upstream for a timed period of three minutes, covering a horizontal distance of approximately 10 m. During sampling, the sampler kicked the substrate to disturb it to a depth of 5–10 cm, while also turning over any large cobbles or small boulders to dislodge invertebrates. Once sampling was complete, the contents were sieved (250 μm mesh), transferred into sample jars, and preserved in the same manner as drift net samples.

Table 13. Invertebrate drift sample timing and sampling duration at the Quinsam River site (QUN-IV) during Year 8.

Sample Date	Start Time[1]	Finish Time[2]	Sampling Duration[3,4] (hh:mm)
06-May-2021	07:14	11:17	4:03
13-May-2021	06:55	11:00	4:05
18-May-2021	06:58	10:58	4:00
25-May-2021	07:03	11:04	4:01
10-Jun-2021	06:43	10:43	4:00
08-Jul-2021	06:58	10:58	4:00
16-Aug-2021	07:15	11:15	4:00
16-Sep-2021	08:11	12:11	4:00
07-Oct-2021	08:20	12:20	4:00

[1] When the first net was set

[2] When the last net was removed

[3] The duration between retrieving the first and last net

[4] For data analysis, start and finish times for individual nets were used to calculate the volume of water filtered for each net

Figure 3. View of invertebrate sampling drift nets across the stream from river right towards QUN-IV, July 08, 2021.

2.3.2. Laboratory Processing

Samples were sent to Ms. Dolecki of Invertebrates Unlimited in Vancouver, BC for processing. Ms. Dolecki is a taxonomist with Level II (genus) certification for Group 2 (Ephemeroptera, Plecoptera, and Trichoptera (EPT)) and for Chironomidae from the North American Benthological Society.

The drift and kick net samples were first processed by removing the formalin (pouring it through a 250 µm sieve), followed by immediate picking and identification of the very large and rare taxa. Samples were split into subsamples if the number of invertebrates was over 1,000. The invertebrates were enumerated using a Leica stereo-microscope with 6 to 8 × magnification, with additional examination of crucial body parts undertaken at higher magnifications (up to 400 ×) using an Olympus inverted microscope where necessary. Individuals from all samples were identified to the highest taxonomic resolution possible and it was noted whether a taxon was aquatic, semi-aquatic, or terrestrial. Life stages were also recorded.

Digitizing software (Zoobbiom v. 1.3; Hopcroft 1991) was used to measure the length of a sub-sample of individuals. Length measurements were then used to calculate average biomass (mg dry weight) of each taxon using standard length–weight regressions. The regressions were developed using un-preserved individuals and therefore the estimates are unaffected by reduction in biomass that can occur due to preservation in alcohol and subsequent drying of tissues inside carapaces (the length measurements are unaffected by preservation). This method is considered more accurate than weighing the invertebrates because it is not influenced by loss of biomass caused by preservation or the presence of debris and does not require invertebrates to be dried. For abundant taxa, up to 25 randomly chosen individuals per taxon were digitized to address the variability in size structure of the group. For the rare taxa, all individuals in the taxon were measured. The damaged or partial specimens were excluded from the measurements. For pupae and emerging Chironomidae, up to 50 individuals were measured.

To provide QA/QC, all the samples were re-picked a second time to calculate the accuracy of picking. This assured that > 90% accuracy was attained, and the accuracy of the methods employed is expected to be over 95%.

2.3.3. Data Analysis

Variables were chosen and calculated as per Lewis *et al.* (2013), and all taxa (aquatic, semi-aquatic, and terrestrial) were considered. Density (# of individuals per volume), total biomass (mg dry weight) and the sum of EPT (Ephemeroptera, Plecoptera, and Trichoptera) biomass (mg dry weight) of each sample were expressed as units per m^3 of water, whereby volume is the amount of water that was filtered through a single net during a set. Volume filtered by each net was calculated based on the duration that the nets were deployed, and the average discharge measured at each net. EPT biomass was calculated because EPT taxa are expected to comprise an important part of salmonid diets in the Quinsam River. Calculation of EPT biomass was an additional task initiated in Year 7 with the aim to calculate invertebrate metrics that are best suited to test H_05. As agreed with BC Hydro, the addition of this new task was offset by assigning less effort to analysis of invertebrate community composition, which is considered peripheral to testing H_05, which concerns food availability.

During Years 2–8, the analysis was undertaken for each combined sample that included the contents of all five nets. For Year 1 (when net samples were not physically combined), data for each net were combined into site-level samples prior to calculating biodiversity metrics (family richness, Simpson's diversity) so that results were directly comparable with the results for Years 2–8. Family richness and Simpson's diversity are both standard metrics used to quantify invertebrate biodiversity. Change in these metrics may indicate change in the quality of food available to rearing fish.

Family richness (i.e., the number of families present) was calculated for each sample as a metric of biodiversity. Simpson's diversity index (1-λ; Simpson 1949) was calculated from family level density data to provide a measure that reflects both richness and the relative distribution or 'evenness' of invertebrate communities (i.e., higher Simpson's diversity index values denote communities that have high family richness, with the total number of individuals also evenly distributed among families). The index value ranges between 0 (no diversity) and 1 (a hypothetical scenario of infinite diversity). A Simpson's diversity index closer to 1 is associated with greater diversity and, thus, potentially greater food quality for fish.

The Canadian Ecological Flow Index (CEFI) was calculated using family level data for aquatic taxa following Armanini *et al.* (2011). Taxa present in <5% of the samples were not excluded from the CEFI calculation (Armanini, pers. comm. 2013). Relative abundances of taxa at the site were calculated considering only aquatic taxa, and only aquatic taxa used to develop the CEFI were considered when calculating the index. The top five families contributing to biomass at the site on each date were also identified.

3. RESULTS

3.1. Fish Population Assessments

3.1.1. Quinsam River Salmon Escapement, 2020

Salmon escapement data for the Quinsam River are presented for 2020 (Year 7; Table 14), which are the most recent results available at the time of reporting. Summary statistics for the period of record are also provided in Table 14 to provide points of reference; Figure 4 presents salmon escapement data for the period of record.

Pink, Coho and Chinook salmon were the most abundant returning species in 2020, as well as historically (Table 14). Escapement of Chinook Salmon in the Quinsam River in 2020 (8,236) was above-average (4,320), although the values in the late 2010s were lower than the values observed in the late 1980s, early 1990s and early 2000s. Estimated escapement of Coho Salmon (12,721) in 2020 was slightly higher than the mean value (12,157) for the period of record (1957–2019); the values estimated during the last decade are generally higher than those observed between the late 1950s and late 1970s, but lower than those observed between the early 1980s and early 2000s. The estimated Chum Salmon escapement (20) was particularly low[3]; it was the 4th lowest count recorded in the 61 years for which there are counts, with the count in 1993 (6) the lowest count. Pink Salmon escapement in the Quinsam River in 2020 (513,567) was higher than the mean value (136,840) for the period of record (1957–2019). The estimated escapement of Sockeye Salmon in 2020 (3) was the 2nd lowest count recorded during 49 years of records (equal to 2010 and 2012), while the count in 2019 (2) was the lowest count; for further context, the annual escapement values estimated during the last two decades (range from 2 to 25) are generally lower than those observed between 1970–2000 (range from 6 to 691; few data are available prior to 1970).

During the seven years of available data for the JHTMON-8 study period, a notable result was the occurrence of a record high Pink Salmon escapement (1.42 million) in Year 1 (2014). Escapement of Chinook Salmon (a priority species) in the Quinsam River increased steadily over the first four years from 2,366 fish to 9,131 fish, decreased in 2018 and 2019 to 6,774 and 6,793 fish, respectively, and increased in 2020 to 8,236 fish. By contrast, escapement of Coho Salmon (also a priority species)

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[3] Note that the end of the Chum Salmon sampling period (November 18; Table 4) was ~4 weeks prior to the end of the defined migration period (Table 1) and therefore this value is expected to be an underestimate. Nonetheless, the sampling period spanned the majority of the migration period and the end date of sampling was within the range of dates monitored in previous years. Thus, it is appropriate to conclude that Chum Salmon returns to the Quinsam River were low in 2020 relative to returns in other years, although total escapement is expected to be greater than the reported value. Note that DFO records salmon escapement to the Campbell River (downstream) separately; Chum Salmon escapement to the Campbell River in 2020 was 4,000 fish (DFO 2021).

decreased steadily over the first four years from 14,883 fish to 5,865 fish and increased in subsequent years to 12,721 fish in 2020.

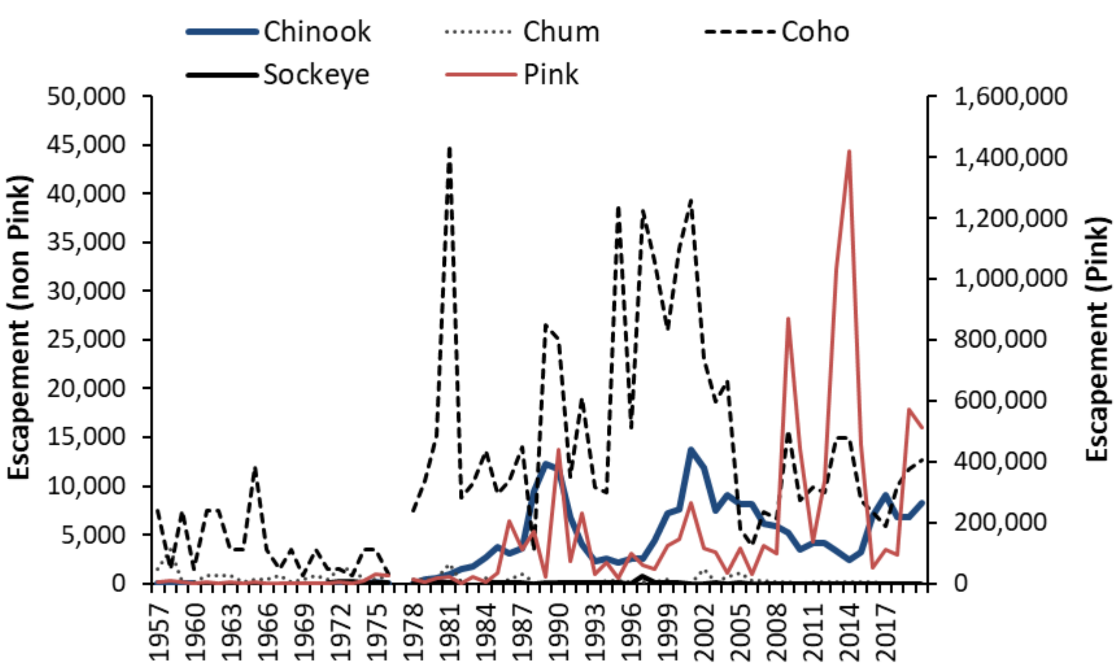
Table 14. 2020 salmon escapement data for the Quinsam river (DFO 2021).

Statistic	Salmon Species				
	Chinook[1]	Chum	Coho[1]	Pink	Sockeye
2020 count	8,236	20	12,721	513,567	3
Mean (1957-2019)	4,320	472	12,157	136,840	52
Median (1957-2019)	3,431	255	9,310	31,995	23
10th percentile (1957-2019)	35	52	1,500	1,500	6
90th percentile (1957-2019)	9,395	1,458	31,077	442,989	128
Percent of years sampled (1957-2019)[2]	81	95	98	98	76

[1] Priority species for JHTMON-8

[2] "Percent of years sampled" is approximate; uncertainty in data recording means that a count of zero is not always distinguished from a record of "not measured"

Figure 4. Salmon escapement for the Quinsam River (1957–2020; DFO 2021). Note the different scale for Pink Salmon.



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3.1.2. Quinsam River Hatchery Salmon Counting Fence Operations

Data collected at the salmon counting fence are summarized in Table 15. Following installation on March 8, the traps were monitored daily from March 9 to June 18.

The monitoring period provided good coverage of the Pink Salmon fry migration period in 2021, although a low number (273) of fry was captured on the first day of sampling, suggesting that the migration period started slightly prior to March 9. The migration was seemingly complete by May 13 as no Pink Salmon fry were captured after that date. Total estimated migration of Pink Salmon fry has been highly variable in the eight years of the monitoring program and was ~11 million in 2021 (Year 8) (Table 15). Estimates varied by an order of magnitude among years since 2014, ranging from a minimum of 1.5 million fry in 2017 to a maximum of 22 million fry in 2014.

Total outmigration estimates for the three JHTMON-8 priority species in the Quinsam River (Coho Salmon smolts, Steelhead smolts, and Chinook Salmon fry) are presented for the JHTMON-8 period in Figure 5. To provide broader context, outmigration estimates of priority species are presented in Figure 6 for the full period of record (since the mid 1970s), based on a data compilation exercise undertaken in Year 5 (Abell *et al.* 2019). Annual values presented in Figure 6 are considered directly comparable, although there was some variability in sampling methods among years that contributes to variability in sampling error. Readers should consult the historical data review undertaken in Year 5 (Abell *et al.* 2019) and the review of capture efficiency estimates undertaken in Year 6 (Suzanne *et al.* 2020) for further details.

In Year 8 (2021), total estimated outmigration of colonized Coho Salmon (28,004) was the third lowest recorded during JHTMON-8. Total estimated outmigration of wild Coho Salmon (23,339) in 2021 was low among the eight years, with the highest recorded in Year 7 (57,244). A small number of wild Coho Salmon (i.e., 12 fish captured corresponding to a total estimate of 152 fish) was recorded outmigrating on the last day of trapping (June 18). The total estimated outmigration of Steelhead smolts (6,609; 522 fish captured) in 2021 was relatively low (~50% of the estimate for Year 7), although there is uncertainty regarding the accuracy of Steelhead smolt outmigration estimates as capture efficiency is based on mark-recapture experiments undertaken with Coho Salmon, which may not be well-representative of Steelhead smolt catchability (see Abell *et al.* 2019 for further discussion of sources of uncertainty). Estimated outmigration of wild Chinook Salmon in 2021 (269,022) was the 2nd highest value recorded during the eight years of JHTMON-8, during which estimated Chinook Salmon outmigration has been highly variable. Chinook Salmon fry were noted to still be outmigrating on June 18 when the traps were removed, with 147 captured on the final day of sampling. Estimated outmigration of colonized Chinook Salmon (188,609) in 2021 was the highest value recorded during the eight years of JHTMON-8. Estimated outmigration of all priority species during JHTMON-8 has been within the range of historical estimates for the study, with the exception of wild Chinook Salmon in 2020, which is the highest value recorded in the dataset (Figure 6).

The survival of out-planted juvenile salmon was estimated by calculating the percentage of outmigrating juvenile colonized salmon that comprise the total number of fish out-planted, as shown in Figure 7, which also shows estimated survival for years prior to the start of JHTMON-8, based on an additional task completed in Year 5 of JHTMON-8 (Abell *et al.* 2019). After a break of approximately 10 years, Chinook Salmon out-planting operations resumed in 2015, and therefore estimates of survival rate are available for 2015–2019 and 2021 (no Chinook Salmon were out-planted in 2020). Estimated survival of colonized juvenile Chinook Salmon during JHTMON-8 was highest in 2021 and has varied between 65% and 84% during five of the six years, with a lower value (28%) estimated in 2016. Colonized juvenile Coho Salmon survival estimates are available for seven years of monitoring, ranging between 13% and 36%, with survival generally lower than for Chinook Salmon, at least partly reflecting that this Coho Salmon spend longer in freshwater. The survival estimate for Coho Salmon in 2020 (36%) was the highest during the seven years for which estimates are available for JHTMON-8. Note that the estimates for Coho Salmon assume that fish outmigrate at age 1+ (no 2+ smolts were observed at the fence in 2021[4]). Thus, the Coho Salmon survival estimate in 2020 (for example), is based on dividing the estimated smolt outmigration in 2021 by the number of hatchery fry released in 2020.

[4] Burt (2003) suggests that 2+ smolts (observed in some years) represent fish that were trapped in off-channel habitats, preventing them from outmigrating the previous year.

Table 15. Summary of downstream migration data and total migration estimates from sampling at the Quinsam River Hatchery salmon counting fence, March 9 to June 18, 2021.

Species	Life Stage	Total Counts	Total Estimated Migration[1]	Peak Migration	Migration Period
Colonized Coho	Smolt	2,212	28,004	15-May-21	26 Apr-18 Jun
Wild Coho	Smolt	1,841	23,339	15-May-21	06 Apr-18 Jun
2 Year old Coho	Smolt	0	-	n/a	n/a
Coho	Fry	5,375	170,750	20-Apr-21	09 Mar-18 Jun
Steelhead	Smolt	522	6,609	14-May-21	24 Apr-31 May
Steelhead	Fingerling	15	190	14-May-21	25 Apr-28 May
Steelhead	Kelts	0	-	n/a	n/a
Cutthroat	Fingerling	12	152	14-May-21	03 May-15 Jun
Cutthroat	Smolt	11	139	15-May-21	24 Apr-09 Jun
Cutthroat	Kelts	14	177	05-May-21	02 May-16 Jun
Trout Fry	Fry	1	13	23-May-21	23 May-23 May
Chinook	Fry	19,595	269,022	16-May-21	29 Mar-18 Jun
Colonized Chinook	Fry	14,853	188,609	18-May-21	06 May-18 Jun
Chum	Fry	149	6,718	24-Mar-21	13 Mar-21 Apr
Sockeye	Fry	1	45	18-Apr-21	18 Apr-18 Apr
Pink	Fry	242,875	10,942,050	11-Apr-21	09 Mar-13 May
Dolly Varden	Smolt	0	-	n/a	n/a
Lamprey (2 species)	all	82	1,078	May 1,2,16	29 Mar-18 Jun
Sculpin	all	63	1,025	01-May-21	16 Mar-15 Jun

[1] Based on capture efficiency measured for Pink Salmon and Coho Salmon
"n/a" indicates no peak or migration period identified

Figure 5. Total estimated outmigration of priority species on the Quinsam River during Years 1–8 (2014–2021). Coho Salmon and Steelhead were captured at the smolt stage and Chinook Salmon at the fry stage.

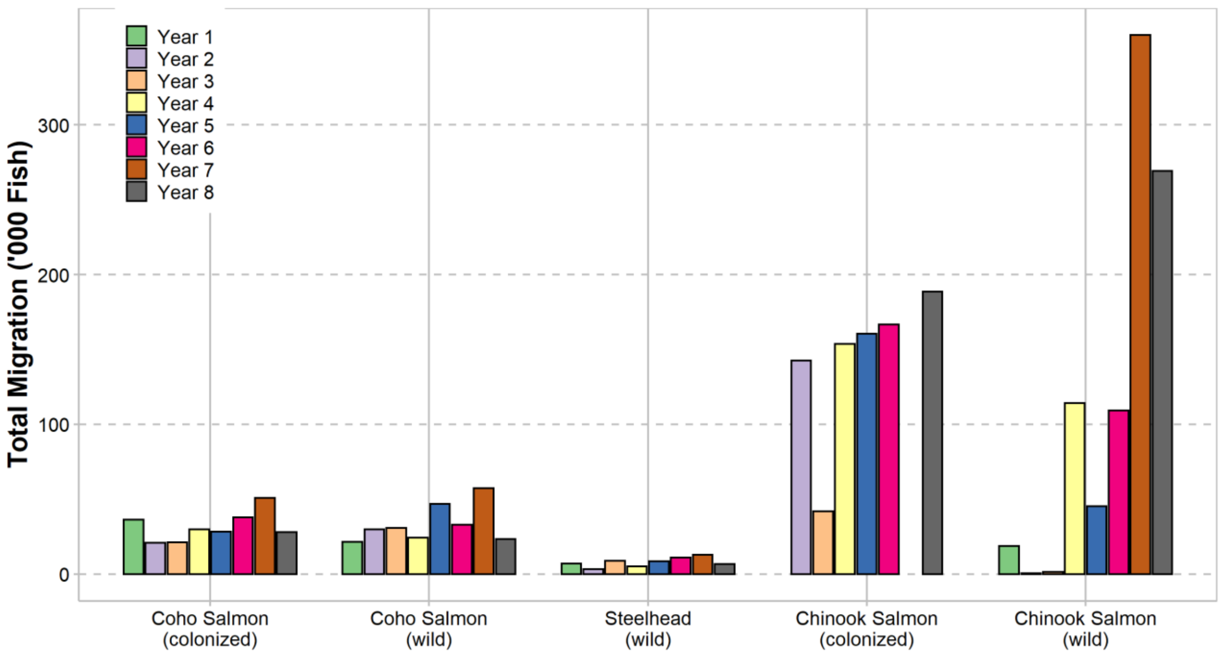


Figure 6. Estimated outmigration of priority species in the Quinsam River during 1979-2021, distinguished between colonized and wild fish. Coho Salmon and Steelhead were captured at the smolt stage and Chinook Salmon at the fry stage (0+).

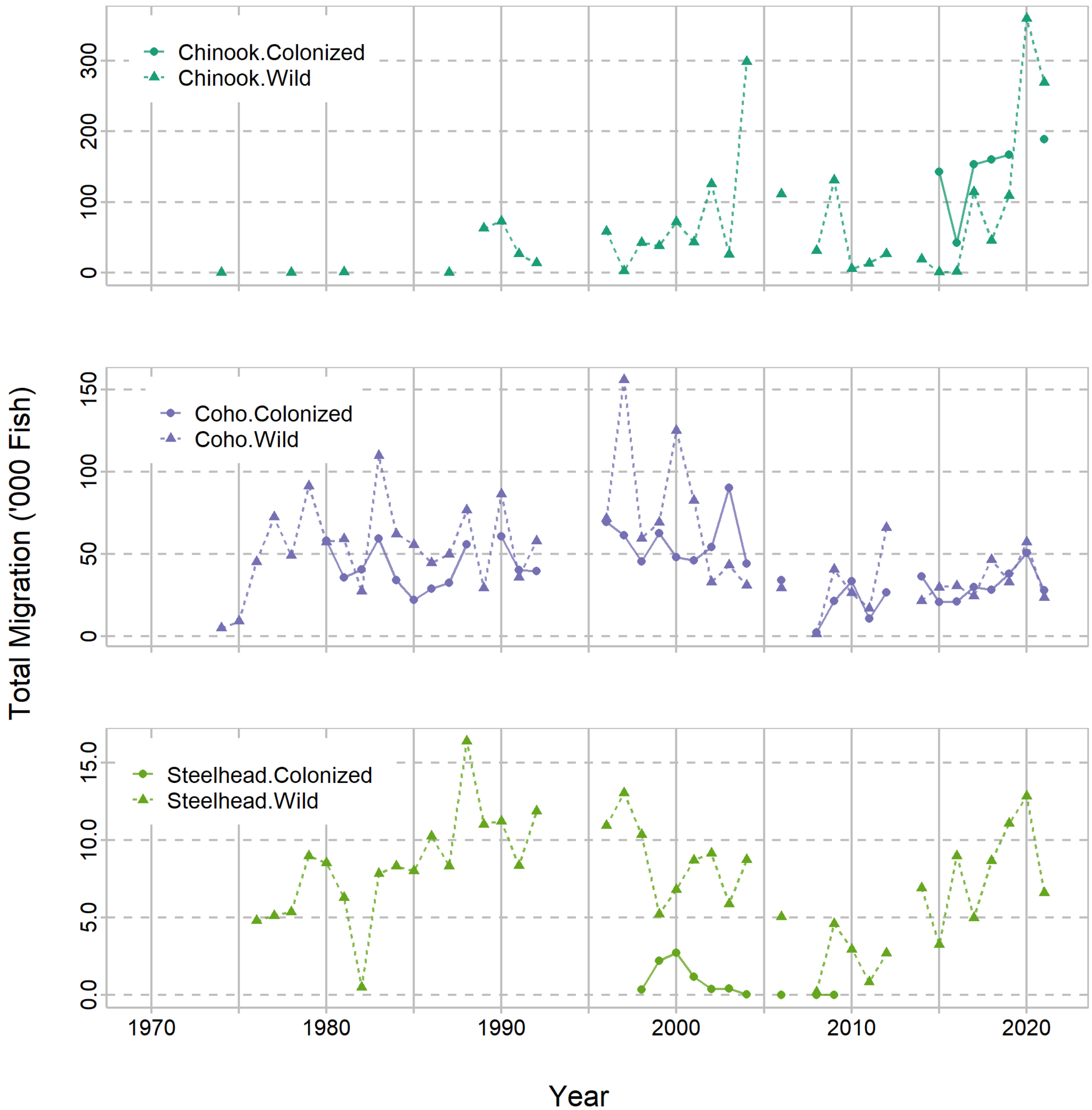
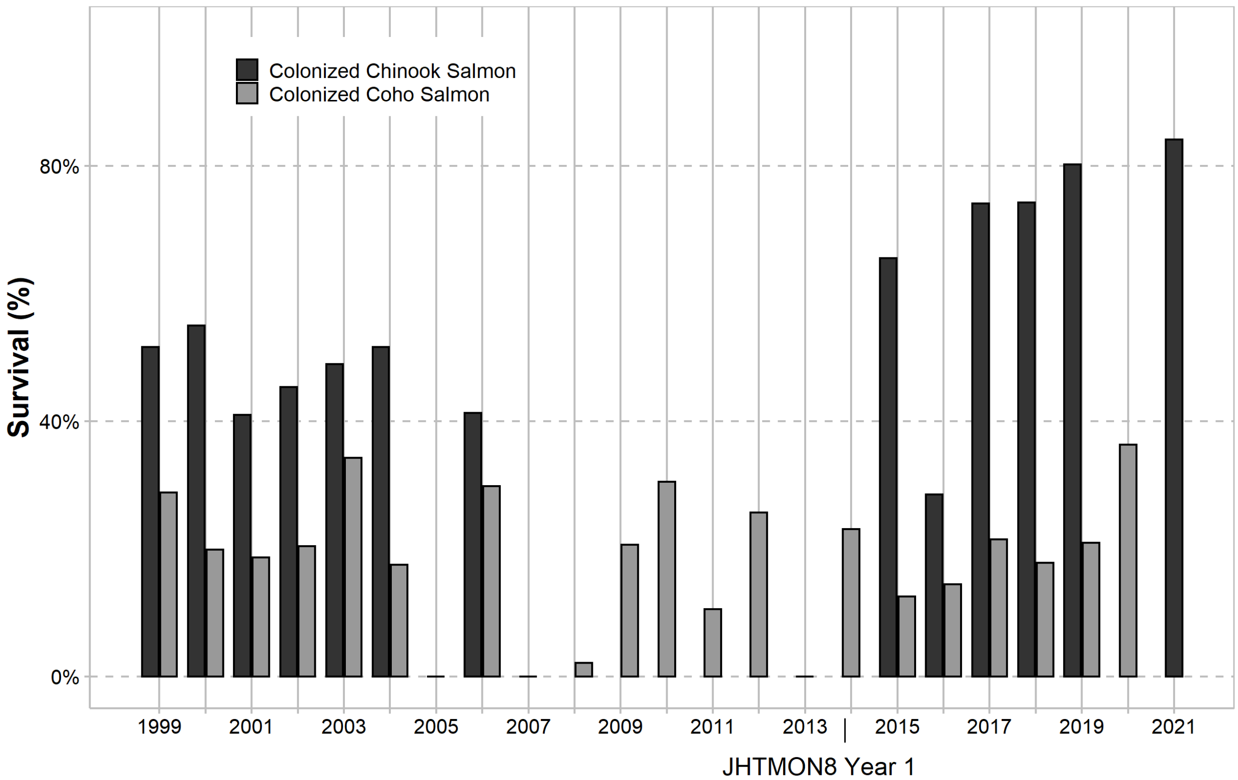


Figure 7. Estimated survival of out-planted salmon raised at the hatchery, based on the proportion of out-planted fish estimated to outmigrate at the salmon counting fence. Estimates correspond to the year of release; Chinook Salmon outmigrate during the year of release, whereas Coho Salmon are assumed to outmigrate during the year following release.



3.1.3. Effects of Flow on Production of Juvenile Fish

Based on the pre-defined threshold for statistical significance (α=5%, i.e., p <0.05), only one test was statistically significant: the maximum discharge during the incubation period negatively affected the recruitment of Pink Salmon (odd year spawning stock only; p=0.037, $\widehat{\gamma_{maxflow}} = 0.02$ (95% CI: - 0.03 - 0; Table 16). No relationship was found between the low flow metric and the abundance of the two stream-rearing species analyzed.

The Ricker stock recruitment model, modified with the inclusion of maximum discharge during incubation, provided a parsimonious description of Pink Salmon recruitment, whereby the level of recruitment and compensation at high stock levels decreases as a function of maximum discharge (Figure 8). This effect of discharge is consistent with an interpretation that redd scouring by high discharge resulted in depressed recruitment for this stock (discussed further in Section 4.5).

Given that estimates of Steelhead escapement were lacking, we fitted a linear regression to test the effect of discharge on the recruitment of Steelhead (Section 2.1.3) and found that the maximum discharge during incubation had a weak positive effect on the level of recruitment (Figure 9), although the statistical significance of the relationship (*p*=0.064) was slightly lower than the predefined threshold. Accordingly, it is appropriate to retain the null hypothesis and attribute the weak positive relationship to chance. Nonetheless, the result suggests that this relationship warrants closer scrutiny in Year 10 when data for two additional years will be available, potentially increasing statistical power. Although a positive relationship between recruitment and this high flow metric is inconsistent with our *a priori* hypothesis that high discharge may reduce recruitment success, there are potential mechanisms to explain a positive effect of high discharge, as discussed further in Section 4.5.

Table 16. Magnitude and statistical significance of the parameters of the relationships between discharge and production of five species of juvenile Pacific salmonids. Statistically significant results at the α = 5% level are highlighted in red, and significant results at the α = 10% level are highlighted in orange.

Species	Model	Environmental variable	Parameter value (95% CI)	*pvalue*
Chinook	Density independent	Maximum flow during incubation period	-73.3 (-575.3, 428.7)	0.77
Coho	Ricker	Minimum flow during summer	0.16 (-0.48, 0.79)	0.62
		Maximum flow during incubation period	0.000925 (-0.01, 0.01)	0.85
Chum	Ricker	Maximum flow during incubation period	0.000479 (-0.02, 0.02)	0.96
Pink Even Years	Ricker	Maximum flow during incubation period	-0.0093 (-0.03, 0.02)	0.44
Pink Odd Years	Ricker	Maximum flow during incubation period	-0.02 (-0.03, 0.00)	0.037
Steelhead	Linear regression	Minimum flow during summer	649.9 (-2162.6, 3462.4)	0.646
		Maximum flow during incubation period	71.9 (-4.3, 148.3)	0.064

Figure 8. **Influence of stock abundance and maximum discharge during the incubation period on recruitment of Pink Salmon (odd year spawning stock) in the Quinsam River. Colours indicate the expected number of recruits, red colours indicate relatively high values, and blue colours indicate relatively low values.**

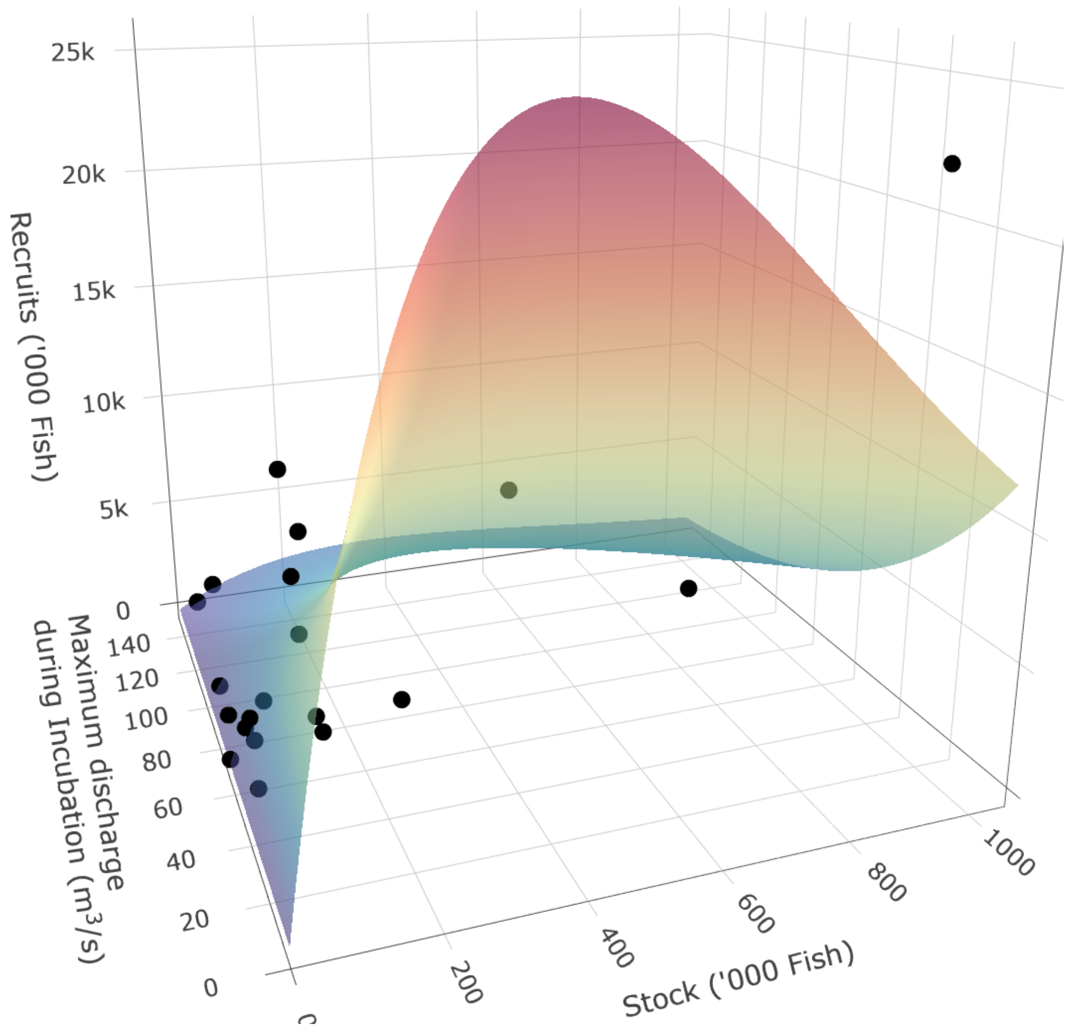
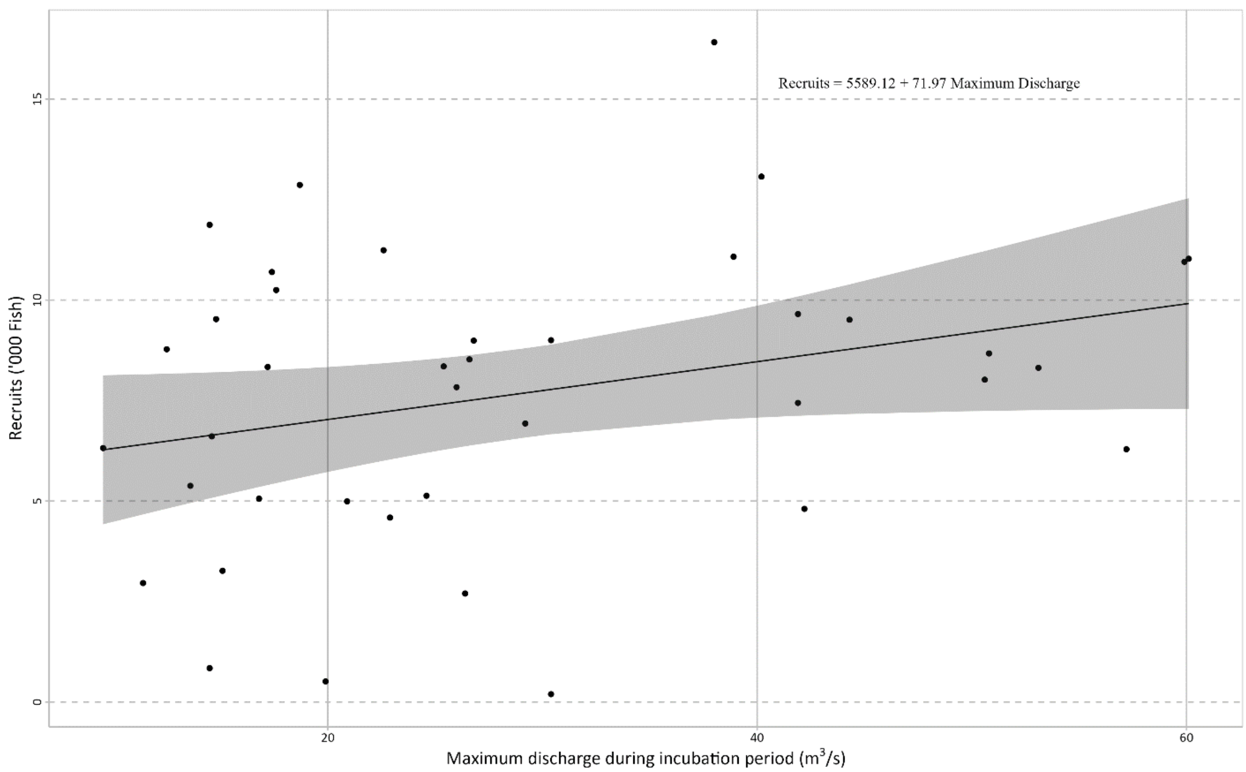


Figure 9. Influence of maximum discharge during the incubation period on outmigration of Steelhead smolts in the Quinsam River.



3.2. Water Quality

3.2.1. QA/QC

All laboratory analyses were conducted within the recommended hold times (see Table 17 of Appendix A), with the exception of turbidity analysis of the sample collected on June 10, 2021 and all pH values. The hold time for the turbidity sample was four days, thus exceeding the recommended hold time of three days. The turbidity measurements collected on June 10, 2021 are within historical ranges, the magnitude of the exceedance is minor, and the samples were well preserved (immediately placed on ice and kept cool); therefore, no substantive effect on data quality is anticipated.

All pH measurements from QUN-WQ that corresponded to laboratory analysis exceeded the recommended hold time of 0.25 hours, as occurred in all previous years and is inevitable given the sampling location. Both laboratory and field data for pH are presented in the following sections.

Clark (2013) and RISC (2003) recommend that results for duplicate samples should have relative percent difference or relative standard error values of 20% or less (provided that the concentrations are greater than five times higher than the MDL), otherwise it can indicate a potential issue with the sample. Contamination is suspected when the relative variability between duplicates exceeds 50% (Clark 2013).

In 2021, considering only parameters with concentrations five times higher than the MDL, the relative standard error threshold of 20% was exceeded for duplicate alkalinity (as $CaCO_3$) measurements corresponding to samples collected on October 7, 2021 (relative standard error of 27.5%). It is unlikely that the high variability in the alkalinity measurement for this set of duplicates was due to contamination of the sample since values for other parameters measured in the same samples do not show high variability. The alkalinity measurements in both duplicate samples were relatively low and are within historical ranges; accordingly, the high variability for these duplicates does not affect our ability to test the applicable hypothesis ($H_0 3$).

One field and one trip blank were collected in 2021. Values for all parameters were below the respective MDLs for both blanks, indicating that contamination was avoided during sampling. Values of pH were slightly higher in the trip blank (5.50) than the field blank (5.41), with the difference in values within the range observed in previous years (Table 18 of Appendix A).

3.2.2. Field Measurements

A summary of Year 8 (2021) water quality results for the Quinsam River (QUN-WQ) are presented in Table 17, and the range of values for water quality variables for Year 8 are compared to ranges in previous years in the monitor (Years 1 to 7) and to typical ranges in BC waterbodies.

In addition, the following sections provide further discussion of each water quality variable. The Year 8 laboratory and *in situ* water chemistry results for the Quinsam River at QUN-WQ are summarized in Table 18 (general variables measured at ALS laboratories), Table 19 (general variables measured *in situ*), Table 20 (dissolved oxygen (DO) measured *in situ*), and Table 21 (low level nutrients measured at ALS laboratories). Combined results from Years 1 to 8 (2014 to 2021) of water quality monitoring are tabulated in Section 2 of Appendix A.

Table 17. Summary of water quality Year 8 (2021) measurements in the Quinsam River (QUN-WQ), compared to Years 1-7 (2014-2020) values and typical ranges in BC waterbodies.

Water Quality Variable	Units	Year 8 Range	Years 1–7 Range	Typical Range for BC Waters	Comments
Alkalinity (as $CaCO_3$)	mg/L	32.2 to 53.0	23.5 to 54.0	Natural waters almost always have concentrations less than 500 mg/L; waters in coastal BC typically range from 0 to 10 mg/L and waters in interior BC can have values greater than 100 mg/L (RISC 1997a).	Alkalinity is consistently > 20 mg/L, indicating low sensitivity to acidic inputs.
pH	pH units	6.86 to 7.97	5.92 to 8.05	Natural fresh waters have a pH range from 4 to 10; BC lakes tend to have a pH ≥ 7.0 and coastal streams commonly have pH values of 5.5 to 6.5 (RISC 1997a).	pH is typical of BC waters.
Specific conductivity	µS/cm	120.0 to 236.0	69.4 to 206.0	~100 µS/cm for coastal BC streams (RISC 1997a).	Specific conductivity is higher than typical coastal BC streams; possible influence of two upstream lakes.
Turbidity	NTU	0.23 to 0.51	0.23 to 1.19	In BC natural concentrations of suspended solids vary extensively from waterbody to waterbody and can have large variation within a day and among seasons (Singleton 1985).	Turbidity is low, indicating high water clarity.
Total suspended solids	mg/L	<1.0 to 1.2	<1.0 to 2.4	In BC natural concentrations of suspended solids vary extensively from waterbody to waterbody and can have large variation within a day and among seasons (Singleton 1985).	TSS is low, indicating high water clarity.
Dissolved oxygen	mg/L	7.94 to 10.80	6.99 to 11.75	In BC surface waters are generally well aerated and have DO concentrations >10 mg/L (MOE 1997).	DO (mg/L) below the most conservative provincial guideline (DO instantaneous minimum of 9 mg/L) for the protection of buried embryos/alevins has routinely been measured but is not expected to affect fish.
	% saturation	82.7 to 96.8	76.6 to 103.0	In BC surface waters are generally well aerated and have DO concentrations close to equilibrium with the atmosphere (i.e., close to 100% saturation) (MOE 1997).	DO (%) is consistently close to 100% saturation, indicating generally well-oxygenated waters.
Total Ammonia (as N)	µg/L	<5 to 10.6	<5 to 24.5	<100 µg/L for waters not affected by waste discharges (Nordin and Pommen 2009).	Ammonia is low, and well below the WQG-AL.
Orthophosphate (as P)	µg/L	<1 to 1.6	<1 to 2.1	Coastal BC streams typically have concentrations <1 µg/L (Slaney and Ward 1993; Ashley and Slaney 1997).	Orthophosphate is low, and typical of coastal BC streams.
Nitrate (as N)	µg/L	6.9 to 38.0	7.1 to 47.8	In oligotrophic (low productivity) lakes and streams, nitrate concentrations are expected to be <100 µg/L; in most streams and lakes not impacted by anthropogenic activities, nitrate is typically <900 µg/L (Nordin and Pommen 2009).	Nitrate is low, indicative of an oligotrophic river.
Nitrite (as N)	µg/L	<1	<1 to 1.5	Due to its unstable nature, nitrite concentrations are very low, typically present in surface waters at concentrations of <1 µg/L (RISC 1997b).	Nitrite is very low, and typical of surface waters.
Total phosphorus (P)	µg/L	<2 to 6.3	<2 to 7.4	Oligotrophic (low productivity) water bodies have total phosphorus concentrations that are between 4 to 10 µg/L, while concentrations are typically between 10 to 20 µg/L in mesotrophic water bodies. Total phosphorus can vary seasonally and with turbidity and TSS (CCME 2004).	Total phosphorus is low, indicative of an oligotrophic river.

Alkalinity

Alkalinity (as $CaCO_3$) measured at ALS laboratories ranged from 32.2 mg/L (May) to 53.0 mg/L (October; Table 18) in 2021, similar to previous years. Alkalinity concentrations were consistently greater than 20 mg/L, indicating that the Quinsam River has low sensitivity to acidic inputs (RISC 1997b).

pH

pH values measured in the laboratory in Year 8 ranged from 7.77 to 7.97, while *in situ* pH ranged from 6.86 to 7.59 (Table 18 and Table 19, respectively). Natural fresh waters typically have a pH range from 4 to 10; BC lakes tend to have pH ≥ 7.0, and coastal streams commonly have pH values of 5.5 to 6.5 (RISC 1997b). The pH values measured *in situ* are expected to be more accurate than the laboratory pH, given that the pH measurements for the laboratory samples exceeded the recommended hold time.

Specific Conductivity and Total Dissolved Solids

Laboratory values for specific conductivity (conductivity normalized to 25°C) in Year 8 ranged from 123.0 µS/cm (May) to 236.0 µS/cm (July; Table 18), which ranged slightly higher than previous years. *In situ* specific conductivity measured in Year 8 ranged from 120.0 µS/cm (May) to 235.0 µS/cm (July; Table 19). Coastal BC streams generally have specific conductivity of ~100 µS/cm (RISC 1997b). Most specific conductivity values in the Quinsam River were higher than typical levels in coastal streams. This may reflect the influence of primary productivity in the two lakes upstream of the monitoring site. Alternatively, high values of specific conductivity measured in the past have previously been linked with coal mining activities in the watershed (Redenbach 1990, cited in Burt 2003).

Total dissolved solids measured in the laboratory for the Quinsam River ranged from 76 mg/L (May) to 143 mg/L (July; Table 18) in Year 8 (2021).

Turbidity and Total Suspended Solids (TSS)

Turbidity in the Quinsam River at QUN-WQ was low in all eight monitoring years, indicating high water clarity (values in Year 8 ranged from 0.23 NTU to 0.51 NTU; Table 18). Similarly, TSS concentrations in Year 8 were low and consistent with previous years, with values ranging from below the MDL of 1.0 mg/L to slightly above the MDL (1.2 mg/L).

Dissolved Oxygen

Concentrations of DO in the Quinsam River were highest in May and October 2021 (when water temperatures were coolest; Table 20), when average DO concentrations were 10.60 mg/L and 10.80 mg/L, respectively. During June, July, and August 2021 sampling, DO measurements were lowest and the average DO concentration did not meet the more conservative provincial WQG-AL (DO instantaneous minimum of 9 mg/L) for the protection of buried embryos/alevins (Table 20; MOE 1997). The measurement in June (average of 8.93 mg/L on June 10, 2021; Table 20) indicate that the 9 mg/L WQG-AL was not achieved during part of the incubation period for resident Rainbow Trout and Steelhead, which spans from February 16 to June 15 (see Table 15 of Appendix A for periodicity information). July and August do not coincide with the incubation periods of fish species in the river (Table 1) and therefore the lower DO concentrations measured in those months are not expected to have caused adverse effects to fish, recognizing that all values were above the long-term chronic and instantaneous minimum guideline values that apply to free-swimming life stages of fish (MOE 1997). DO concentrations below the most conservative provincial WQG-AL have routinely been measured in previous years (see Table 8 of Appendix A).

All samples met the WQG-AL for life stages other than buried embryo/alevin (DO instantaneous minimum of 5 mg/L). In BC, surface waters generally exhibit DO concentrations greater than 10 mg/L, and are close to equilibrium with the atmosphere (i.e., ~100% saturation; RISC 1997b).

Total Gas Pressure (TGP)

Monitoring TGP was discontinued in Year 2 following evaluation of results in Year 1, and the limited or lack of potential of the Quinsam River diversion facility to cause elevated TGP (Abell *et al.* 2020). Results from TGP monitoring in Year 1 are presented in Appendix A.

Nitrogen

Total ammonia concentrations in the Quinsam River at QUN-WQ were less than the detection limit of 5.0 μg N/L during five of the six sampling events in Year 8 (Table 21). During the July sampling event, a total ammonia concentration was detectable in one of the duplicates (10.6 μg N/L). All measurements were well below the WQG-AL. Ammonia is usually present at low concentrations (<100 μg N/L) in waters not affected by waste discharges (Nordin and Pommen 2009).

Nitrite concentrations were below the detection limit of 1.0 μg N/L during sampling in Year 8 (Table 21). Nitrite is an unstable intermediate ion serving as an indicator of recent contamination from sewage and/or agricultural runoff; levels are typically <1.0 μg N/L (RISC 1997b).

Nitrate concentrations were low and ranged from 6.9 μg N/L (May) to 38.0 μg N/L (June) during Year 8, similar to previous years (Table 21). In oligotrophic lakes and streams, nitrate concentrations are usually lower than 100 μg N/L (Nordin and Pommen 2009).

Phosphorus

Orthophosphate concentrations were below the detection limit of 1.0 μg P/L during three of the six sampling events in Year 8 (Table 21). During the June, September, and October sampling events, orthophosphate concentrations were detectable in one of the duplicates, with values of 1.6 μg P/L, 1.3 μg P/L, and 1.6 μg P/L, respectively. Low orthophosphate concentrations are typical of coastal BC streams, which generally have orthophosphate concentrations <1.0 μg P/L (Slaney and Ward 1993; Ashley and Slaney 1997).

Total phosphorus concentrations over the Year 8 sampling period were low, similar to previous years, ranging from below MDL (<2.0 μg/L) to 6.3 μg/L (Table 21).

Table 18. Quinsam River (QUN-WQ) general water quality variables measured at ALS laboratories during Year 8 (2021).

Year	Date	Alkalinity, Total (as $CaCO_3$) mg/L				Specific Conductivity µS/cm				Total Dissolved Solids mg/L				Total Suspended Solids mg/L				Turbidity NTU				pH pH units			
		Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD
2021	13-May	32.6	32.2	33.0	0.6	123.0	123.0	123.0	0.0	79	76	81	4	<1	<1	<1	0.0	0.28	0.26	0.29	0.02	7.78	7.77	7.79	0.01
	10-Jun	41.8	41.8	41.8	0.0	184.0	184.0	184.0	0.0	127	116	138	16	<1	<1	<1	0.0	0.45	0.42	0.47	0.04	7.84	7.83	7.84	0.01
	08-Jul	50.5	50.4	50.5	0.1	236.0	236.0	236.0	0.0	143	142	143	1	<1	<1	<1	0.0	0.49	0.46	0.51	0.04	7.84	7.82	7.85	0.02
	16-Aug	47.1	47.0	47.2	0.1	201.0	201.0	201.0	0.0	124	124	124	0	<1.1	<1	1.2	0.1	0.36	0.35	0.37	0.01	7.88	7.87	7.88	0.01
	16-Sep	45.5	45.0	45.9	0.6	198.0	197.0	198.0	1.0	135	134	136	1	<1	<1	<1	0.0	0.27	0.26	0.27	0.01	7.88	7.85	7.91	0.04
	07-Oct	46.6	40.2	53.0	9.1	149.0	146.0	152.0	4.0	94	93	95	1	<1	<1	<1	0.0	0.25	0.23	0.26	0.02	7.89	7.81	7.97	0.11

[1] Average of two duplicates (n=2) on each date unless otherwise indicated.

Parameters that have a concentration below the detection limit are assumed to have a concentration equal to the detection limit for calculation purposes.

Table 19. Quinsam River (QUN-WQ) general water quality variables measured *in situ* during Year 8 (2021).

Year	Date	Air Temperature °C				Conductivity µS/cm				Specific Conductivity µS/cm				Water Temperature °C				pH pH units			
		Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD
2021	13-May	10	10	10	0	80.4	80.3	80.4	0.1	120.0	120.0	120.0	0.1	8.6	8.6	8.6	0.0	6.89	6.86	6.92	0.03
	10-Jun	12	12	12	0	138.0	138.0	138.0	0.0	184.0	184.0	184.0	0.0	12.3	12.3	12.3	0.0	7.12	7.09	7.14	0.03
	08-Jul	20	20	20	0	218.0	218.0	218.0	0.0	235.0	235.0	235.0	0.1	21.3	21.3	21.3	0.0	7.29	7.29	7.30	0.01
	16-Aug	20	20	20	0	169.0	169.0	169.0	0.0	186.0	186.0	187.0	0.1	20.4	20.4	20.4	0.0	7.58	7.57	7.59	0.01
	16-Sep	7	7	7	0	154.0	154.0	154.0	0.0	196.0	196.0	196.0	0.0	13.9	13.9	13.9	0.0	7.32	7.31	7.32	0.01
	07-Oct	6	6	6	0	103.0	103.0	103.0	0.0	145.0	145.0	145.0	0.0	9.8	9.8	9.8	0.0	7.54	7.53	7.55	0.01

[1] Average of three replicates (n=3) on each date unless otherwise indicated.

Table 20. Quinsam River (QUN-WQ) dissolved gases measured *in situ* during Year 8 (2021).

Year	Date	Oxygen Dissolved %				Oxygen Dissolved mg/L			
		Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD
2021	13-May	90.4	90.3	90.4	0.1	10.60	10.50	10.60	0.01
	10-Jun	83.4	82.7	83.7	0.6	8.93	8.89	9.01	0.07
	08-Jul	89.8	89.7	89.9	0.1	7.94	7.94	7.94	0.00
	16-Aug	95.6	93.6	96.8	1.7	8.63	8.54	8.75	0.11
	16-Sep	94.5	94.1	94.7	0.3	9.74	9.70	9.76	0.03
	07-Oct	95.1	95.0	95.2	0.1	10.80	10.80	10.80	0.01

[1] Average of three replicates (n=3) on each date unless otherwise indicated.
Blue shading indicates that the more conservative provincial guideline (DO instantaneous minimum of 9.0 mg/L) for the protection of aquatic life was not met.

Table 21. Quinsam River (QUN-WQ) nutrient concentrations measured at ALS laboratories during Year 8 (2021).

Year	Date	Ammonia, Total (as N) µg/L				Dissolved Orthophosphate (as P) µg/L				Nitrate (as N) µg/L				Nitrite (as N) µg/L				Total Phosphorus (P) µg/L			
		Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD
2021	13-May	<5	<5	<5	0	<1	<1	<1	0	8.2	6.9	9.4	1.8	<1	<1	<1	0	2.6	2.5	2.6	0.1
	10-Jun	<5	<5	<5	0	<1.3	<1	1.6	0.4	37.6	37.1	38.0	0.6	<1	<1	<1	0	6.0	5.6	6.3	0.5
	08-Jul	<7.8	<5	10.6	4.0	<1	<1	<1	0	15.7	15.6	15.8	0.1	<1	<1	<1	0	<2.3	<2	2.5	0.4
	16-Aug	<5	<5	<5	0	<1	<1	<1	0	15.1	14.4	15.7	0.9	<1	<1	<1	0	4.4	4.2	4.5	0.2
	16-Sep	<5	<5	<5	0	<1.2	<1	1.3	0.2	13.0	13.0	13.0	0.0	<1	<1	<1	0	3.4	3.2	3.5	0.2
	07-Oct	<5	<5	<5	0	<1.3	<1	1.6	0.4	18.2	18.1	18.3	0.1	<1	<1	<1	0	3.4	3.4	3.4	0.0

[1] Average of two duplicates (n=2) on each date unless otherwise indicated.
Parameters that have a concentration below the detection limit are assumed to have a concentration equal to the detection limit for calculation purposes.

3.2.3. Water and Air Temperature Monitoring

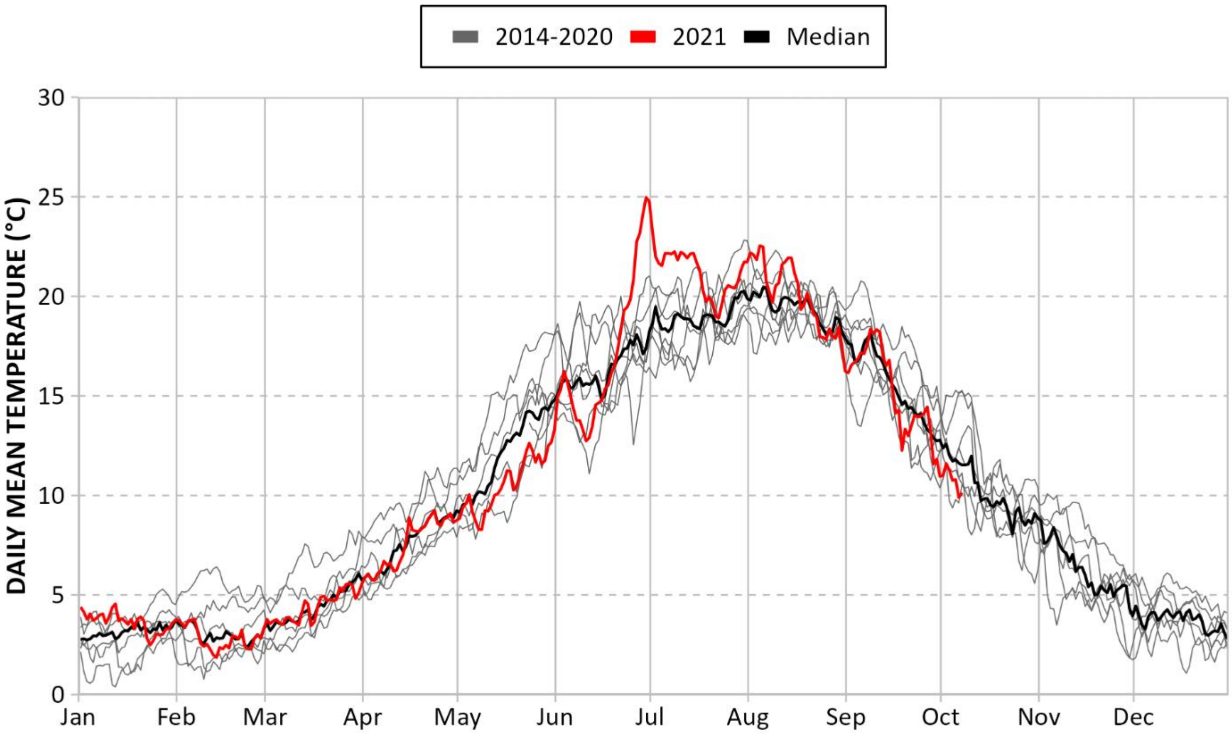
Summary of Water Temperature Records

Figure 10 shows the daily average water temperatures at QUN-WQ from May 2014 to October 2021. In 2021 (January to September), monthly average water temperatures ranged between 2.8°C (February) and 21.2°C (July; Table 11 of Appendix A).

The water temperature records for the Quinsam River show occurrences of warm water temperatures from a fisheries biology perspective. Available data for Year 8 (2021) data were consistent with previous years with the notable exception of late June and early July when the highest water temperatures measured during JHTMON-8 to date were recorded, with a maximum daily mean temperature of 25.0°C recorded on June 29, 2021. The maximum instantaneous water temperature measurement of 26.1°C was recorded the following day. Water temperatures measured during this period coincided with a prolonged period of unusually high pressure that was associated with an unprecedented heat wave throughout BC (Environment and Climate Change Canada 2021).

In 2021, there were 69 days (25% of record) with daily mean temperatures above 18°C, and 47 days (17% of record) with daily mean temperatures above 20°C; 2021 had the highest number of days with daily mean temperatures above 20°C for the period of record, with values for other years ranging from zero (2019) to 30 (2018) (Table 12 of Appendix A). Over the period of record between 2014 and 2020, there were 51 to 77 days per year (14% to 21%) with daily mean temperatures above 18°C, and 0 to 30 days per year (0% to 8%) with daily mean temperatures above 20°C. There was one day in 2020 and seven days in 2017 with mean water temperature <1°C (Table 12 of Appendix A).

Figure 10. Mean daily water temperature (°C) for the Quinsam River (QUN-WQ) between May 2014 and October 2021. The grey lines represent daily mean water temperatures between 2014 and 2020, the red line represents daily mean water temperature for 2021, and the black line represents the median daily water temperatures between 2014 and 2021.



Rates of Change

Statistics relating to rates of change of water temperature at QUN-WQ are summarized in Table 13 of Appendix A. For the period of record, the hourly rates of temperature change at QUN-WQ were between - 0.2°C/hr and +0.2°C/hr for at least 90% of the time (based on the 5th and 95th percentiles) and were between -0.3°C/hr and +0.4°C/hr for at least 98% of the time (based on the 1st and 99th percentiles).

For the period of record, the maximum rate of temperature increase was +1.2°C/hr, and the maximum rate of temperature decrease was -1.9°C/hr (Table 13 of Appendix A). Both these maximum values occurred prior to Year 8 (2021) (Figure 1 of Appendix A). Rates of temperature change with magnitudes >1°C/hr occurred for 0.017% of the records. Based on our experience on other streams in BC, it is normal for hourly rates of water temperature change to occasionally exceed ±1°C.

Growing Season and Accumulated Thermal Units

The length of the growing season and accumulated thermal units (or degree days) are important indicators of the productivity of aquatic systems. As explained in Table 12, the growing season was assumed to begin when the weekly average water temperature exceeded and remained above 7°C, and to end when the weekly average temperature dropped below 7°C.

The growing season at QUN-WQ was determined for 2015 to 2020 (Years 2 to 7), which are the study years for which complete annual records exist (Table 14 of Appendix A). The most recent growing season for which data are available was 2020 (Year 7), for which the growing season commenced on April 9th, ended on November 5th, covering a period of 211 days, and accumulating 3,018 degree days. This was shorter than the growing season length calculated for Year 2 (232 days) and Year 3 (240 days) but longer than for Year 4 (197 days), Year 5 (206 days), and Year 6 (200 days). Growing season statistics for the 2021 growing season will be presented in the Year 9 Annual Report when all 2021 data are available.

Mean Weekly Maximum Water Temperatures (MWMxT)

Fish species of primary interest for JHTMON-8 in the Quinsam River are Steelhead, Coho Salmon, and Chinook Salmon, although Pink Salmon are also particularly important to fishery managers. Steelhead and Coho Salmon are present both upstream and downstream of QUN-WQ, although falls and cascades downstream of Lower Quinsam Lake are complete barriers to Chinook Salmon and Pink Salmon (Burt 2003). Thus, results for the latter two species should be interpreted with caution.

The MWMxT data for 2014 through 2021 are compared to optimum temperature ranges for Chinook Salmon, Coho Salmon, Pink Salmon, and Steelhead in Figure 11, Figure 12, Figure 13, and Figure 14, respectively. A precise synthesis of MWMxT data is presented in Table 15 of Appendix A.

Specifically, for each life stage, Table 15 of Appendix A shows the percentage of MWMxT data that are above, within, and below the optimum ranges for fish life stages during baseline monitoring. The percentages of MWMxT data above and below the optimum ranges by more than 1°C are also shown. Comparisons to the provincial WQG-AL are not made when records are ≤50% complete for the period of interest (Table 15 of Appendix A). In addition, if the water temperature records are only slightly >50% complete for a particular species/life stage, comparisons to the provincial WQG-AL should be interpreted with caution. In Year 8 (2021), data were downloaded on October 7, 2021, prior to the end of the rearing period for stream rearing species or life stages.

Considering all years and all species/life stages, MWMxT in the Quinsam River exceeded optimum ranges by more than 1°C for an average 16.4% of the time, and were below optimum ranges by more than 1°C for an average of 28.7% of the time (Table 15 of Appendix A).

For Chinook Salmon (Figure 11), temperatures were within optimum ranges during the migration stage for all years (2014 to 2020). Temperatures for spawning were mostly within the optimum range (50.8% to 100% of the time) with instances where ranges were exceeded by more than 1°C only occurring in 2014, 2015, 2019, and 2020. Temperatures during incubation were cooler than the optimum range at times during all years, particularly in 2016, when 48.6% of values exceeded the lower bound by more than 1ºC. Water temperatures were outside the optimum range during most of the Chinook Salmon rearing period (temperatures were within the optimum range for 12.4% to 36.5% of the time). In Year 8 (2021), 44.5% of values were below the optimum rearing range and 28.5% of values above the optimum rearing range.

For Coho Salmon (Figure 12), temperatures were typically below the upper bound of the optimum ranges for migration, spawning, and incubation stages (except migration in 2014, 2019, and 2020, where 6.5%, 0.9%, and 9.3% of the temperatures, respectively, were > 1°C higher than the upper bound). Water temperatures during the rearing period were highly variable, with the majority of values outside the optimum range (both above and below) for all years. In Year 8 (2021), water temperatures during the Coho Salmon rearing period were below the lower bound (38.4%) more often than above the upper bound (32.6%) of the optimum temperature range, although the record is only 76% complete as the data were downloaded on October 7, 2021.

For Pink Salmon (Figure 13), the analysis indicates that for all years except Year 2 (2015), the majority of MWMxT values were above the upper bound for migration and spawning, with some years exceeding the upper bound by more than 1ºC for the majority of the time (e.g., up to 90% of the spawning period in 2020). In Year 8 (2021), MWMxT values were above the upper bound by more than 1 ºC for migration (70.1%) and spawning (61.9%) for a higher percentage of time than most previous years, although both of these periods were not fully completed when the data were downloaded on October 7, 2021. During the Pink Salmon incubation period, water temperatures were within optimum ranges for the majority of time, except 2016 when 42.6% of values were within the optimum range.

For Steelhead (Figure 14), MWMxT were rarely (0% to 22.3% of the records) within the optimum ranges for any life stage. Most notably, water temperatures during the spawning stage between 2015 and 2021 were below the optimum range by more than 1°C for 75.0% to 100% of the time. In 2021, water temperatures were never recorded within the optimum bounds during the spawning stage, whereas water temperatures were within the optimum bounds for 11.7% of the incubation stage, and 6.5% of the rearing stage (incomplete at the time of data retrieval).

Note that the WQG-AL temperature ranges for Steelhead life stages are based on those for Rainbow Trout (Oliver and Fidler 2001) and are not specific to fish with an anadromous life history (i.e., Steelhead). Data specific to Steelhead (Carter 2005 and references therein) indicate that Steelhead are adapted to tolerate MWMxT considerably lower than the optimum ranges presented in Figure 14 and Table 15 of Appendix A during spawning and incubation, although survival is likely to be affected by temperatures that exceed these ranges. For example, Carter (2005) cites WDOE (2002), which reports that the low end of the range of preferred spawning temperatures for Steelhead is 4.4°C, rather than the MWMxT value of 10.0°C reported in Table 15 of Appendix A for Rainbow Trout. Thus, although the alternative values cited above may not be fully representative of Steelhead populations on Vancouver Island, the occurrence of MWMxT in the Quinsam River that are below 10.0°C do not necessarily indicate poor conditions for spawning and incubation life stages of Steelhead.

Figure 11. Mean weekly maximum temperatures (MWMxT) in the Quinsam River from 2014 to 2021 compared to optimum temperature ranges for Chinook Salmon. Periodicity information is from Burt (2003).

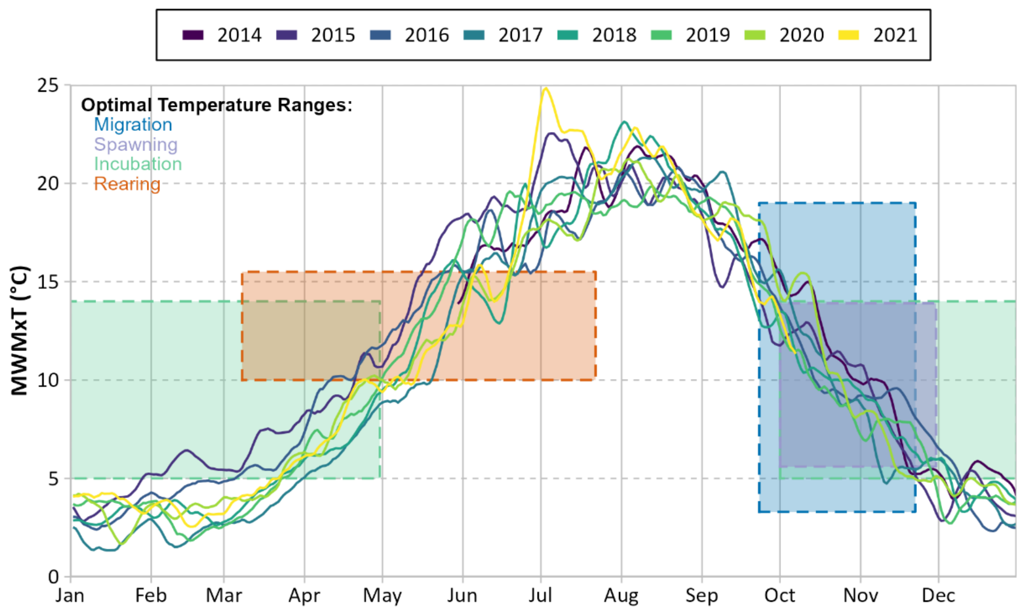


Figure 12. Mean weekly maximum temperatures (MWMxT) in the Quinsam River from 2014 to 2021 compared to optimum temperature ranges for Coho Salmon. Periodicity information is from Burt (2003).

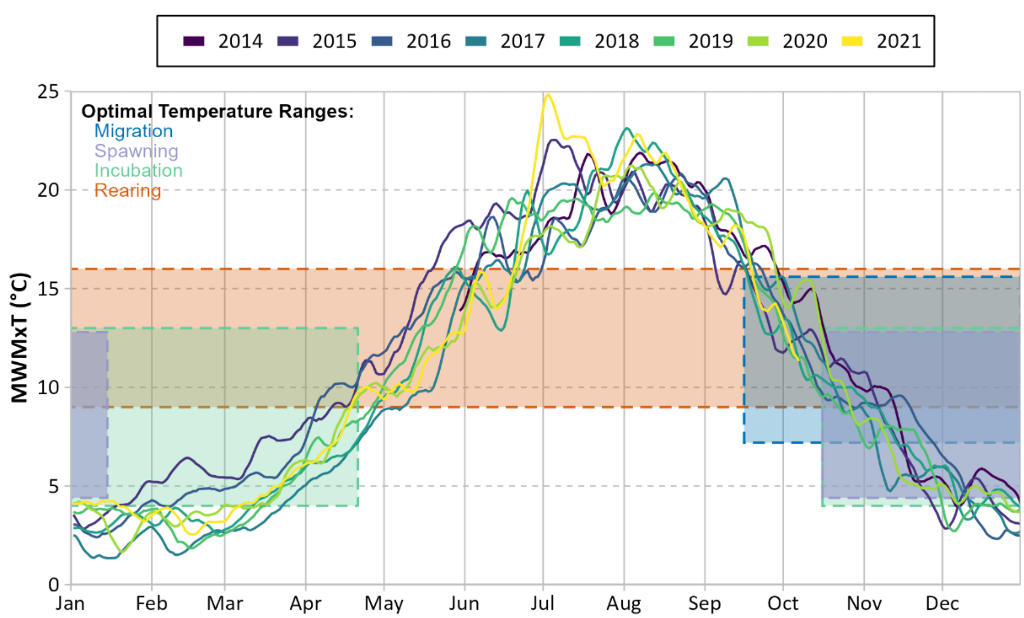


Figure 13. Mean weekly maximum temperatures (MWMxT) in the Quinsam River from 2014 to 2021 compared to optimum temperature ranges for Pink Salmon. Periodicity information is from Burt (2003).

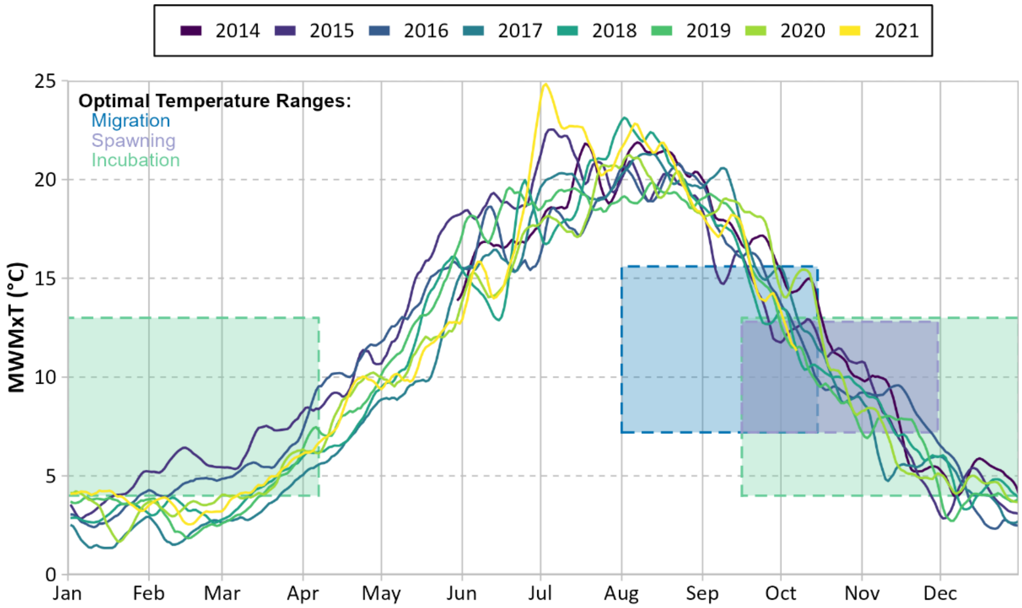
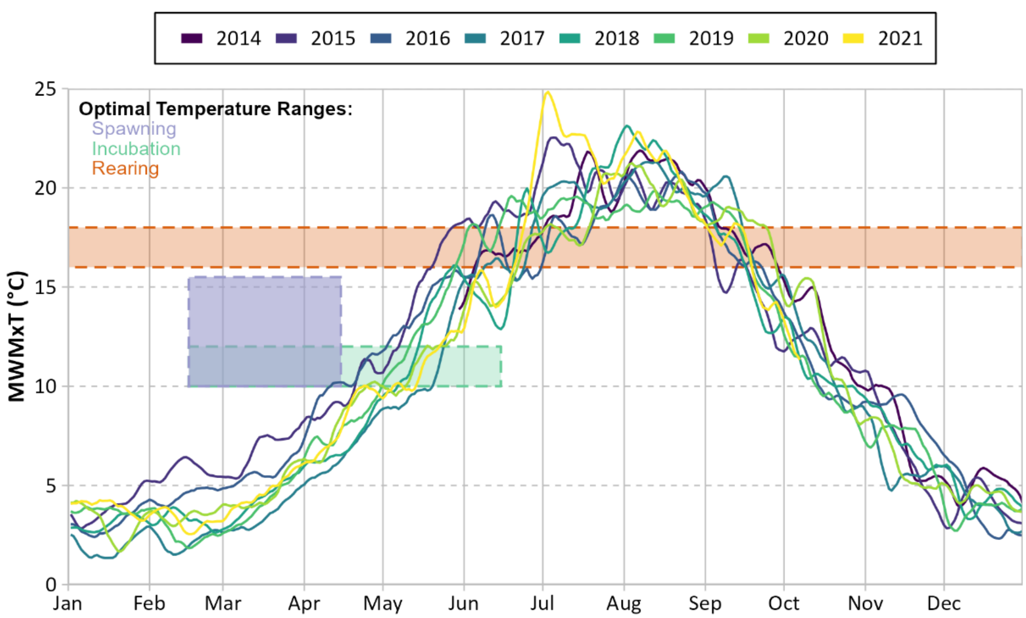


Figure 14. Mean weekly maximum temperatures (MWMxT) in the Quinsam River from 2014 to 2021 compared to optimum temperature ranges for Steelhead. Periodicity information is from Burt (2003).



Air Temperature

Air temperature data are provided in Section 3 of Appendix A.

Figure 2 of Appendix A shows the daily average air temperature for the period of record from May 2014 to October 2021. The monthly average, minimum, and maximum air temperatures are shown in Table 16 of Appendix A. The mean monthly air temperature ranged from -2.2°C to 18.8°C during the period of record. The lowest air temperature measured during the monitoring period was -12.5ºC measured in February 2019, while the highest air temperature was 36.4°C in June 2021. The maximum monthly mean air temperature (18.8°C) was in July 2015. Mean monthly air temperatures during summer 2021 were generally higher than previous years of JHTMON-8; e.g., the mean monthly air temperature during July 2021 (18.7ºC) was higher than six of the previous years, and mean monthly air temperature during June 2021 (16.5°C) was higher than four of the previous years.

Air and water temperatures were highly correlated (Figure 3 of Appendix A). Daily mean water temperatures typically exceeded daily mean air temperatures, which likely partly reflected the influence of warming in lakes upstream.

3.3. Invertebrate Drift

3.3.1. Quinsam River Invertebrate Drift

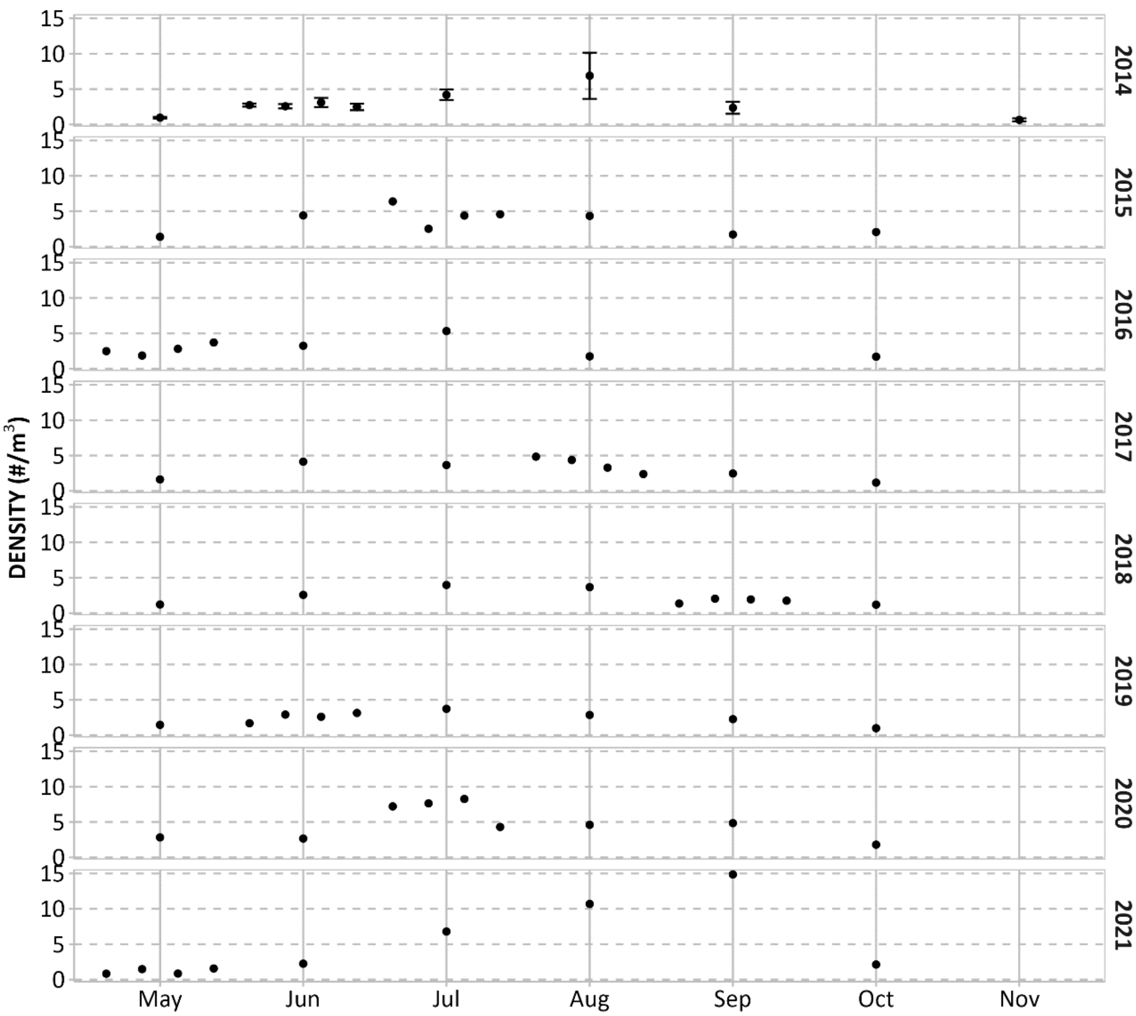
3.3.1.1. Overview

Results relating to invertebrate drift density (individuals/m^3) and biomass (mg/m^3) are provided in subsequent sections for the Quinsam River to provide indicators that could potentially be used to analyze drivers of changes in fish abundance to test H_05. Supplementary invertebrate drift results relating to Simpson's family-level diversity index (1-λ), richness (# families), and CEFI are provided in Appendix B. Standard deviation values are provided for Year 1 (2014) data only, which is the only year when samples from all five drift nets were analyzed separately. All values except for the CEFI (for which only aquatic taxa are considered) were calculated based on results for all taxa (aquatic, semi-aquatic, and terrestrial).

3.3.1.2. Density

Invertebrate drift density in the Quinsam River was variable among sampling dates in Year 8 (2021) (Figure 15). The lowest density was observed on the first sampling date (0.83 individuals/m^3 on May 6, 2021); density then generally increased to reach a peak of 14.85 individuals/m^3 in September 2021, before declining to 2.12 individuals/m^3 on October 7, 2021 (Figure 15). Density measured at weekly intervals during May ranged from 0.83 – 1.56 individuals/m^3 (Figure 15). In Year 8, density ranged from 0.83 – 14.85 individuals/m^3, which is higher than the range of values observed in previous years (0.65 – 8.26 individuals/m^3; Figure 15). A notable result from Year 8 was the particularly high invertebrate density recorded in August and September (10.68 individuals/m^3 and 14.85 individuals/m^3, respectively). These high densities are largely attributed to a high abundance of Ostracoda (small crustaceans), which typically comprised only a minor proportion of the invertebrate community in previous years of JHTMON-8, except for August 2014 (Year 1) and August 2017 (Year 4) when high abundances of Ostracoda were also recorded in the Quinsam River.

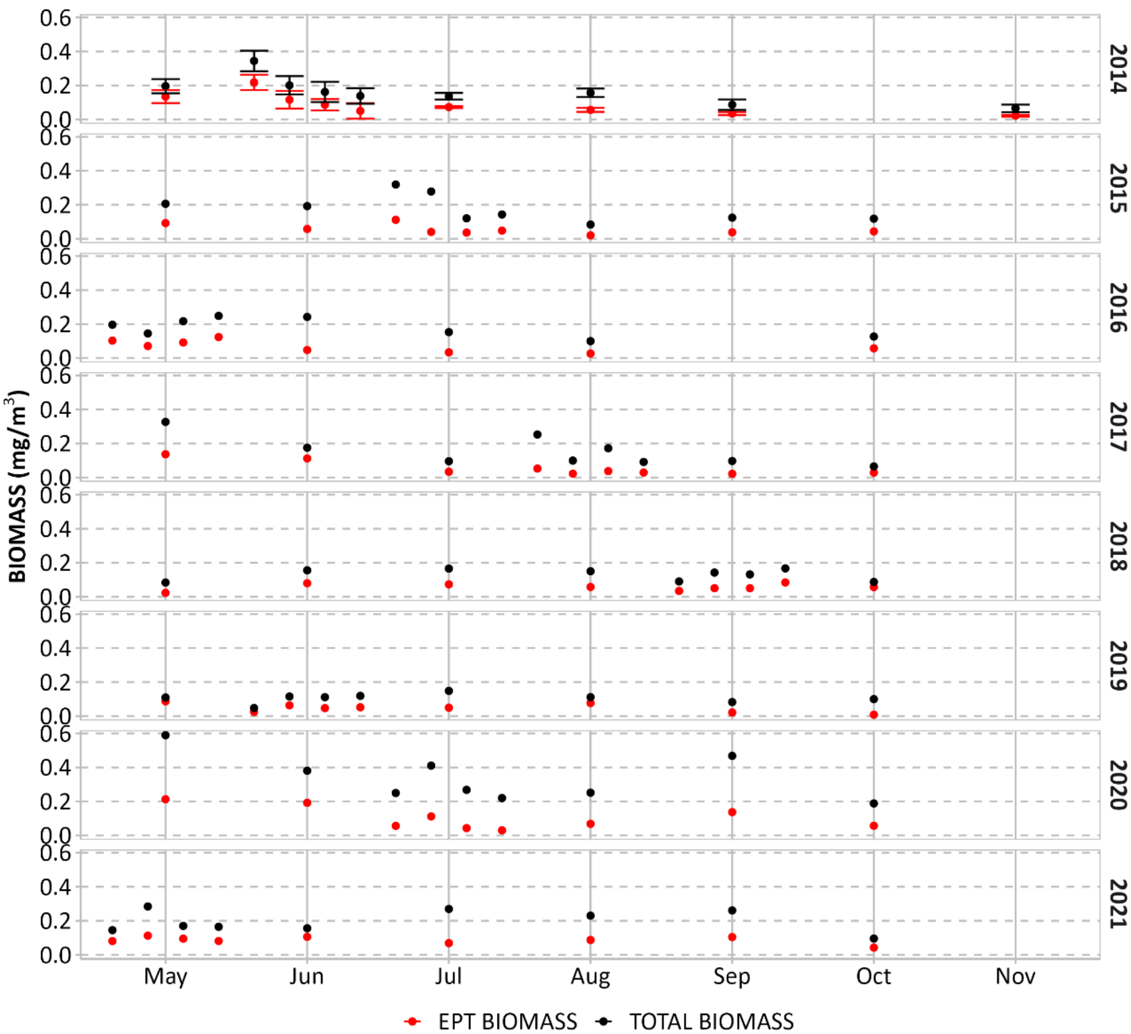
Figure 15. Drift invertebrate density (all taxa) in the Quinsam River, 2014 – 2021. Standard deviation (vertical bars) is provided for Year 1 (2014) only, which is the only year when samples from all five drift nets were analyzed separately.



3.3.1.3. Biomass

Total invertebrate drift biomass in the Quinsam River ranged from 0.10 – 0.28 mg/m^3 in Year 8 (2021), which is within the range observed in previous years (0.05 – 0.59 mg/m^3; Figure 16). Thus, despite the high abundance of ostracods in late summer (Section 3.3.1.2), the small body size of this taxon meant that biomass values recorded in August and September were not notably high. Total biomass was variable throughout Year 8, with the annual maximum value of 0.28 mg/m^3 observed in May 2021, which was greater than the maximum value in three of the preceding seven years (Years 3, 5, and 6; Figure 16). EPT biomass was also variable in Year 8 and contributed to a large portion of the total biomass on most sampling dates, although the relative proportion of EPT taxa was generally higher in spring than in the summer (Figure 16).

Figure 16. Total drift invertebrate biomass (all taxa) and EPT biomass in the Quinsam River throughout 2014 – 2021. Standard deviation (vertical bars) is provided for Year 1 (2014) only, which is the only year when samples from all five drift nets were analyzed separately.



3.3.1.4. Top Five Families Contributing to Biomass

A summary of the top five families contributing to biomass of the invertebrate drift community in Year 8 (2021) is provided in Table 22. Note that, in some instances, a taxonomic level higher than family is listed (e.g., Ephemeroptera), as this was the lowest taxonomic level enumerated.

The invertebrate community was dominated (in terms of biomass) by mayflies (notably Baetidae) and true flies (most notably Chironomidae and Simuliidae) in Year 8. Mayflies were present in the top five families on all nine sampling dates and were the most dominant family on six of the nine sampling dates. True flies were also consistently present in the top five, with one or more true fly families present on eight of nine sampling dates. The contribution to biomass of individual mayfly families ranged from 4.7% to 36.3%, while individual true fly families ranged from 3.8% to 25.3% on the eight out of nine dates when these taxa were included in the top five families.

Other taxa sometimes present in the top five included caddisflies (Philopotamidae), beetles (Cantharidae), spiders (Araneae), horsehair worms (Nematomorpha), aquatic worms (Lumbriculidae), crustaceans (Ostracoda), stoneflies (Plecoptera), and butterflies/moths (Lepidoptera). It is notable that Ostracoda were present on two of nine sampling dates in Year 8 (2021; September and October), whereas Ostracoda have not been present in the top five families since Year 1 (2014).

A summary of the top five families contributing to biomass across all JHTMON-8 monitoring years in the Quinsam River is provided in Table 23. These results show similarities in the top five families across years, with Baetidae comprising the top family in six of eight years (including Year 8) and present in all eight years, as were two other families (Chironomidae and Simuliidae). In all years, these three families comprised 31.4–49.5% of the biomass (43.3% in Year 8). Ostracoda were only present in the top five families in Year 8 (2021).

Ephemeroptera, Trichoptera, and Plecotera can be particularly important invertebrate prey for juvenile salmonids in streams (Johnson and Ringler 1980; Rader 1997). Ephemeroptera taxa were present in the top five families during each sampling date in Year 8 (2021) as well as across years. Trichoptera were present in the top five families during three sampling dates in Year 8 (2021) and were present in the top five families overall in 2014, 2016, 2018, and 2021. Plecoptera were only present in the top five families on two sampling dates in Year 8 (2021) and were only present in the top five families overall in 2015.

Table 22. Top five families contributing to invertebrate drift biomass (all taxa) in the Quinsam River in Year 8 (2021). Names in parentheses represent taxa higher than families in instances where family level classifications were unavailable.

QUN-IV	6-May-21	QUN-IV	13-May-21	QUN-IV	18-May-21	QUN-IV	25-May-21	Key	
Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	True Flies	Horsehair Worm
								Mayflies	Crustacean
Baetidae	35.8	Baetidae	21.9	Baetidae	24.7	Baetidae	36.3	Caddisflies	Aquatic Worm
Simuliidae	11.2	Bibionidae	10.2	(Nematomorpha)	18.5	Simuliidae	11.9	Stoneflies	
Bibionidae	9.2	Ephemerellidae	6.7	(Ephemeroptera)	17.0	Chironomidae	11.7	Butterflies/Moths	
(Plecoptera)	7.6	(Lepidoptera)	6.3	Cantharidae	7.1	(Nematomorpha)	10.2	Spiders	
Cantharidae	7.6	Simuliidae	6.3	Ephemerellidae	7.1	(Ephemeroptera)	4.7	Beetles	
Sum	71.3	Sum	51.4	Sum	74.4	Sum	74.8		

QUN-IV	10-Jun-21	QUN-IV	8-Jul-21	QUN-IV	16-Aug-21	QUN-IV	16-Sep-21	QUN-IV	7-Oct-21
Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass
Baetidae	23.0	Chironomidae	17.9	Chironomidae	22.4	(Trichoptera)	30.4	Baetidae	25.9
(Ephemeroptera)	17.4	Simuliidae	15.6	Philopotamidae	17.0	(Ostracoda)	23.3	Chironomidae	25.3
Chironomidae	12.8	Baetidae	15.5	Baetidae	16.4	Chironomidae	11.1	(Plecoptera)	8.1
Philopotamidae	11.3	Sciaridae	6.6	(Araneae)	11.3	Baetidae	8.2	(Ostracoda)	7.1
Simuliidae	8.5	Empididae	6.1	Simuliidae	11.1	Lumbriculidae	5.3	Simuliidae	3.8
Sum	72.9	Sum	61.7	Sum	78.2	Sum	78.3	Sum	70.1

Table 23. Top five families contributing to invertebrate drift biomass (all taxa) in the Quinsam River each year throughout Years 1 to 8. Names in parentheses represent taxa higher than families in instances where family level classifications were unavailable.

QUN-IV	2014	QUN-IV	2015	QUN-IV	2016	QUN-IV	2017
Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass
Baetidae	20.2	Chironomidae	14.4	Baetidae	15.9	Baetidae	18.0
Limnephilidae	15.8	Simuliidae	13.2	Chironomidae	15.3	Chironomidae	12.0
Chironomidae	9.5	Baetidae	11.5	Simuliidae	12.0	Simuliidae	9.4
Simuliidae	7.5	Chrysomeloidea	6.7	Limnephilidae	5.8	Empididae	8.6
(Ephemeroptera)	5.8	(Plecoptera)	4.2	Cicadellidae	3.5	Bibionidae	5.7
Sum	58.8	Sum	50.0	Sum	52.5	Sum	53.8

Key	
True Flies	Mites
Mayflies	Crustacean
Caddisflies	
True Bugs	
Stoneflies	
Spiders	
Beetles	

QUN-IV	2018	QUN-IV	2019	QUN-IV	2020	QUN-IV	2021
Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass
Baetidae	21.3	Baetidae	28.3	Chironomidae	14.8	Baetidae	21.3
Simuliidae	12.6	Simuliidae	12.8	Baetidae	9.8	Chironomidae	12.9
Chironomidae	12.1	Chironomidae	8.4	Simuliidae	6.8	Simuliidae	9.1
Hydropsychidae	6.0	Torrenticolidae	7.8	Sciaridae	6.0	(Ostracoda)	5.6
(Araneae)	3.8	Heptageniidae	3.1	Empididae	5.5	(Trichoptera)	4.9
Sum	55.9	Sum	60.4	Sum	43.0	Sum	53.7

3.3.2. Comparison of Kick Net and Drift Net Sampling Methods

As a proportion of total biomass, invertebrates collected using kick net sampling in the Quinsam River in Year 8 (2021) were almost exclusively aquatic taxa (99.6%), whereas drift net sampling captured a mixture of aquatic (61.5%) and semi-aquatic (35.6%) taxa, with a small proportion of terrestrial taxa (2.9%; Table 24), for samples collected on the same date and location. These results were generally consistent with all years, in general: kick net sampling has almost exclusively captured aquatic taxa (99.6–100%), whereas drift sampling has captured 49.8-79.3% aquatic invertebrates (based on biomass; Table 24). The kick net method involves holding the collection net completely under the stream surface for three minutes, so the greater dominance of aquatic taxa is expected. Drift nets are installed with the top of the net above the stream surface, so that any invertebrates suspended on the surface are collected, in addition to submerged invertebrates. These invertebrates at the surface are more likely to have entered the stream from terrestrial or riparian habitats.

The contribution of individual families to invertebrate biomass also differed between the two sampling methods (Table 25). In the Quinsam River, two groups (true flies and mayflies) accounted for most of the biomass in drift net samples in all sampling years except Year 8 (2021), when Trichoptera and Ostracoda accounted for most of the biomass in drift samples. A wider range of families were present during kick sampling, including Hydropsychidae (caddisflies), Gomphidae (dragonflies), Astacidae (crayfish), and Lumbricidae (earthworms). Overall, the taxa present in the kick net samples were more diverse within and among sampling dates than taxa present in drift net samples. Both sampling methods are appropriate for sampling streams and the methods are expected to provide suitable data to support the study; however, this comparison of methods demonstrates that neither method provides data that fully reflect the diversity of potential prey items available to juvenile fish, thus supporting an approach of considering both datasets in combination.

Table 24. Contribution of invertebrate taxa to total biomass by habitat type on the Quinsam River. Kick net data were not collected in 2014 and 2016.

Sample Date	Collection Method	Relative Contribution to Biomass (%)		
		Aquatic Taxa	Semi-Aquatic Taxa	Terrestrial Taxa
16-Sep-2015	Driftnet	75.0	19.2	5.8
	Kicknet	100.0	0.0	0.0
13-Sep-2017	Driftnet	64.5	15.7	19.8
	Kicknet	100.0	0.0	0.0
12-Sep-2018	Driftnet	64.2	24.9	10.9
	Kicknet	100.0	0.0	0.0
12-Sep-2019	Driftnet	79.3	2.3	18.4
	Kicknet	99.6	0.4	0.0
10-Sep-2020	Driftnet	49.8	28.2	22.0
	Kicknet	100.0	0.0	0.0
16-Sep-2021	Driftnet	61.5	35.6	2.9
	Kicknet	99.6	0.4	0.0

Table 25. Top five families contributing to invertebrate biomass collected using drift nets and a kick net in the Quinsam River. Names in parentheses represent taxa higher than families in instances where family level classifications were unavailable. Key includes habitat types the collected invertebrate taxa are associated with.

Date	Driftnet		Kicknet	
	Family	% of Biomass	Family	% of Biomass
16-Sep-2015	Simuliidae	39.0	Hydropsychidae	16.5
	Chironomidae	15.5	Tipulidae	14.5
	(Ephemeroptera)	13.7	(Trichoptera)	13.7
	Ameletidae	6.3	Chironomidae	7.3
	Sperchontidae	4.7	Lumbriculidae	5.9
13-Sep-2017	Chironomidae	25.4	Astacidae	26.5
	Simuliidae	17.5	Naididae	11.8
	Baetidae	11.3	Gomphidae	10.8
	Curculionidae	8.6	Elmidae	9.0
	Aphididae	6.2	Chironomidae	6.0
12-Sep-2018	Baetidae	21.1	Heptageniidae	33.6
	Psychodidae	20.7	Perlidae	17.9
	Simuliidae	17.9	Hydropsychidae	13.0
	Chironomidae	7.9	Tipulidae	8.8
	(Plecoptera)	7.5	Baetidae	7.9
12-Sep-2019	Chironomidae	22.0	Hydropsychidae	21.2
	Baetidae	19.5	Tipulidae	13.6
	Simuliidae	14.3	Lumbricidae	11.9
	Coccinellidae	8.1	Heptageniidae	11.3
	Aphididae	7.4	Chironomidae	10.3
10-Sep-2020	Simuliidae	16.8	Lumbriculidae	51.9
	Empididae	15.7	Tipulidae	14.4
	Baetidae	14.3	Heptageniidae	5.5
	Hydropsychidae	10.9	Leptophlebiidae	5.1
	(Lepidoptera)	6.9	Chironomidae	4.2
16-Sep-2021	(Trichoptera)	30.4	Astacidae	53.3
	(Ostracoda)	23.3	Leptophlebiidae	20.0
	Chironomidae	11.1	Lumbriculidae	7.4
	Baetidae	8.2	Elmidae	3.6
	Lumbriculidae	5.3	Chironomidae	3.5

Key	Habitat Type
True Bugs	Terrestrial
Aquatic Worms	Aquatic
Mites	Aquatic
True Flies	Aquatic, Semi-Aquatic, Terrestrial
Mayflies	Aquatic, Semi-Aquatic
Caddisflies	Aquatic, Semi-Aquatic
Crustaceans	Aquatic
Dragonflies	Aquatic
Stoneflies	Aquatic, Semi-Aquatic
Beetles	Aquatic, Semi-Aquatic, Terrestrial
Earthworms	Aquatic
Butterflies/Moths	Semi-Aquatic, Terrestrial

4. SUMMARY

4.1. JHTMON-8 Status

JHTMON-8 is ongoing and analyses to test the management hypotheses and address the management questions will be undertaken in Year 10 when data collection is complete. For each hypothesis, this section summarizes of the status of data collection to date and describes key results. Hypotheses are described in full in Section 1.4 and paraphrased in the subheadings below.

4.2. H_01: Juvenile Fish Abundance Does Not Vary in Time

The JHTMON-8 results and historical data compiled so far show considerable inter-annual variability in juvenile fish abundance, suggesting that this hypothesis will be rejected in Year 10. For example, Figure 6 shows that juvenile abundance of JHTMON-8 priority species has varied by at least a factor of four for juvenile Chinook Salmon, Coho Salmon and Steelhead throughout the period of record. For the JHTMON-8 period to date (2014–2021), variability in annual outmigration data provided by DFO has been greatest for wild Chinook Salmon (~600 to ~360,000 fry) and lower for wild Coho Salmon (~22,000 to ~57,000 smolts) and Steelhead (~3,000 to ~13,000 smolts) (Figure 5).

A key result from Year 8 (2021) was the particularly high abundance of outmigrating juvenile Chinook Salmon recorded at the Quinsam Hatchery fence (~269,000), which was the second highest value recorded during the eight years of JHTMON-8 (Figure 5), and the third highest value recorded overall in the period of record (Figure 6). The abundance of spawners that correspond to this cohort (~8,236 in 2020) was moderate relative to the period of record (Figure 4), therefore suggesting that egg to fry survival of wild Chinook Salmon was unusually high for the cohort that outmigrated in Year 8.

4.3. H_02: Juvenile Fish Abundance is Not Correlated with Habitat Availability

Annual habitat availability can be expressed based on WUA (in m^2), which provides an index of habitat availability calculated using relationships between flow and habitat area, accounting for differences in habitat suitability across different flows (Lewis *et al.* 2004). Flow-habitat relationships have not been previously developed for Pacific salmon rearing habitat. This issue is only potentially applicable to Coho Salmon because the other two species spend limited time rearing in the river (Burt 2003). Accordingly, we plan to use Steelhead fry rearing habitat WUA estimates as a proxy for juvenile Coho Salmon rearing habitat in the final analysis in Year 10. Such analysis will quantitatively analyze the relationship between habitat availability and juvenile fish recruitment to test H_02 and evaluate whether there is a detectable and biologically significant relationship between the two variables.

Annual average WUA for Steelhead life stages varied throughout the dataset, with variability highest for Steelhead spawning WUA (note that our ability to test this hypothesis will partly depend on the magnitude of variability (i.e., the range in the independent variable) observed). Results from Year 5 showed that variability in annual average spawning habitat WUA was similar among the three

Pacific salmon species, with maximum differences among years of approximately 100% (i.e., approximately two-fold differences).

4.4. <u>*$H_0$3*: Juvenile Fish Abundance is Not Correlated with Water Quality</u>

Year 8 (2021) water quality results were generally consistent with results for Year 1 through Year 7, except for the occurrence of unusually warm water temperatures recorded in early summer (discussed below). The Quinsam River is typical of streams in coastal BC watersheds with low nutrient concentrations (oligotrophic), near-neutral pH, and low turbidity during baseflow.

Measurements of some water quality variables were, at times, outside of the biologically optimum ranges for fish species present in the watershed. Specifically, water temperatures were recorded in the Quinsam River that exceeded WQG-AL temperature ranges for suitable salmonid rearing conditions. This has been observed in previous years; however, the temporary occurrence of undesirably warm water temperatures from a biological perspective was most pronounced in Year 8, when the highest water temperatures measured during the study to date were recorded, with a maximum daily mean temperature of 25.0°C recorded on June 29, 2021, in association with an unprecedented heat wave throughout BC (Environment and Climate Change Canada 2021). Mean weekly maximum temperatures measured in Year 8 exceeded the upper limit of the optimum temperature ranges for the rearing life stage of all fish species (Section 3.2.3), and the number of days with water temperature >20°C (47 days) in Year 8 was substantially higher than in previous years (range: 0–30 days). In addition to the temporary occurrence of undesirably warm water temperatures, MWMxT in the Quinsam River were below optimum ranges by more than 1ºC for an average of 28.7% of the time, although water temperatures in 2021 were not abnormally cool (Table 15 in Appendix A) and, based on our experience of monitoring streams elsewhere in BC, the frequency and magnitude of cool temperatures were typical of coastal BC systems, including those with productive fisheries.

As in previous years, concentrations of DO were lower than the provincial WQG-AL for the protection of buried embryos/alevins (DO instantaneous minimum of 9 mg/L). However, the minimum average DO concentration measured in the incubation period (8.93 mg/L on June 10, 2021; Table 20) was only marginally (~1%) less than the WQG-AL, which limits the potential for the low DO concentrations to be a biological concern.

Thus, based on results to date, water temperature is preliminarily considered to be the water quality variable with the greatest potential to affect fish production in the watershed. The potential for water quality variables including water temperature and DO concentrations to limit fish production will be considered in more detail during the final analysis in Year 10, as discussed in Section 5.3.

4.5. <u>*$H_0$4*: Juvenile Fish Abundance is Not Correlated with the Occurrence of Flood Events</u>

Multiple gauges maintained by the Water Survey of Canada (Table 7) provide data to characterize hydrologic variability in the Quinsam River. Preliminary analyses undertaken in Year 8 (2021) provide initial insight into potential links between hydrologic variability and juvenile fish abundance. As described in Section 3.1.3, analysis of stock-recruitment curves suggests that productivity of

Pink Salmon (odd years spawning stock) was negatively affected by high discharge during the incubation period (Figure 8), whereas no link was identified for other stocks, including all JHTMON-8 priority species (Table 16). A possible mechanism to explain this is that high flows during the incubation period reduce incubation success due to redd scour and associated mortality of embryos. The observation that the effect of high flows was only apparent for Pink Salmon likely reflects that Pink Salmon is the most abundant salmonid species in the stream (Figure 4), and spawning is generally confined to the lower reaches of the river mainstem (Burt 2003). Moreover, Pink Salmon spawn at high densities and construct relatively shallow redds (Quinn 2005). The observation that this effect was significant for the odd year stock but not the even year stock presumably reflects that only the odd year stock was exposed to sufficiently adverse high flow conditions for the effect to be identified. For context, the maximum incubation flow used in the analysis of the odd year stock was 147.0 m^3/s (1997), whereas the maximum incubation flow used in the analysis of the even year stock was 107.0 m^3/s (2010).

By contrast to the result for the Pink Salmon odd year stock, high discharge during the incubation period had a weak positive effect on the production of juvenile Steelhead (Figure 9). Nonetheless this relationship warrants closer scrutiny in Year 10 when data for two additional years will be available.

Although the study hypothesis is premised on the expectation that high flows during incubation may adversely affect recruitment (Section 1.5.5), there are mechanisms by which elevated flows in winter/spring could cause positive effects, e.g., due to improved passage conditions during upstream migration (Marriner *et al.* 2020). Such effects will be further examined in Year 10, e.g., by incorporating flow metrics related to the adult migration period into the analysis.

As an exploratory analysis, we also extended our consideration of H_04 to evaluate hydrologic variability more generally by examining whether there was a relationship between low flows in the growing season and the recruitment of Coho Salmon and Steelhead. There was no link between recruitment and minimum 7-day average discharge. The effects of flow on rearing habitat availability will be considered more directly in Year 10 by considering WUA (Section 5.2).

The initial analysis undertaken in Year 8 (2021) will be extended in Year 10 to further evaluate H_04, as described in Section 5.4. One important addition will be to also consider the potential for high flows to affect other life stages of juvenile fishes, aside from incubation. For example, Scheuerell *et al.* (2021) studied the Skagit River (WA, USA) and found that juvenile Steelhead survival was negatively associated with peak winter flows, based on analysis of daily peak flows occurring from October through March in the first freshwater rearing year. The authors attributed the adverse effect of peak winter flows to direct fish mortality caused by channel avulsion or debris movement, or transport of fish downstream to lower quality habitats.

4.6. *H_05*: Juvenile Fish Abundance is Not Correlated with Food Availability

Invertebrates typically form the bulk of the diet of salmonids in rivers (Quinn 2005) and a change in invertebrate community structure can affect food quality (i.e., a decrease in the biomass of taxa preferred by salmonids), which could theoretically affect juvenile growth and abundance.

Invertebrate drift data have now been collected for eight growing seasons for the Quinsam River. There are no clear differences in invertebrate drift biomass among years, with Year 8 (2021) biomass within the range of previous years (Figure 16). Otherwise, invertebrate drift biomass has generally tended to decline during the growing season, although this trend has not been pronounced during the last four years and there was no clear seasonal trend in invertebrate biomass during Year 8 (Figure 16).

A notable result from Year 8 (2021) was the high invertebrate densities (but not necessarily biomass) observed in August and September. These high densities are largely attributed to high abundance of Ostracoda, which contributed to the highest total density measured in a sample to date of 14.85 individuals/m^3 in September 2021. Ostracods are small crustaceans that are common in freshwater benthic habitats and can be known as "mussel shrimps" (Martens *et al.* 2008). Ostracods have been shown in other studies to be a prey item for juvenile salmonids in streams (e.g., McNicol *et al.* 1985). Ostracods were one of the top five families contributing to invertebrate drift biomass for Year 8 (2021), which had not been observed in previous years. Ostracoda have typically been present in low abundance in previous years, except for August 2014 (Year 1) and August 2017 (Year 4) when they were also relatively abundant. The cause of variability in the biomass of this taxon is unknown.

4.7. *H06*: Annual Smolt Abundance is Not Correlated with the Number of Adult Returns

Analysis to test this hypothesis was initiated in Year 7, when we developed initial stock (spawner)-recruitment relationships for priority species to quantify the relationship between the abundance of adult spawners and the subsequent recruitment of juvenile fish each year (Suzanne *et al.* 2021). To increase statistical power, the analysis drew on historical juvenile abundance data collected since the 1970s that were compiled as part of an additional task completed during in Year 5 (Abell *et al.* 2019).

Initial stock-recruitment relationships were consistent with general patterns expected for Pacific salmon stock recruitment. For most species, there was some evidence that the abundance of recruits reached an asymptote ("plateau"), or the relationship showed overcompensation at high spawner abundance. However, this was not clearly the case for Chinook Salmon, for which there is lowest spawner abundance of the four species analyzed. The stock-recruitment relationships were used in Year 8 to complete preliminary analyses to evaluate how flow metrics affect recruitment (see Section 4.5). Stock-recruitment curves will be further updated in Year 10 to formally test *H_06* (Section 5.6).

5. FUTURE TASKS

This section provides an overview of the planned approach to test each hypothesis, including how work undertaken in previous years will be used in the analysis. Additional tasks proposed for the remaining years of JHTMON-8 are summarized in Section 5.7 below.

5.1. H_01: Juvenile Fish Abundance Does Not Vary in Time

In Year 10, variability in juvenile fish abundance will be analyzed by reviewing time series graphs and calculating summary statistics (e.g., standard deviation and percentile values). Where feasible, stock-recruitment relationships will be constructed and analyzed to isolate variability in juvenile fish abundance that is due to variability in freshwater survival, from variability due to fluctuations in the abundance of adult fish. Analysis in Year 10 will draw on work undertaken in Year 5 (Abell *et al.* 2019) to compile, digitize, and analyze juvenile fish outmigration data collected at the Quinsam Hatchery fence prior to JHTMON-8 (since the 1970s; Figure 6), which will substantially increase the statistical power of analysis to quantify variability in juvenile fish abundance in the Quinsam River. Furthermore, analysis in Year 10 will draw on the outcomes of a review of capture efficiency estimates completed in Year 6 (Suzanne *et al.* 2020), which examined how to reduce uncertainty associated with the results of juvenile mark-recapture experiments conducted at the Quinsam Hatchery salmon counting fence.

5.2. H_02: Juvenile Fish Abundance is Not Correlated with Habitat Availability

The WUA analysis initiated in Year 5 (Abell *et al.* 2019) will be updated in Year 10 and used to test H_02. We propose to test this hypothesis separately for each of the JHTMON-8 priority species. For Chinook Salmon and Coho Salmon, we propose to construct stock-recruitment relationships (discussed further in Section 4.7) and then test whether variability in WUA explains variability in the stock-recruitment relationships, which would indicate that variability in WUA affects juvenile fish recruitment (indicating that H_02 can be rejected). For these two species, the flow-habitat relationships that have been previously developed relate to spawning (not rearing) habitat. For Chinook Salmon, this is reasonable because this species only spends up to a few months rearing in the Quinsam River (Burt 2003). Coho Salmon typically rear in freshwater for 1–2 years in the Quinsam River (Burt 2003) and therefore we will consider whether it is feasible to also analyze whether variability in rearing habitat WUA affects juvenile Coho abundance.

At this time, we propose to use Steelhead fry rearing habitat WUA estimates as a proxy for juvenile Coho Salmon rearing habitat, since both Steelhead fry and juvenile Coho Salmon prefer habitats with low water velocity; however, we plan to examine this assumption further in Year 10 (e.g., by comparing the HSI curve used to calculate Steelhead fry habitat with curves developed elsewhere for juvenile Coho Salmon). In addition to these two priority salmon species, we also propose to test H_02 using the same approach for Pink Salmon, which is a species of interest in the Quinsam River watershed. For Steelhead, H_02 will be tested in relation to spawning habitat, as well as rearing habitat for two life stages (fry and parr). We do not expect to construct stock-recruitment relationships for Steelhead because adult Steelhead abundance is not monitored in the Quinsam River; instead, we plan to complete the analysis using total Steelhead smolt outmigration as the dependent variable.

5.3. H_03: Juvenile Fish Abundance is Not Correlated with Water Quality

Analyses to test H_03 will be undertaken separately for individual species and water quality variables. The analyses will initially focus on the ten-year period of the monitoring program, although there are opportunities to use water temperature data collected by other parties to extend the time period over which the potential effects of water temperature are considered, as identified during a review conducted in Year 2 (Dinn *et al.* 2016). In Year 4, an initial screening analysis of the water quality variables showed that alkalinity (or specific conductivity), DO, and water temperature are expected to be the most suitable predictor variables to include in statistical models to quantify the effect of water quality on juvenile fish abundance (Sharron *et al.* 2018), although all variables that are monitored as part of JHTMON-8 will be considered. The Year 4 screening analysis generally showed that inter-annual variability in many of the water quality variables was low, which may limit the power of the final analysis to quantify potential effects of water quality (should effects be present) on fish abundance. As an alternate line of evidence, it will therefore be important to also continue to evaluate water quality results in the context of WQG-AL to make inferences about the potential for water quality to limit juvenile fish abundance in the Quinsam River.

The analysis will initially involve evaluating scatter-plots, time series graphs, and correlation metrics to examine whether there is a link between variability in water quality variables and juvenile fish abundance, while critically examining biological relevance and significance (e.g., based on effect size).

5.4. H_04: Juvenile Fish Abundance is Not Correlated with the Occurrence of Flood Events

This hypothesis will be tested by extending the analysis undertaken in Year 8 (Section 4.5) to further analyze the potential effects of high flow metrics on juvenile fish recruitment. Furthermore, we propose to extend the analysis to consider hydrologic variability more widely (discussed in Section 1.5.5). Analysis will be completed using a subset of Indicators of Hydrologic Alteration (Richter *et al.* 1996), which were identified in Year 3. Candidate metrics include measures of both high and low flows to provide an opportunity to extend the analysis to consider hydrologic variability more widely, reflecting that the occurrence of low summer flows can be a significant limiting factor for juvenile salmonid productivity (e.g., Grantham *et al.* 2012), in addition to the occurrence of floods. Following the collation of a historical dataset collected at the Quinsam Hatchery fence, we also plan to extend the analysis of H_04 to consider years prior to JHTMON-8, substantially increasing statistical power.

5.5. H_05: Juvenile Fish Abundance is Not Correlated with Food Availability

Relationships between fish abundance and invertebrate drift will be examined in Year 10. To test H_05, we plan to examine whether (and if so, what amount of) variability in invertebrate drift biomass explains variability in species-specific spawner recruitment curves or juvenile fish abundance (e.g., Steelhead) for JHTMON-8 priority species. H_05 will be assessed based on the magnitude of the effect size to infer biological significance. We plan to use both total invertebrate biomass and EPT invertebrate biomass (first quantified in Year 7) as key predictor variables. Furthermore, we plan to trial invertebrate density as a secondary measure of food abundance; however, consistent with the TOR (BC Hydro 2018a), we expect to use invertebrate biomass as the main measure of food availability because it is a direct measure of the energy available for fish to consume.

If strong relationships are detected between fish abundance and invertebrate biomass/density, then we may conduct inferential statistical analysis (modelling) of invertebrate diversity metrics (family richness and Simpson's diversity index) to provide greater insight. As discussed in Section 1.5.6, salmonids can preferentially forage on certain taxa and therefore it is plausible that changes to invertebrate community composition could affect food quality by changing foraging opportunities. However, a clear link between invertebrate diversity and fish productivity is not well-established in the literature and therefore, at this stage, the main purpose of evaluating invertebrate community composition and diversity is to provide a more general understanding of the invertebrate food available to rearing fish.

Variability in invertebrate drift biomass among years is generally low (Figure 16); therefore, as for some water quality metrics (discussed above in Section 4.4), this may limit the statistical power of the analysis conducted in Year 10; i.e., without a clear gradient in invertebrate drift biomass among years, it will be challenging to quantify how variability in this metric affects annual estimates of juvenile fish abundance.

Therefore, as an alternate line of evidence, it will be useful to also compare invertebrate drift biomass and benthic invertebrate biomass (based on kick net sampling) for the Quinsam River with benchmarks such as measurements collected at other streams to inform conclusions about whether a lack of invertebrate drift biomass is expected to limit juvenile fish abundance in the Quinsam River. As with water quality, the study is currently premised on the assumption that invertebrate drift measured at a single index site is representative of conditions experienced by fish in the wider watershed.

5.6. H_06: Annual Smolt Abundance is Not Correlated with the Number of Adult Returns

Updated stock-recruitment relationships will be used in Year 10 to test H_06, i.e., to confirm whether the abundance of outmigrating juveniles is correlated with the abundance of corresponding prior adult returns. Stock-recruitment relationships can then be used in the analysis to test the remaining hypotheses, i.e., to quantify whether variability in the environmental factors can explain variability in the stock-recruitment relationships (assuming such relationships are present; Lawson *et al.* 2004). Such consideration of the potential influence of adult returns on juvenile fish abundance is important to avoid misleading inferences about the role of environmental factors in driving population fluctuations (Walters and Ludwig 1981).

Development of stock-recruitment relationships will extend the work initiated in Year 7 (Suzanne *et al.* 2021), as summarized in Section 4.7. At a minimum, we propose to test H_06 separately for Chinook Salmon, Coho Salmon and Pink Salmon. Quantitative analyses are not proposed to test H_06 for Steelhead because adult abundance is not monitored on the Quinsam River. Instead, we propose to adopt a qualitative approach to assess Steelhead by evaluating historical data and information relevant to BC watersheds more widely (e.g., Lill 2002) to consider whether estimated Steelhead smolt production indicates that the Quinsam River is "fully seeded" for this species, which would indicate that additional adult returns would not affect smolt production.

5.7. Additional Task for Year 9 (2022)

Each year, we have undertaken additional analyses to streamline final hypothesis testing in Year 10, consistent with an evaluation of the study design undertaken during Year 1 (Abell *et al.* 2015). In Year 9, we plan to prepare available data for predictor variables (e.g., WUA, hydrological metrics) and further extend the preliminary analysis described in Section 4.5, prior to final analysis in Year 10.

REFERENCES

Abell, J., A. Buren, P. Little, J. Carter, D. Durston, S. Sharron, J. Krogh, K. Healey, and T. Hatfield. 2019. JHTMON-8: Quinsam River Smolt and Spawner Abundance Assessments – Year 5 Interim Summary Report. Draft V1. Consultant's report prepared for BC Hydro by Laich-Kwil-Tach Environmental Assessment Ltd. Partnership and Ecofish Research Ltd., July 2, 2019.

Abell, J., J. Bruce, and T. Hatfield. 2020. Quinsam River Diversion Facility: Total Dissolved Gas Risk Assessment – Version 3. Consultant's report prepared for BC Hydro by Ecofish Research Ltd. and Creekside Aquatic Sciences Ltd. March 18, 2020.

Abell, J., S. Sharron, P. Dinn, J. Ellenor, M. Thornton, and T. Hatfield. 2017. JHTMON-8: Salmon River and Quinsam River Smolt and Spawner Abundance Assessments – Year 3 Annual Monitoring Report. Consultant's report prepared for BC Hydro by Laich-Kwil-Tech Environmental Assessment Ltd. Partnership and Ecofish Research Ltd., November 9, 2017.

Abell, J., T. Jensma, H. Wright, J. Ellenor, and T. Hatfield. 2015. JHTMON-8: Salmon River and Quinsam River Smolt and Spawner Abundance Assessments – Year 1 Annual Monitoring Report. Consultant's report prepared for BC Hydro by Laich-Kwil-Tech Environmental Assessment Ltd. Partnership and Ecofish Research Ltd., April 16, 2015.

Armanini, D.G., N. Horrigan, W.A. Monk, D.L. Peters, and D.J. Baird. 2011. Development of a benthic macroinvertebrate flow sensitivity index for Canadian rivers. River. Res. Applic. 27: 723-737.

Ashley, K. and P.A. Slaney. 1997. Fish habitat rehabilitation procedures. Watershed Restoration Technical Circular No. 9, Chapter 13, Accelerating recovery of stream, river and pond productivity by low-level nutrient replacement. Available online at: https://www.for.gov.bc.ca/hfd/library/FFIP/Slaney_PA1997_A.pdf. Accessed on October 3, 2022.

BC Hydro. 2012. Campbell River System Water Use Plan Revised for Acceptance by the Comptroller of Water Rights. November 21, 2012 v6. 46 p.

BC Hydro. 2018a. JHTMON-8 – Quinsam River Smolt and Spawner Abundance Assessments Monitoring Program Terms of Reference Revision 1. 21 p.

BC Hydro. 2018b. JHTMON-6 – Campbell Watershed Riverine Fish Production Assessment Monitoring Program Terms of Reference Revision 1. January 25, 2018. 31 p.

Burt, D.W. 2003. Fisheries and Aquatic Resources of the Quinsam River System. Report prepared for BC Hydro and Power Authority, Burnaby, BC. 171 p.

Campbell River WUP CC (Campbell River Water Use Plan Consultative Committee). 2004. Consultative Committee Report. March 2004. 350 p.

Carter, K. 2005. The Effects of Temperature on Steelhead Trout, Coho Salmon, and Chinook Salmon Biology and Function by Life Stage – Implications for Klamath Basin TMDLs. Report prepared by the California Regional Water Quality Control Board. 27 p. Available online at: https://www.waterboards.ca.gov/northcoast/water_issues/programs/tmdls/shasta_river/060707/28appendixaetheeffectsoftemperatureonsteelheadtroutcohosalmonandchinooksalmonbiologyandfunction.pdf. Accessed on October 3, 2022.

Cattanéo, F., N. Lamouroux, P. Breil and H. Capra. 2002. The influence of hydrological and biotic processes on brown trout (*Salmo trutta*) population dynamics. Canadian Journal of Fisheries and Aquatic Sciences 59:12–22.

CCME (Canadian Councils of Ministers of the Environment). 2004. Canadian water quality guidelines for the protection of aquatic life: Phosphorus: Canadian Guidance Framework for the Management of Freshwater Systems. In: Canadian environmental quality guidelines, 2004, Canadian Council of Ministers of the Environment, Winnipeg. Available online at: http://ceqg-rcqe.ccme.ca/download/en/205/. Accessed on October 3, 2022.

Clark, M.J.R. (editor). 2013. British Columbia Field Sampling Manual. Water, Air and Climate Change Branch, Ministry of Water, Land and Air Protection, Victoria, BC, Canada. 312 pages. Available online at: https://www2.gov.bc.ca/assets/gov/environment/research-monitoring-and-reporting/monitoring/emre/bc_field_sampling_manual_complete.pdf. Accessed on October 3, 2022.

Coleman, M.A. and K.D. Fausch. 2007. Cold summer temperature limits recruitment of age-0 Cutthroat Trout in high-elevation Colorado streams. Transactions of the American Fisheries Society 136:1231–1244.

DFO (Fisheries and Oceans Canada). 2017. Quinsam River Hatchery. Available online at: http://www.pac.dfo-mpo.gc.ca/sep-pmvs/projects-projets/quinsam/quinsam-eng.html. Accessed on November 25, 2021.

DFO (Fisheries and Oceans Canada). 2021. New Salmon Escapement Database. Available online at: https://open.canada.ca/data/en/dataset/c48669a3-045b-400d-b730-48aafe8c5ee6. Accessed on October 3, 2022.

Dinn, P., J. Abell, and T. Hatfield. 2016. JHTMON-8: Compilation and Review of Historical Water Quality Data for the Salmon and Quinsam Rivers. Consultant's report prepared for BC Hydro by Laich-Kwil-Tach Environmental Assessment Ltd. Partnership and Ecofish Research Ltd.

Environment and Climate Change Canada. 2021. Environment and Climate Change Canada warns western Canadians about dangerous record-high temperatures. News release, June 29, 2021. Available online at: https://www.canada.ca/en/environment-climate-change/news/2021/06/environment-and-climate-change-canada-warns-western-canadians-about-dangerous-record-high-temperatures.html. Accessed on October 3, 2022.

Environment Canada. 2012. Canadian Aquatic Biomonitoring Network Field Manual Wadeable Streams. April 2012 version. 57 p.

Ewart, D. and S. Kerr. 2014. Quinsam River Juvenile Downstream Migration BC Hydro Water Use Plan Monitoring Study #8. 10 pages.

Gibbins, C., J. Shellberg, H. Moir, and C. Soulsby. 2008. Hydrological Influences on Adult Salmonid Migration, Spawning, and Embryo Survival in *Salmonid Spawning Habitat in Rivers: Physical Controls, Biological Responses, and Approaches to Remediation* (Eds: D. A. Sear and P. Devrie). American Fisheries Society. 376 p.

Grantham, T.E., D.A. Newburn., M.A. McCarthy, and A.M. Merenlender. 2012. The role of streamflow and land use in limiting over summer survival of juvenile steelhead in California streams. Transactions of the American Fisheries Society 141: 585–598.

Hatfield, T., A.F. Lewis, and S. Babakaiff. 2007. Guidelines for the collection and analysis of fish and fish habitat data for the purpose of assessing impacts from small hydropower projects in British Columbia. Prepared by Solander Ecological Research Ltd. and Ecofish Research Ltd., for the BC Ministry of the Environment, March 9, 2007. 21 p. + appnds.

Hopcroft, R.R. 1991. Zooplankton biomass digitizing program. Version 1.3. Zoology Department, Guelph University.

Johnson, J.H. and N.H. Ringler. 1980. Diets of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*) relative to prey availability. Canadian Journal of Zoology 58: 553–558.

Lawson, P.W., E.A. Logerwell, N. J. Mantua, R.C. Francis, and V.N. Agnostini. 2004. Environmental factors influencing freshwater survival and smolt production in Pacific Northwest coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 61: 360–373.

Lewis, A., A. Harwood, C. Zyla, K.D. Ganshorn, and T. Hatfield. 2013. Long term Aquatic Monitoring Protocols for New and Upgraded Hydroelectric Projects. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/166. ix + 88p. 97 p.

Lewis, A., T. Hatfield, B. Chilibeck, and C. Roberts. 2004. Assessment methods for aquatic habitat and instream flow characteristics in support of applications to dam, divert, or extract water from streams in British Columbia. Report prepared for the Ministry of Water, Land and Air Protection and the Ministry of Sustainable Resource Management. 110 p.

Lill, A.F. 2002. Greater Georgia Basin Steelhead Recovery Action Plan. Prepared for the Pacific Salmon Foundation in conjunction with the Ministry of Water, Land and Air Protection. 107 p.

Lirette, M.G., R.S. Hooton, and W.R. Olmsted. 1985. A summary of wild steelhead smolt enumeration and habitat productivity studies at Quinsam River, 1976–1984. Fish. Tech. Circ. No. 67. 17 p.

Malick, M.J., S.P. Cox, F.J. Mueter, B. Dorner, and R.M. Peterman. 2017. Effects of the North Pacific Current on the productivity of 163 Pacific salmon stocks. Fish. Oceanogr., 26: 268-281. https://doi.org/10.1111/fog.12190.

Marriner, A., M. Thornton, T. Hatfield, H. Wright, E. Vogt, E. Smyth, I. Murphy, and J. Abell. 2020. JHTMON-6 Campbell Watershed Riverine Fish Production Assessment – Component 2 Fish Passage Study Final Program Summary Report. Consultant's report prepared for BC Hydro by Laich-Kwil-Tach Environmental Assessment Ltd. Partnership and Ecofish Research Ltd., April 13, 2020.

Martens, K., I. Schön, C. Meisch, and D.J. Horne. 2008. Global diversity of ostracods (Ostracoda, Crustacea) in freshwater. Hydrobiologia 595: 185–193.

Matthews, S.B. and F.W. Olson. 1980. Factors affecting Puget Sound coho salmon (*Oncorhynchus kisutch*) runs. Canadian Journal of Fisheries and Aquatic Sciences 37: 1373–1378.

McNichol, R.E., E. Scherer, and E.J. Murkin. 1985. Quantitative field investigations of feeding and territorial behaviour of young-of-the-year brook charr, *Salvelinus fontinalis*. Environmental Biology of Fishes 12: 219–229.

MOE (B.C. Ministry of Environment). 1997. Ambient water quality criteria for dissolved oxygen: overview report. Prepared pursuant to Section 2(e) of the *Environment Management Act*, 1981. Signed by Don Fast, Assistant Deputy Minister, Environment Lands HQ Division. Available online at: https://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/waterquality/water-quality-guidelines/approved-wqgs/dissolvedoxygen-or.pdf. Accessed on October 3, 2022.

Nordin, R.N. and Pommen L.W. 2009. Water Quality Guidelines for Nitrogen (Nitrate, Nitrite, and Ammonia) Overview Report Update. Water Stewardship Division Ministry of Environment Province of British Columbia. Available online at: https://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/waterquality/water-quality-guidelines/approved-wqgs/bc_env_nitrate_waterqualityguideline_overview.pdf. Accessed on October 3, 2022.

Oliver, G.G. and L.E. Fidler. 2001. Towards a water quality guideline for temperature in the Province of British Columbia. Prepared for Ministry of Environment, Lands and Parks, Water Management Branch, Water Quality Section, Victoria, B.C. Prepared by Aspen Applied Sciences Ltd., Cranbrook, B.C., 53 pp + appnds. Available online at: https://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/waterquality/water-quality-guidelines/approved-wqgs/temperature-tech.pdf. Accessed October 3, 2022.

Ptolemy, R. and A. Lewis. 2002. Rationale for Multiple British Columbia Instream Flow Standards to Maintain Ecosystem Function and Biodiversity. Draft prepared for Ministry of Water, Land and Air Protection Ministry of Sustainable Resource Management. 48 p.

Quinn, T.P. 2005. The behavior and ecology of Pacific salmon and trout. University of Washington Press. 378 p.

R Core Team. 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available online at: https://www.R-project.org/.

Rader, R.B. 1997. A functional classification of the drift: traits that influence invertebrate availability to salmonids. Canadian Journal of Fisheries and Aquatic Sciences 54: 1211-1234.

Richter, B.D., J.V. Baumgartner, J. Powell, and D.P. Braun. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology 10: 1163-1174.

Ricker, W.E. 1954. Stock and recruitment. Journal of the Fisheries Research Board of Canada. 11(5): 559–623.

RISC (Resources Information Standards Committee). 1997a. Guidelines for Designing and Implementing a Water Quality Monitoring Program in British Columbia. Prepared by the BC Ministry of Environment, Lands and Parks for the Resource Inventory Commission. Available online at: https://www2.gov.bc.ca/assets/gov/environment/natural-resource-stewardship/nr-laws-policy/risc/guidelines_for_designing_and_implementing_a_water_quality_monitoring_program_in_british_columbia.pdf. Accessed on October 3, 2022.

RISC (Resources Information Standards Committee). 1997b. Guidelines for Interpreting Water Quality Data. Prepared by the BC Ministry of Environment, Lands and Parks for the Resource Inventory Commission. Available online at: https://www2.gov.bc.ca/assets/gov/environment/natural-resource-stewardship/nr-laws-policy/risc/guidlines_for_interpreting_water_quality_data.pdf. Accessed on October 3, 2022.

RISC (Resources Information Standards Committee). 2003. Ambient Freshwater and Effluent Sampling Manual. Prepared by the Resource Inventory Standards Committee. Available online at: https://www2.gov.bc.ca/assets/gov/environment/natural-resource-stewardship/nr-laws-policy/risc/part_e_water_and_wastewater_sampling_ambient_freshwater_and_effluent_sampling_simulate_template.pdf. Accessed on October 3, 2022.

Scheuerell, M.D., C.P. Ruff, J.H. Anderson, and E.M. Beamer. 2021. An integrated population model for estimating the relative effects of natural and anthropogenic factors on a threatened population of steelhead trout. Journal of Applied Ecology 58: 114–124.

Sharron, S., L. Shelley, M. Thornton, J. Ellenor, J. Carter, J. Abell, and T. Hatfield. 2018. JHTMON-8: Salmon River and Quinsam River Smolt and Spawner Abundance Assessments – Year 4 Annual Monitoring Report Draft V2. Consultant's report prepared for BC Hydro by Laich-Kwil-Tach Environmental Assessment Ltd. Partnership and Ecofish Research Ltd., July 3, 2018.

Simpson, E.H. 1949. Measurement of diversity. Nature, 163: 688.

Singleton, H.J. 1985. Water Quality Criteria for Particulate Matter: Technical Appendix. Ministry of the Environment Lands and Parks, Victoria, B.C. pp.82-82.

Slaney, P.A. and B.R. Ward. 1993. Experimental fertilization of nutrient deficient streams in British Columbia. Pages 128-141 in G. Shooner et S. Asselin [ed.] Le developpement du Saumon atlantique au Quebec: connaitre les regles du jeu pour reussir. Colloque international de la Federation quebecoise pour le saumon Atlantique. Quebec, decembre 1992. Collection *Salmo salar* no 1: 201p.

Solander (Solander Ecological Research), D. Burt and Associates, MJL Environmental Consultants and Westland Resource Group. 2004. Quinsam River Detailed Biophysical Assessment V3. May 10, 2004. Report prepared for BC Hydro. 104 p.

Strachan, S., A. Ryan, H. McDermott, and C. MacKinlay. 2009. Benthic invertebrate and water quality assessment of the Quinsam River watershed in British Columbia 2001-2006. Environment Canada, Science and Technology Branch, Vancouver, BC. Available online at: http://publications.gc.ca/collections/collection_2011/ec/En84-84-2011-eng.pdf. Accessed on December 22, 2021.

Suzanne, C., A. Buren, D. Greenacre, J. Abell, K. Akaoka, and T. Hatfield. 2021. JHTMON-8: Quinsam River Smolt and Spawner Abundance Assessment – Year 7 Annual Report. Consultant's report prepared for BC Hydro by Laich-Kwil-Tach Environmental Assessment Ltd. Partnership and Ecofish Research Ltd., December 16, 2021.

Suzanne, C., A. Buren, J. Abell, and T. Hatfield. 2020. JHTMON-8: Quinsam River Smolt and Spawner Abundance Assessment – Year 6 Annual Report. Consultant's report prepared for BC Hydro by Laich-Kwil-Tach Environmental Assessment Ltd. Partnership and Ecofish Research Ltd., September 20, 2021.

Walters, C.J. and D. Ludwig. 1981. Effects of measurements on the assessment of stock-recruitment relationships. Canadian Journal of Fisheries and Aquatic Sciences 38: 704-710.

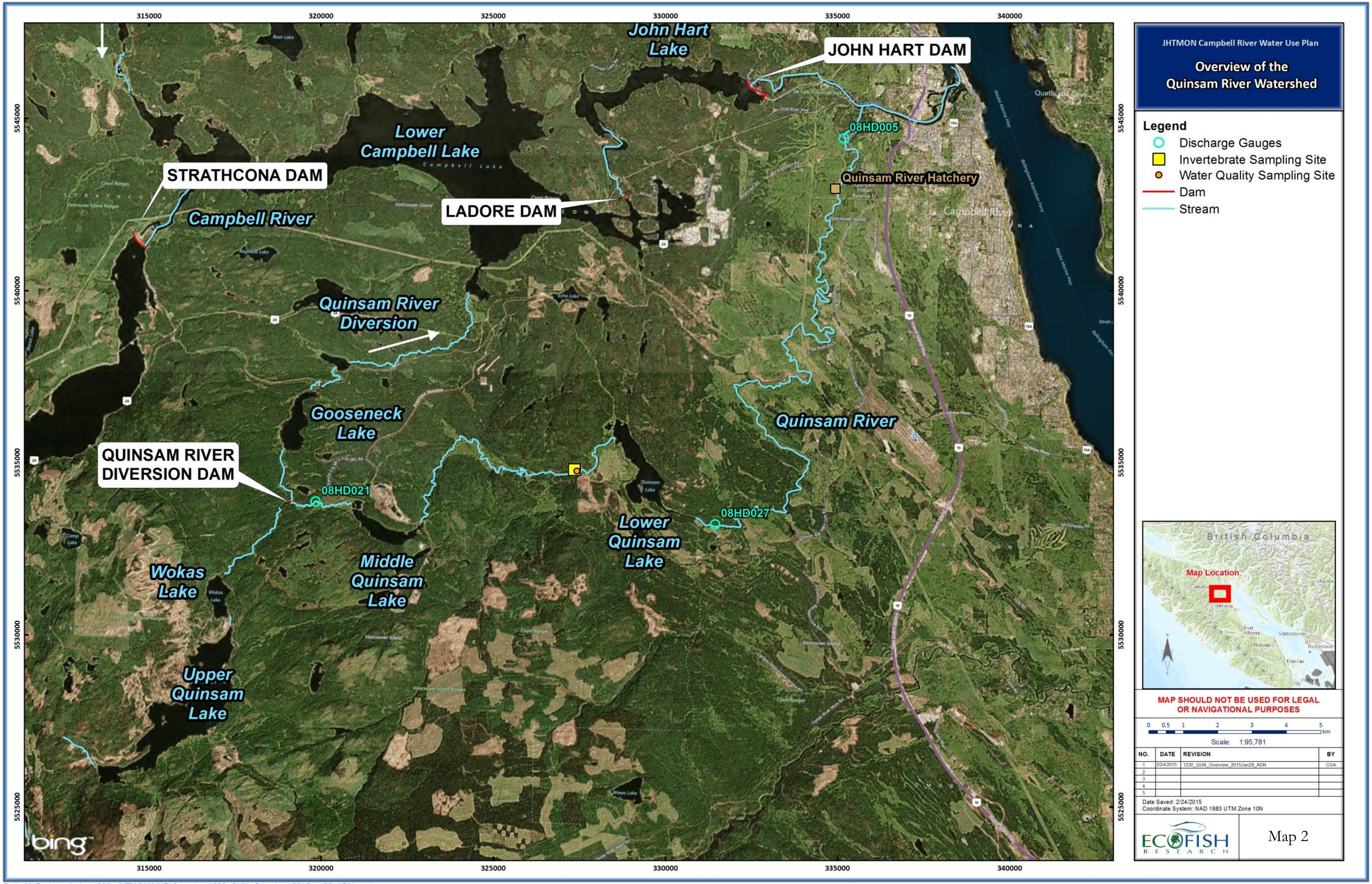
Personal Communications

Armanini, D. 2013. Managing Director, Prothea Group, Milano, Italy. Personal communication. Email correspondence with K. Ganshorn, Ecofish Research Ltd., January 2013.

Fortkamp, E. 2019. Enhancement Technician, Quinsam River Hatchery, Fisheries and Oceans Canada. Email correspondence with J. Abell, Ecofish Research Ltd., November 2019.

Fortkamp, E. 2022. Enhancement Technician, Quinsam River Hatchery, Fisheries and Oceans Canada. Email correspondence with J. Abell, Ecofish Research Ltd., January 2022.

PROJECT MAP



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APPENDICES