Campbell River Project Water Use Plan

Quinsam and Salmon Rivers Smolt and Spawner

Abundance Assessment

Implementation Year 6

Reference: JHTMON-08

Annual Monitoring Report

Study Period: April 1, 2019 to March 31, 2020

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

September 20, 2021

JHTMON-8: Quinsam River Smolt and Spawner Abundance Assessment

Year 6 Annual Monitoring Report

Prepared for:

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Title page photographs – top left: LKT technician undertaking a mark recapture study at Quinsam Hatchery salmon counting fence (June 06, 2019); top right: drift nets deployed on the Quinsam River (June 20, 2019); bottom left: sampling drift nets on the Quinsam River (September 12, 2019); bottom right: LKT technician sampling at Quinsam Hatchery salmon counting fence (June 06, 2019).

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EXECUTIVE SUMMARY

Water Use Plans (WUPs) were developed for BC Hydro's hydroelectric facilities through a consultative process. As the Campbell River WUP process reached completion, a number of 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 JHTMON-8 is to understand whether BC Hydro operations, through changes to streamflow, have potential to change fish abundance. To address these objectives, JHTMON-8 monitors fish abundance and a range of environmental factors (Table i). Final data analysis will involve examining links between fish abundance and environmental factors to better understand what limits 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	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 six years of data collection (Table i) have now been completed. In Year 10, 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 data for additional Pacific salmon species, as well as collecting abundance data for outmigrating juvenile life stages of a range of species at the Quinsam Hatchery salmon counting fence.

Fish abundance data collected in Year 1–6 in the Quinsam River show that fish abundance has varied, including for the three primary species. For example, annual outmigration data provided by DFO vary among years, with variability greatest for wild Chinook Salmon (~600 to ~114,000 fry) and smaller for wild Coho Salmon (~22,000 to ~47,000 smolts) and steelhead (~3,000 to ~11,000 smolts). Wild Chinook Salmon outmigration was substantially higher in Years 4–6 (three-year average = 102,658 fish) than in Years 1–3 (three-year average = 6,978 fish). Historical data collected at the Quinsam Hatchery salmon counting fence since the 1970s were collated in Year 5 and these data will be integrated into the final JHTMON-8 analysis in Year 10 to increase the statistical power to quantify variability in juvenile fish abundance in the Quinsam River. In Year 6, we completed a review of capture efficiency estimates calculated based on juvenile mark-recapture experiments conducted at the Quinsam Hatchery salmon counting fence. This review characterized trends in capture efficiency estimates and we plan to use the results to make minor modifications to how estimates are applied in order to improve the accuracy of results.

In Year 5, we quantified the Weighted Usable Area (WUA; in m^2) to provide a measure of habitat availability (*H_02*) for different life stages of priority species. Further analysis of WUA was not undertaken in Year 6, although the WUA calculations will be updated in Year 10.

Water quality data (*H_03*) collected at an index site in 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, maximum mean weekly maximum water temperature measured in Year 6 (19.8ºC) exceeded the upper limit of the optimum temperature ranges for the rearing life stage of species such as juvenile Coho Salmon (upper limit of 16ºC) and Rainbow Trout (upper limit of 18ºC). Low dissolved oxygen concentrations were occasionally recorded that were below guideline ranges for the protection of buried embryos/alevins of some species. 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, which may limit the power of the final analysis to quantify effects of water quality (if present) on fish abundance. It will therefore be important to continue to evaluate water quality results in the context of guidelines to support qualitative conclusions regarding *H_03*.

To test *H_04* (floods), flow data collected by the Water Survey of Canada were used to calculate a range of hydrological metrics based on a subset of the Indicators of Hydrologic Alteration (Richter *et al.* 1996). These metrics will be used to examine whether hydrologic variability among years affects juvenile fish abundance. Key observations to date include the occurrence of notable floods (>80 m^3/s) in December 2014 and November 2016, and the occurrence of low discharge (<1 m^3/s) each year during the summer period when the diversion facility was not operating.

Invertebrate drift biomass (H_05) in the Quinsam River generally tends to decline during the growing season, with distinct communities present early in the growing season (May and June) relative to later in the growing season. In Year 6, it was notable that invertebrate biomass was low relative to previous years (average 2019 biomass was ~37% lower than the average of all data collected in previous years); the reason for this is uncertain, although low flow in the Quinsam River in spring 2019 due to low precipitation may have been a contributing factor. In Year 10, we will examine the relationship between invertebrate biomass (i.e., fish food) and juvenile fish abundance to test H_05. Interannual variability in invertebrate biomass has so far been generally low, despite the observations described above.

Pacific salmon escapement data collected by DFO have been compiled and analyzed each year to test H_06 (adult returns). In Year 6, data were available for the period to 2018 when, consistent with previous years, Pink, Coho and Chinook salmon were the most abundant returning species. Escapement of Chinook Salmon in the Quinsam River in 2018 (6,774) was above the mean value (4,270) for the period of record (1953–2018), whereas estimated escapement of Coho Salmon in 2018 (10,025) was close to the mean value (12,165) of the dataset. Pink Salmon escapement in the Quinsam River in 2018 (95,836) was lower than the mean value for the dataset (129,713) but it substantially exceeded the median (50th percentile) value (31,390). The estimated Chum Salmon escapement in 2018 (69) was particularly low as it was the 7th lowest count recorded in the 59-year dataset, although the count in 2017 (50) was lower, being the 5th lowest count. In Year 10, we propose to construct spawner-recruitment relationships for priority species to quantify the relationship between the abundance of adult fish and the subsequent recruitment of juvenile fish each year.

Table ii. Status of JHTMON-8 objectives, management questions and hypotheses after Year 6.

Study Objective	Management Questions	Management Hypotheses	Year 6 (2019/2020) 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	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.	*$H_0$1: Annual population abundance does not vary with time (i.e., years) over the course of the Monitor*	-Juvenile fish have been sampled annually at the Quinsam Hatchery salmon counting fence to derive total outmigration estimates -In Year 5, we worked with DFO to collate data for the period since the 1970s, which will substantially improve statistical power to test H01. In Year 6, we conducted a review of capture efficiency estimates. -Inter-annual variability has been observed in the abundance of priority species so we expect to reject this hypothesis in Year 10
	2. Have WUP-based operations changed the influence of these primary factors on fish abundance, allowing carrying capacity to increase?	*$H_0$2: 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 6; relationships will be updated in Year 10 for the final analysis. -Analysis will be undertaken to test this hypothesis in Year 10

Study Objective	Management Questions	Management Hypotheses	Year 6 (2019/2020) Status
abundance in the Quinsam River.	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_0$3: 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 -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_0$4: 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 -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 (Pacific salmon incubation) -Analysis will be undertaken to test this hypothesis in Year 10

Study Objective	Management Questions	Management Hypotheses	Year 6 (2019/2020) Status
		$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 -Clear seasonal patterns have been observed but inter-annual variability in mean invertebrate drift biomass is less clear -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
		$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 and will be used to construct spawner-recruitment curves 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 Water Use Plan process reached completion, a number of 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, environmental perturbations or 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 (Map 1). 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.

Project Overview

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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).

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												
	Emergence												
	Rearing												
	Juvenile migration				F	S							
Chum Salmon	Adult migration												
	Spawning											P P	
	Incubation												
	Emergence												
	Juvenile migration			F									
Coho Salmon	Adult migration												
	Spawning											P P P	
	Incubation												
	Emergence												
	Rearing												
	Juvenile migration			F	S								
Pink Salmon	Adult migration												
	Spawning									P			
	Incubation												
	Emergence												
	Juvenile migration			F									
Rainbow Trout	Adult migration												
	Spawning												
	Incubation												
	Rearing												
	Juvenile migration												
Sockeye Salmon	Adult migration												
	Spawning												
	Incubation												
	Emergence												
	Rearing												
	Juvenile migration			F									
Steelhead (winter run)[1]	Adult migration												
	Spawning			P P									
	Incubation												
	Emergence												
	Rearing												
	Juvenile migration				S								

Critical times

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

[1] There are no summer run Steelhead on the Quinsam River.

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 (being 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 out 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.

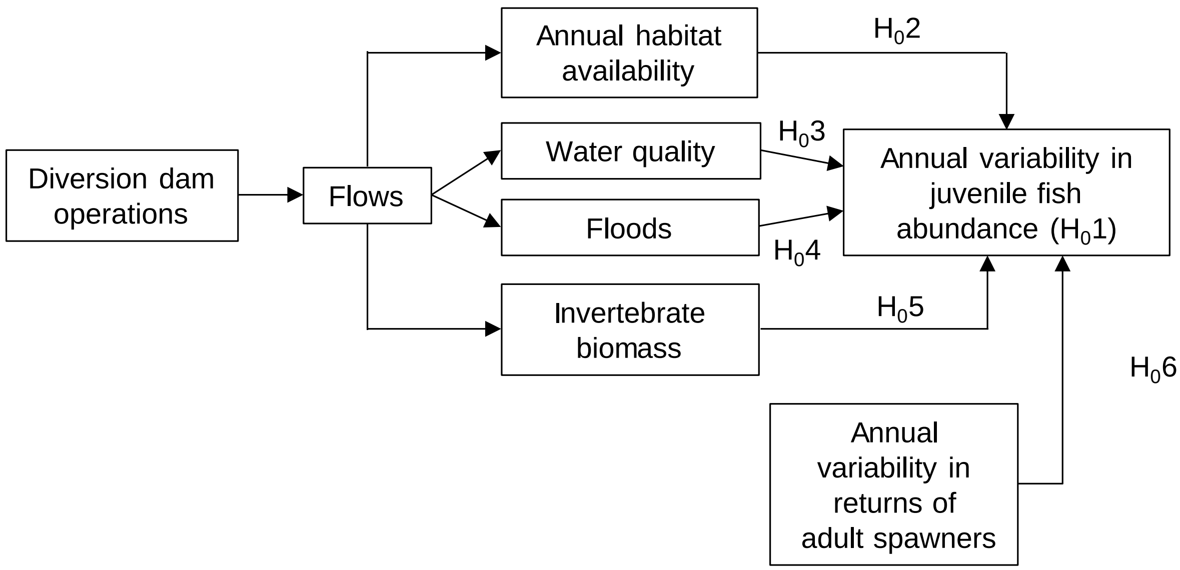
H_05 Annual population abundance is not correlated with food availability as measured by aquatic invertebrate sampling.

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 have been 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 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 for both rivers as part of JHTMON-8. Water

[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.

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). This hypothesis will be tested for the Quinsam River, where the salmon counting fence is monitored to provide estimates of total juvenile fish out-migration. 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. Testing $H_0$6 will involve comparing the productivity of naturally-spawned Coho and Chinook salmon with the productivity of colonization programs that out-plant juvenile fish to areas in the upper Quinsam River watershed, e.g., Lower Quinsam Lake. This comparison will further help to examine whether spawning areas are fully seeded. This will need to consider the potential for lower fitness of hatchery-reared fish compared with wild fish, as has been observed during previous field studies in the watershed (Burt, pers. comm. 2016).

For at least some species and life stages, we anticipate that 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 the remaining hypotheses to determine whether there are any 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).

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_0$2: *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_0$2, 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_0$3: '*annual population abundance is not correlated with water quality'* (Section 1.4). Approaches to incorporate water quality data into 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 there is a relationship between fish abundance and water quality. 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 objective of this aspect of the study is that water quality data are collected that suitably reflect variability of water quality in time and space and are representative of the conditions experienced by fish communities. A single mainstem index site was selected on 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; these 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

[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).

opportunities shortly after emergence (reviewed in Gibbins *et al*. 2008). Discharge data are collected at numerous sites on both study streams 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_0$4: '*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 this, we quantified the maximum daily mean discharge each year that occurs during the spawning and incubation periods of key species on both study streams. In future years, we will consider calculating additional metrics (e.g., based on the duration of high flows), which can be easily calculated by modifying the existing code. Analysis will be undertaken in Year 10 to determine whether variability in these values explains variability in fish abundance, providing a test of $H_0$4. The proposed analysis will focus on the spawning and incubation life stages because these life stages have been shown to be particularly sensitive to the effects of high flows (e.g., Cattanéo *et al*. 2002). We recognize that there is a range of mechanisms by which high flows can affect these life stages (see list above); therefore, if $H_0$4 is rejected, it may be necessary to undertake further analysis to characterize the most sensitive periods and threshold flows at which high flow events adversely affect juvenile fish abundance. We also recognize that, although $H_0$4 specifically focuses on floods, other aspects of hydrological variability could affect juvenile fish productivity. For example, the occurrence of low flows during summer can potentially limit the abundance of juvenile fish species that rear in freshwater throughout the summer, e.g., Coho Salmon (Matthews and Olson 1980). Accordingly, we propose to calculate a range of annual minimum flow metrics for each stream so that this analysis can be extended to evaluate whether low flows affect juvenile fish abundance; further details are provided in Section 2.2.

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_0$5 *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 factors such as flow velocity or sedimentation. These changes can affect total invertebrate biomass and, thus food availability for fish. Further, 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 in the Quinsam River 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 sampling targets benthic invertebrates and is therefore less representative of the total abundance of food available to fish. However, kick sampling based on the 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. Although these estimates are collected as part of wider salmon stock assessment work, they provide important data to support JHTMON-8. The results of summer and fall 2018 surveys were finalized during Year 6. These 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 2018 surveys are summarized in Table 4 for the Quinsam River, based on information provided in the nuSEDS database (DFO 2020). 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 2018 salmon spawner escapement counts on the Quinsam River (DFO 2020). See Table 5 for descriptions of estimate classes.

	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	8-Aug	1-Sep	30-Aug	20-Jul	2-Aug
Date of last inspection	22-Nov	1-Dec	15-Dec	3-Nov	1-Dec
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 2020).

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

2.1.2.1. Juvenile Outmigration Monitoring in Year 6

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 rivers occurs either soon after emergence or a few months later. Those Chinook Salmon that rear for a full summer and winter before smolting are believed to do so in the estuary (Burt 2003). The strategies adopted by steelhead, Cutthroat Trout and

Coho Salmon are more variable, and emigration from the river varies from emigrating during the first spring to emigrating three years after emergence.

In Year 6, sampling was undertaken from March 11 to June 14, 2019. 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 out-migrate to the ocean (Figure 2). 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 were typically fished, while during the period of smolt migration five openings were typically fished (Forktamp, pers. comm. 2018). Pink Salmon fry typically migrate at night and therefore traps were set overnight from approximately 15:00 to 09:00 during sampling in March 11 to April 26. 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 four releases were undertaken on March 26, April 1 April 5, and April 11; a total of 18,509 fish were released (4,032–5,374 per experiment). Separate catch efficiency estimates were derived for Coho Salmon smolts based on two 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 15 (200 fish) and May 21 (300 fish), with a total of 500 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 and Chinook 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. Colonized fish are distinct from other hatchery-raised fish that are released at the hatchery at the smolt stage, or hatchery Chinook Salmon out-planted to incubation boxes in the Campbell River (which do not outmigrate past the hatchery counting fence). All transplanted Coho Salmon were marked with an adipose fin clip. The abundance of colonized Coho Salmon and Chinook Salmon were estimated following the same protocol, and assuming equal catchabilities as wild fish.

Quinsam Hatchery staff have out-planted salmon fry during each year of JHTMON-8 (in addition to previous years; Table 6). During 2010 and 2011, approximately 100,000 Coho Salmon fry were released; during 2014-2018 approximately 150,000 Coho Salmon fry were released in the Upper Quinsam Lake. 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 (Table 6). These releases will be considered in the final JHTMON-8 analysis (see Section 4).

Figure 2. LKT technician undertaking a mark-recapture study at Quinsam Hatchery salmon counting fence, June 2019.

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

Species	Life Stage	Waterbody	Year[1]	Date of Release	Number Released	Comments
Coho Salmon	Fry	Upper Quinsam River	2018	6 May - 7 May	159,336	
			2017	23 May - 6 Jun	139,570	
			2016	30 May - 1 Jun	146,547	
			2015	29 Apr - 20 May	167,030	
			2014	9 Jun - 13 Jun	157,661	
Chinook Salmon	Fry	Lower Quinsam Lake	2019	7 May - 8 May	207,736	
			2018	7 May - 8 May	215,952	
			2017	9 May - unknown day in May	207,319	
			2016	12 May - 13 May	147,549	
			2015	11 May - 12 May	217,603	First time in 10 years

[1] Note that DFO annually reports the number of outplanted Chinook Salmon that same year and the number of Coho Salmon outplanted the previous year

2.1.2.2. Review of Capture Efficiency Estimates

During data review, DFO Quinsam Hatchery staff noted there was high uncertainty regarding some capture efficiency values estimated in Year 6 (Fortkamp pers. comm. 2019). Specifically, DFO noted there was high variability in capture efficiency estimates calculated using mark-recapture experiments undertaken with Coho Salmon smolts in Year 6. DFO hypothesized that this variability was due to experimental error, such as trap malfunction, and that using these anomalous capture efficiency estimates without appropriate adjustment would inflate uncertainty in total outmigration estimates of JHTMON-8 priority species (Chinook Salmon, Coho Salmon, and steelhead).

DFO's communication prompted a review of Year 6 and historical capture efficiency estimates as part of data JHTMON-8 QA/QC procedures. This was informed by a phone conference between Ecofish staff and DFO Quinsam Hatchery staff on December 19, 2019, as well as associated e-mail discussions between DFO and Ecofish. The objective of the review was to examine variability in capture efficiency estimates to inform how to best use capture efficiency estimates in JHTMON-8 analysis to maximize the accuracy of the results. Analysis was undertaken to test the following hypotheses regarding variability in capture efficiency estimates:

i. There are intra-seasonal trends in capture efficiency estimates for fry and smolts.

ii. There is a long-term declining trend in capture efficiency estimates, possibly due to increasing predation of marked fish.

iii. Removal of booms in late 2014 affected capture efficiency estimates for fry and smolts.

Details of analysis methods are provided in Appendix A.

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 the guidelines of the British Columbia Field Sampling Manual (Clark 2013) and the Ambient Fresh Water and Effluent Sampling Manual (RISC 2003). QUN-WQ (Figure 3) 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. Coordinates, site elevation, and sampling dates (*in situ* and laboratory samples) for the site are provided in Table 7.

Water quality has been monitored during Year 1 through Year 6 at QUN-WQ, with monitoring scheduled to continue for the remainder of JHTMON-8. Water quality has been monitored once a month from May through October during each year (for a total of six times per year). 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 laboratory 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 6 are presented in Table 8 (*in situ*) and Table 9 (laboratory). Laboratory method detection limits (MDL) for each analyte occasionally differed (Table 9) due to matrix effects in the sample, or variations in laboratory analytical instruments.

Table 7. Quinsam River water quality index site (QUN-WQ) details and sampling dates, Years 1 to 6.

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

Figure 3. Looking upstream to QUN-WQ on September 12, 2019.

Table 8. Water quality variables measured *in situ* and meters used in Year 6.

Parameter	Unit	Meter
Water temperature	ºC	YSI Pro Plus, YSI 85
pH	pH units	YSI Pro Plus
Salinity	ppt	YSI Pro Plus, YSI 85
Conductivity	µS/cm	YSI Pro Plus, YSI 85
Specific conductivity	µS/cm	YSI Pro Plus, YSI 85
Oxidation reduction potential	mV	YSI Pro Plus, YSI 85
Dissolved oxygen	mg/L	YSI Pro Plus, YSI 85
Dissolved oxygen	% Saturation	YSI Pro Plus, YSI 85

Table 9. Variables analyzed in the laboratory by ALS Environmental and corresponding units and method detection limit (MDL).

Parameter	Unit	MDL
General Water Quality		
Specific conductivity	µS/cm	2
pH	pH	0.1
Total suspended solids	mg/L	1 to 3
Total dissolved solids	mg/L	10 to 20
Turbidity	NTU	0.1
Alkalinity, Total (as $CaCO_3$)	mg/L	1 to 2
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 from each meter 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.

In Year 6, one field blank and one trip blank were collected on May 13, 2019. Values for all parameters for both blanks were below the respective MDLs. Overall, for the sampling program on the Quinsam River, the total number of quality assurance/quality control (QA/QC) samples collected over six years (22 out of 72 samples, or 31%) exceeded recommendations; the BC field sampling manual recommends that 20% to 30% of samples consist of QA/QC samples (Clark 2013), while the RISC (1997a) manual recommends a minimum of 10% of samples consist of QA/QC samples.

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 Appendix B.

In Vancouver Island streams, concentrations of a number of 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 actual 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 B. Any results for water chemistry variables that approximated or exceeded WQG-AL, or ranges typical for BC, are noted in Section 3.1.3.

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 6 for the Quinsam River. Water temperature data have now been collected at the water quality index site for the period May 2014 to October 2019 for the Quinsam River. 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 have a resolution of 0.02°C. For most of the record duration, 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. 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 which 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 10) that were chosen based on the provincial WQG-AL (Oliver and Fidler 2001; Table 11). 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 10. Parameters calculated based on water and air temperature data.

Parameter	Description	Method of Calculation
Monthly water- and air- temperature statistics	Average, minimum, and maximum temperatures 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 from temperatures observed at or interpolated to 15-min intervals. The hourly rate of cahange was set to the difference between temperature data points that are separated by one hour and was assigned to the avarage time for these data points.
Degree days in growing season	The beginning of the growing season is defined as the beginning of the first week that average stream temperatures exceed and remain above 5°C; the end of the growing season is defined as the last day of the first week that average stream temperature dropped below 4°C (as per Coleman and Fausch 2007).	Daily average water temperatures were summed over this period (i.e., from the first day of the first week when weekly average temperatures reached and remained above 5°C until the last day of the first week when weekly average temperature dropped below 4°C)
Number of days with extreme daily-mean temperature	>18°C , >20°C , and <1°C	Total number of days with daily-mean water temperature >18°C , >20°C , and <1°C
MWMxT	Mean Weekly Maximum Temperature	A 1-week moving-average filter is applied to the record of daily-maximum water temperatures inferred from hourly data; e.g., if MWMxT = 15°C on August 1, 2008, this is the average of the daily-maximum water temperatures for the 7 days from July 29 to August 4. MWMxT is calculated for every day of the year.

Table 11. Water temperature guidelines for the protection of freshwater aquatic life (Oliver and Fidler 2001).

Category	Guideline
All Streams	the rate of temperature change in natural water bodies not to exceed 1°C/hr
Streams with Known Fish Presence	temperature metrics to be described by the mean weekly maximum temperature (MWMT)
	mean weekly maximum water temperatures should not exceed ±1°C beyond the optimum temperature range for each life history phase of the most sensitive salmonid species present
Streams with Bull Trout or Dolly Varden	maximum daily temperatures should not exceed 15°C
	maximum spawning temperature should not exceed 10°C
	preferred incubation temperatures should range from 2°C to 6°C
	±1°C change from natural condition[1]
Streams with Unknown Fish Presence	salmonid rearing temperatures not to exceed MWMT of 18°C
	maximum daily temperature not to exceed 19°C
	maximum temperature for salmonid incubation from June until August not to exceed 12°C

[1] provided natural conditions are within these guidelines, if they are not, natural conditions should not be altered (Deniseger, pers. comm. 2009).

2.3. Hydrology

The Water Survey of Canada measures discharge at multiple gauges on the Quinsam River (Map 2). Available discharge data collected since the start of the study were plotted to evaluate flow conditions at the following sites downstream of the diversion facility: 'Quinsam R. near Campbell R.' and 'Quinsam R. at Argonaut Bridge' sites (Table 12). To provide historical context, discharge was plotted alongside summary statistics (10th, 50th and 90th percentiles) for the periods of record. At the time of reporting, quality assured historical data were only available until the end of 2018.

In addition, several annual hydrological metrics were calculated using data for each gauge to quantify key flow characteristics that have potential to influence fish productivity (Table 13). The metrics quantify the occurrence of high flows during biologically sensitive periods of the year to support analysis to test $H_0$4, which relates to floods (Section 1.5.5). For Pacific Salmon species (fall spawners), the maximum discharge during the incubation period was calculated based on the discharge measured between the start of incubation in fall the previous year, and the end of incubation during spring of the current year. Low flow metrics were also calculated to support future analysis to test whether low summer flows affect the abundance of juvenile salmonids that rear in freshwater through the summer

(Coho Salmon and steelhead). All metrics are based on a subset (Group 2) of the Indicators of Hydrologic Alteration (Richter *et al*. 1996) that were developed to quantify the magnitude and duration of hydrological extremes. Metrics were either calculated based on annual records of mean daily discharge (m^3/s), or using records for the spawning and incubation periods of specific fish species, based on fish periodicity information for the Quinsam River reported by Burt (2003; Quinsam River). Metrics were calculated using the Indicators of Hydrologic Alteration package developed for R (R Core Team 2016) by The Nature Conservancy. Metrics were calculated based on discharge data collected at the gauges at Argonaut Bridge (08HD021) and near the confluence with the Campbell River (08HD005).

Table 12. 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 13. 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

2.4. Invertebrate Drift

2.4.1. Sample Collection

One invertebrate drift sampling site was established on the Quinsam River (Map 2, Figure 4), located close (<150 m) to the water quality index site. The site location was consistent among years; UTM coordinates (Zone 10) were: 309,304 E and 5,556,468 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 June in Year 6 (the month that is sampled weekly is rotated among study years to quantify the variance in monthly data). In total, sampling occurred on nine dates in the Quinsam River in Year 6 (Table 14).

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 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 4). 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. Any 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 14). 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 2–6. This is a method change 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 6, kick net sampling was also undertaken on September 12, 2019 at QUN-IV. The CABIN standardized sampling method was followed (MoE 2009), 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 10 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 14. Invertebrate drift sample timing and sampling duration at the Quinsam River site (QUN-IV) during Year 6. See annual reports for details of sampling in previous years.

Sample Date	Start Time[1]	Finish Time[2]	Sampling Duration[3,4] (hh:mm)
13-May-2019	07:11	11:11	4:00
06-Jun-2019	06:47	10:49	4:02
12-Jun-2019	06:41	10:42	4:01
20-Jun-2019	06:30	10:30	4:00
27-Jun-2019	06:48	10:48	4:00
11-Jul-2019	06:55	10:55	4:00
12-Aug-2019	07:07	11:07	4:00
12-Sep-2019	08:00	12:00	4:00
09-Oct-2019	09:00	13:01	4:01

[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 4. View across the stream from river right towards QUN-IV, August 12, 2019.

2.4.2. 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%.

2.4.3. 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) and biomass (mg dry weight) of each sample were expressed as units per m^3 of water, where 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. During Years 2–6, 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 Year 2–6. 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.

PRIMER (Plymouth Routines in Multivariate Ecological Research) v. 6 software was used to generate a Bray-Curtis similarity matrix for all samples collected from the Quinsam River during JHTMON-8. The similarity matrix was generated from square-root-transformed density data for aquatic, semi-aquatic, and terrestrial taxa at the highest taxonomic resolution available for each taxon. The square root transformation down-weights the effect of the most abundant taxa, allowing for a better representation of the invertebrate community as a whole, rather than having similarity measures dominated by only the most abundant taxa. The similarity matrix was generated by calculating a similarity coefficient for all possible pairs of sample dates with respect to the taxonomic composition and abundance of different taxa on both sample dates.

The resulting Bray-Curtis similarity matrix was then examined using cluster analysis dendrograms in PRIMER to detect similarities among samples, to help assess temporal variation in invertebrate community composition by identifying potential seasonal and interannual trends that may affect the quality of food available to fish. Specifically, the clustering method used is a hierarchical clustering with group-average linking. The method takes a Bray-Curtis similarity matrix as a starting point and successively fuses the samples into groups, and the groups into larger clusters. The method starts with the highest mutual similarities, and then gradually lowers the similarity level at which groups are formed. The significance level for clustering was set at 5% using the SIMPROF tool in PRIMER (1000 permutations were used to calculate the mean similarity profile and 999 to generate the null distribution of the departure statistic). Further discussion of the cluster analysis can be found in Clarke and Warwick (2001) and Clarke and Gorley (2006).

The Bray-Curtis similarity matrices were also examined using non-metric, multi-dimensional scaling (MDS) ordination plots in PRIMER to detect trends in similarity among samples. MDS uses an algorithm that successively refines the positions of the points (samples) until they satisfy, as closely as possible, the dissimilarity between samples (Clarke and Warwick 2001). This algorithm was repeated 1,000 times for each similarity matrix (i.e., with density on each date as samples). The result is a two-dimensional ordination plot in which points that are close together represent samples that are very similar in community composition with respect to the taxa present and their abundances. Conversely, points that are far apart represent samples with a very different community composition. Further discussion of the MDS analysis can be found in Clarke and Warwick (2001) and Clarke and Gorley (2006).

3. RESULTS

3.1. Fish Population Assessments

3.1.1. Quinsam River Salmon Escapement, 2018

Salmon escapement data for the Quinsam River are presented for 2018 (Year 5; Table 15), which are the most recent results available at the time of reporting. Summary statistics for the period of record are also provided in Table 15 to provide points of reference. Figure 5 presents salmon escapement data for the period of record.

Pink, Coho and Chinook salmon were the most abundant returning species in 2018, as well as historically (Table 15). Escapement of Chinook Salmon in the Quinsam River in 2018 (6,774) was above average, although the values in the late 2010s were lower than the values observed in the late 1980s, early 1990s and early 2000s (Chinook Salmon escapement in Year 1 (2,366 fish in 2014) was the lowest escapement recorded since 1995). The estimated escapement of Coho Salmon (10,025) in 2018 was approximately equal to the mean value (12,165) for the period of record (1953–2018); 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 escapement of Chum Salmon (69) was particularly low[3]; it was the 7th lowest count recorded in the 59-year dataset, while the count in 2017 (50) was the 5th lowest count. Pink Salmon escapement in the Quinsam River in 2018 (95,836) was lower than the mean value (129,713) but substantially exceeded the median (50th percentile) value (31,390) for the period of record (1953–2018). The estimated escapement of Sockeye Salmon in 2018 (17) was lower than the average and median values for the dataset.

During the five years of the JHTMON-8 study period, the most notable result was the occurrence of a record high Pink Salmon escapement (1.42 million) in Year 1 (2014). Chinook Salmon escapement

[3] Note that the end of the Chum Salmon sampling period (December 1; Table 4) was ~2 weeks prior to the end of the defined migration period (Table) and therefore this value may 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; e.g., during JHTMON-8, Chum Salmon sampling has ended on November 21 (Year 1 and 2) or December 15 (Year 3 and 4). Thus, it is appropriate to conclude that Chum Salmon returns were low in 2018, relative to returns in other years.

in the Quinsam River increased steadily over the first four years from 2,366 fish to 9,131 fish, and decreased in 2018 to 6,774 fish. By contrast, Coho Salmon escapement decreased steadily over the first four years from 14,883 fish to 5,865 fish, and increased in 2018 to 10,025 fish.

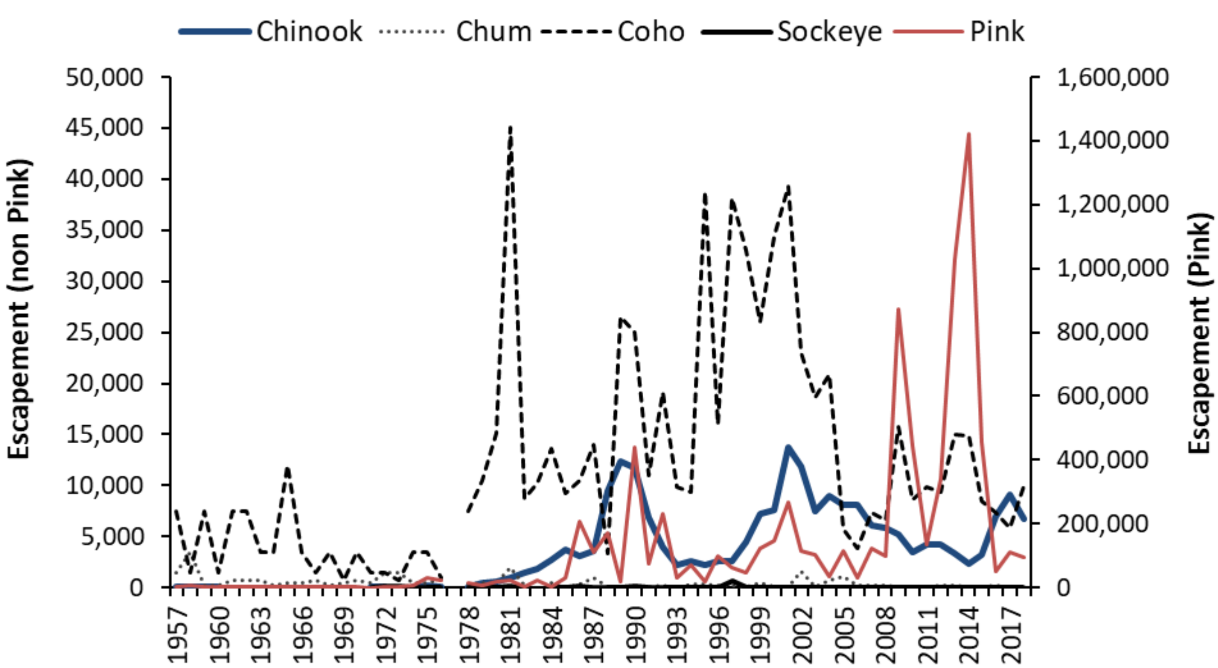
Table 15. 2018 salmon escapement data for the Quinsam river (DFO 2020).

Statistic	Salmon species				
	Chinook[1]	Chum	Coho[1]	Pink	Sockeye
2018 count	6,774	69	10,025	95,836	17
Mean (1953-2018)	4,270	480	12,165	129,713	53
Median(1953-2018)	3,394	261	9,263	31,390	25
10th percentile (1953-2018)	30	68	1,500	1,500	7
90th percentile (1953-2018)	9,428	1,500	31,730	418,404	130
Percent of years sampled (1953-2018)[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 5. Salmon escapement for the Quinsam River (1957–2018; DFO 2020).



3.1.2. Quinsam River Hatchery Salmon Counting Fence Operations

3.1.2.1. Juvenile Outmigration Monitoring in Year 6

Data collected at the salmon counting fence are summarized in Table 16. Following installation on March 11, the traps were monitored daily from March 12 to June 14.

The monitoring period provided good coverage of the Pink Salmon fry migration period in 2019, although low numbers of fry were captured on the first day of sampling, suggesting that the migration period started slightly prior to March 11. The migration was largely complete by April 30 (only 6 Pink Salmon fry were captured after this date). Total estimated migration of Pink Salmon fry has been highly variable in the six years of the monitoring program and was 6.2 million in 2019 (Year 6) (Table 16). Estimates varied by an order of magnitude among years since 2014; 22 million in 2014, 2.7 million in 2015, 9.2 million in 2016, 1.5 million in 2017, 10.7 million in 2018, and 6.2 million in 2019.

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 6. To provide broader context, outmigration estimates of priority species are presented in Figure 7 for the full period of record (since the mid 1970s), based on a data compilation exercise undertaken in Year 5 (Abell *et al.* 2019).

In Year 6 (2019), total estimated outmigration of colonized Coho Salmon (38,017) was the highest recorded during JHTMON-8, slightly higher than during Year 1 (36,339). Total estimated outmigration of wild Coho Salmon (32,968) was the second highest of the six years, approximately 30% lower than in Year 5 (46,679). The total estimated outmigration of steelhead smolts (11,079; 663 fish captured) was the highest recorded during JHTMON-8, although it should be recognized that 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 colonized Chinook Salmon (166,633) and wild Chinook Salmon (109,256) were the highest and second highest, respectively, during the six years of JHTMON-8, during which estimated Chinook Salmon outmigration has been highly variable. Wild Chinook Salmon outmigration was substantially higher in Years 4–6 (three-year average = 102,658 fish) than in Years 1–3 (three-year average = 6,978 fish). Chinook Salmon were noted to still be outmigrating on June 14 when the traps were removed; however, only 17 fish were captured on the final day of sampling (compared to >1000 during the peak of the period), indicating that outmigration was nearly complete. Estimated outmigration of all priority species during JHTMON-8 has been within the range of historical estimates (Figure 7).

The survival of out-planted juvenile salmon was estimated as the percentage of outmigrating juvenile colonized salmon that comprised the total number of fish out-planted (Figure 8). 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 (Years 2–6 of JHTMON-8). Estimated survival of colonized juvenile Chinook Salmon during JHTMON-8 was highest in 2019 and has varied between 65% and 80% during four of the five years, with a lower value (28%) estimated in 2016. Colonized juvenile Coho Salmon survival estimates are available for all six years of monitoring, ranging between 13% and 24%, with survival lower than Chinook Salmon, at least partly reflecting that this Coho Salmon spend longer in freshwater. As for Chinook Salmon, the survival estimate for Coho Salmon in 2019 was the highest during the six years of JHTMON-8. Note that the estimates for Coho Salmon assume that fish outmigrate at age 1+, although a small number of 2+ smolts were recorded at the fence[4].

Table 16. Summary of downstream migration data and total migration estimates from sampling at the Quinsam River Hatchery salmon counting fence, March 11 to June 14, 2019.

Species	Life Stage	Total Counts	Total Estimated Migration[1]	Peak Migration	Migration Period	Comments
Colonized Coho	Smolt	2,276	38,017	10-May-19	1 May- End	
Wild Coho	Smolt	1,971	32,968	13-May-19	6 Apr - 18 Jun	
2 Year old Coho	Smolt	1	17	n/a	n/a	
Coho	Fry	2,651	56,573	7-Apr-19	Start - End	
Steelhead	Smolt	663	11,079	11-May-19	15 Apr - 9 Jun	
Steelhead	Fingerling	147	2,477	12-May-19	24 Mar - End	
Steelhead	Kelts	0	0	n/a	n/a	
Cutthroat	Fingerling	38	625	18-May-19	13 Apr - End	
Cutthroat	Smolt	16	275	7-May-19	24 Apr - 25 May	
Cutthroat	Kelts	5	124	22-Mar-19	22 Mar - 12 Apr	
Trout Fry	Fry	9	159	26-May-19	26 May - 1 Jun	
Chinook	Fry	6,347	109,256	24-May-19	8 Apr - End	Still migrating on June 14
Colonized Chinook	Fry	9,978	166,633	22-May-19	10 May - End	Still migrating on June 14
Chum	Fry	855	21,204	30-Mar-19	25 Apr - End	
Sockeye	Fry	72	1,743	2-Apr-19	20 Mar - 28 Apr	
Pink	Fry	255,218	6,244,204	4-Apr-19	Start - 6 May	
Dolly Varden	Smolt	0	0	n/a	n/a	
Lamprey (2 species)	all	93	1,598	10-May-19	24 Mar - End	
Sculpin	all	124	2,173	11-Apr-18	9 Apr - 17 Jun	

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

[4] Estimated outmigration of 2+ Coho Salmon was 17 fish. Burt (2003) suggests that 2+ smolts represent fish that were trapped in off-channel habitats, preventing them from out-migrating the previous year.

Figure 6. Total estimated outmigration of priority species on the Quinsam River during Years 1-6 (2014-2019). Coho Salmon and steelhead were captured at the smolt stage and Chinook Salmon at the fry stage.

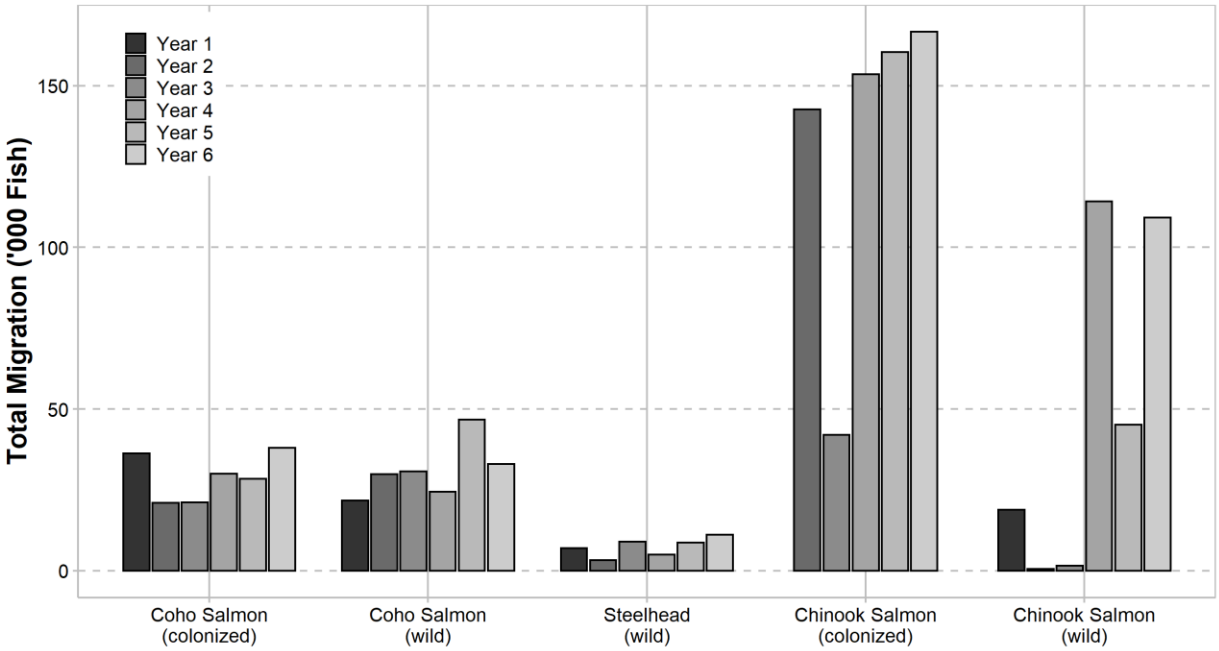


Figure 7. Estimated outmigration of priority species on the Quinsam River during 1979-2019, discriminated between colonized and wild fish. Coho Salmon and steelhead were captured at the smolt stage and Chinook Salmon at the fry stage.

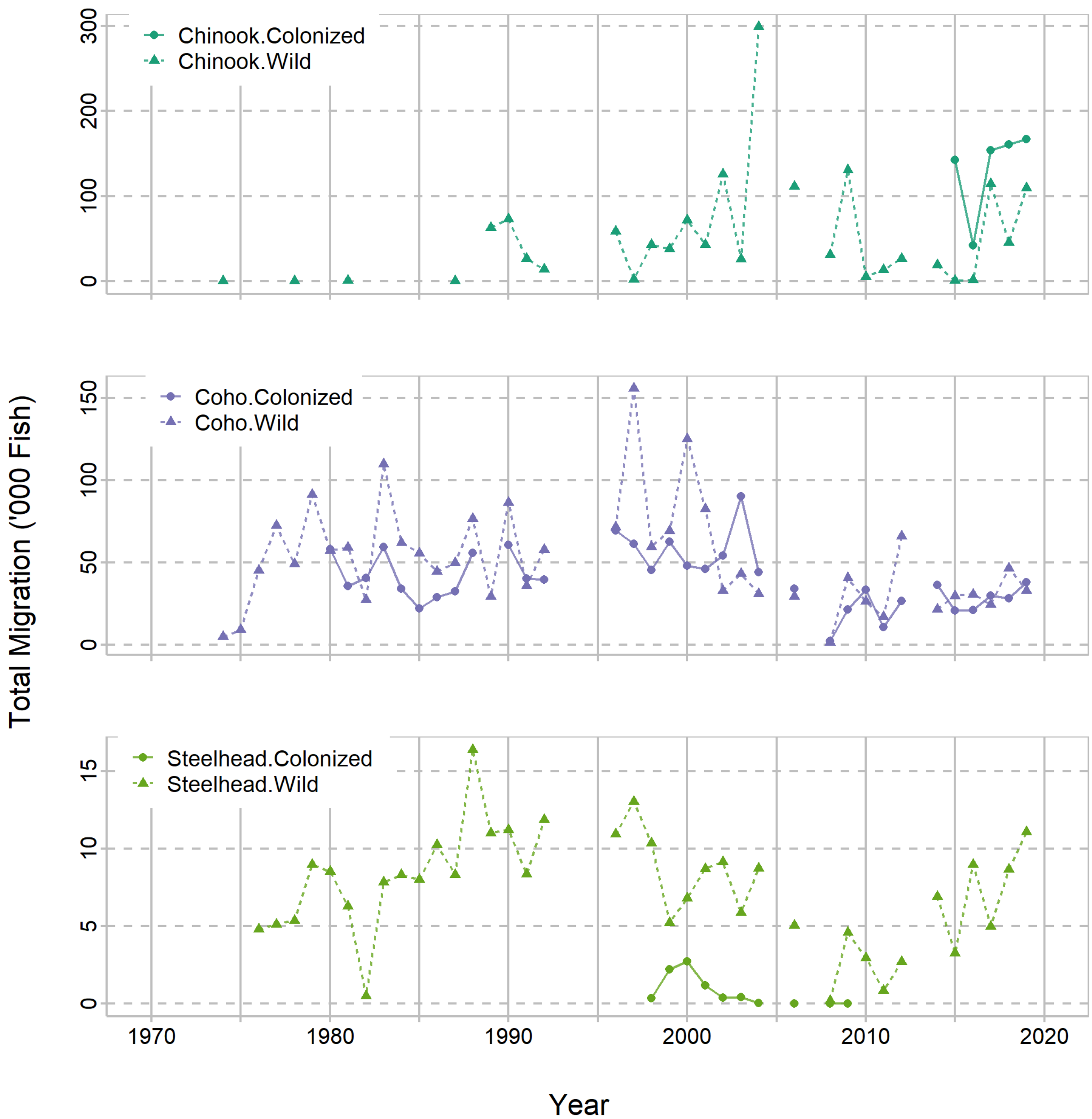
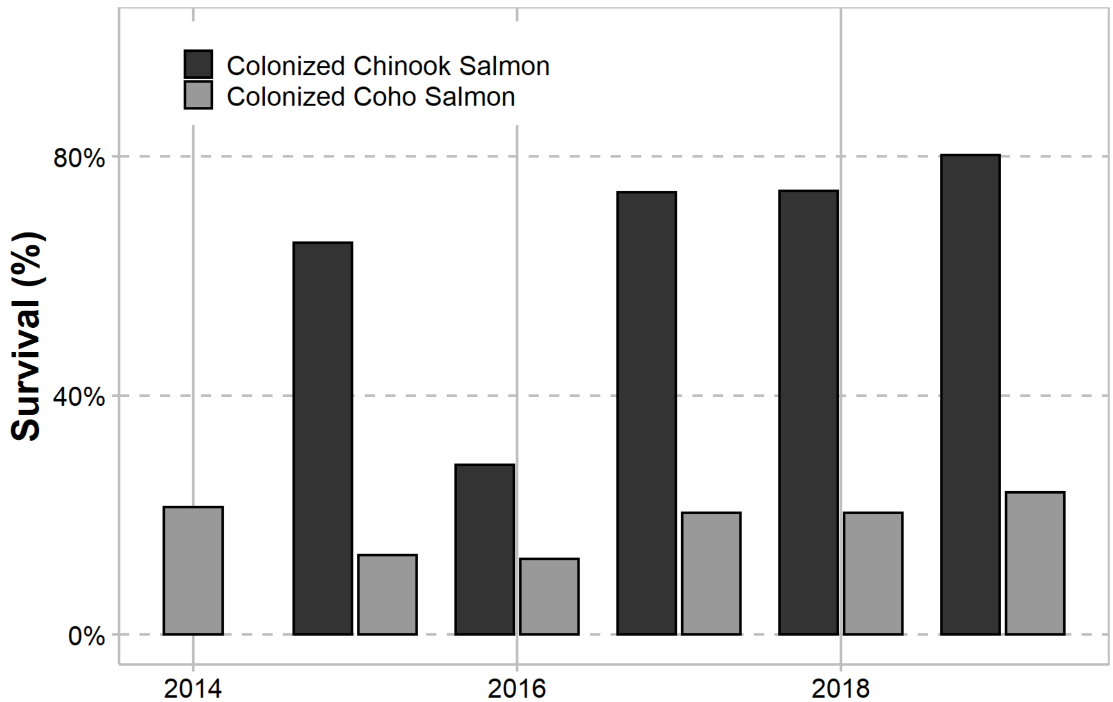


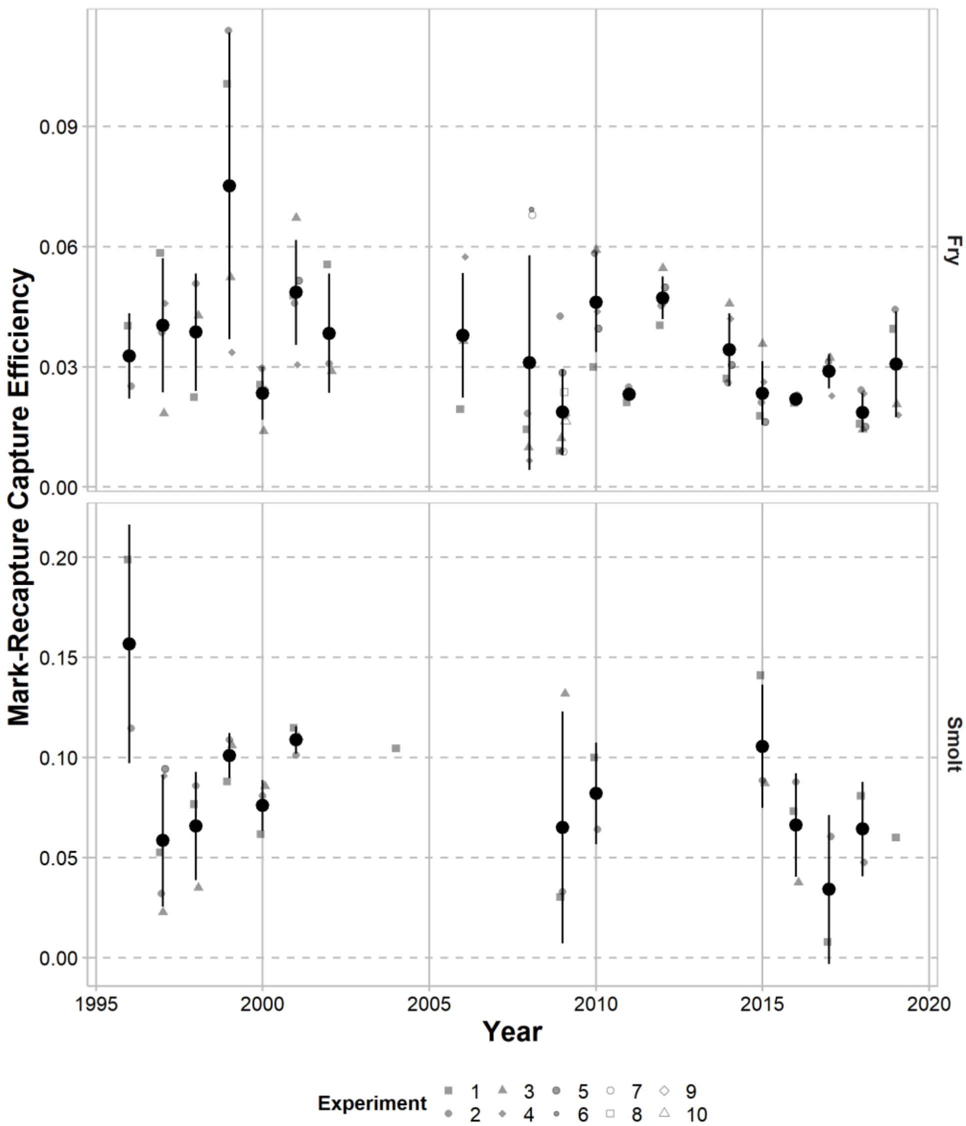
Figure 8. Estimated survival of out-planted salmon raised at the hatchery, based on the proportion of out-planted fish estimated to outmigrate at the salmon counting fence. Outmigrating Chinook Salmon were out-planted during spring (May) of the same year; outmigrating Coho Salmon were out-planted the previous year.



3.1.2.2. Review of Capture Efficiency Estimates

Based on the results of analysis to examine variability in capture efficiency estimates (Appendix A), we propose to apply life-stage-specific annual mean values of capture efficiency estimates (Figure 9) to estimate total juvenile outmigration for the JHTMON-8 study period (see Appendix A for further details). We welcome feedback from DFO or BC Hydro regarding this proposed change.

Figure 9. Life stage-specific capture efficiency estimates of the Quinsam Hatchery salmon counting fence, years 1996-2019. Symbols represent experiments. Solid circles represent annual means, and vertical lines represent 1 SD.



3.1.3. Water Quality

3.1.3.1. QA/QC

All laboratory analyses in Year 6 were conducted within the recommended hold times (see Appendix B), with the exception of all pH values. All pH samples from QUN-WQ 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 2019, considering only parameters with concentrations five times higher than the MDL, no duplicate samples collected at QUN-WQ had values with > 20% relative standard error. One field and one trip blank were collected in 2019. Values for all parameters were below the respective MDLs for both blanks. Values of pH were slightly higher in the field blank (5.41) than the travel blank (5.38).

3.1.3.2. Field Measurements

The Year 6 *in situ* and laboratory water chemistry results for the Quinsam River at QUN-WQ are summarized in Table 17 (general variables measured *in situ*), Table 18 (DO measured *in situ*), Table 19 (general variables measured at ALS laboratories), and Table 20 (low level nutrients measured at ALS laboratories). Combined results from Years 1 to 6 (2014 to 2019) of water quality monitoring are tabulated in Appendix B.

Alkalinity

Alkalinity (as $CaCO_3$) measured at ALS laboratories ranged from 31.1 mg/L (September) to 41.5 mg/L (June; Table 19) in 2019, 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 6 ranged from 7.56 to 7.88, while *in situ* pH ranged from 6.50 to 7.60 (Table 19 and Table 17, respectively). 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 1997b). The pH measured *in situ* are expected to be more accurate than the laboratory pH, given that the pH measured in the laboratory samples exceeded the recommended hold time.

Specific Conductivity and Total Dissolved Solids

In situ specific conductivity (conductivity normalized to 25°C) measured in Year 6 ranged from 78.2 µS/cm (September) to 146.6 µS/cm (June; Table 17). Similarly, laboratory values for conductivity in Year 6 ranged from 77.9 µS/cm (September) to 142.0 µS/cm (June; Table 19). Values were similar to previous years. 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 56 mg/L (August) to 97 mg/L (June; Table 19) in Year 6.

Turbidity and Total Suspended Solids (TSS)

Turbidity in the Quinsam River at QUN-WQ was low in all six monitoring years, indicating high water clarity (values in Year 6 ranged from 0.29 NTU to 0.51 NTU; Table 19). Similarly, TSS concentrations in Year 6 were low and consistent with previous years, with values generally ranging from below the MDL of 1.0 mg/L to slightly above this MDL (1.5 mg/L). An exception was analysis of the September 12 and August 12 samples, for which the laboratory was only able to provide low precision analysis (MDL = 3 mg/L), with the measurement below this MDL (Table 19). Turbidity measured on these dates was low (average values of 0.43 NTU and 0.41 NTU, respectively).

Dissolved Oxygen

Concentrations and saturation (%) of DO in the Quinsam River were highest in October 2019 (when flows were elevated). During June 12 to September 12, 2019, DO measurements were lower 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 18; BC MOE 1997). The measurement in June (average of 8.54 mg/L on June 12, 2019; Table 18) indicates 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 25 for periodicity information). The September measurement (average of 8.63 mg/L on September 12, 2019; Table 18) indicates that the 9 mg/L WQG-AL may not have been achieved during the early stages of the Pink Salmon incubation period, which is reported to start four days after the sample was collected on September 16 (Table 25). DO concentrations below the most conservative provincial WQG-AL have routinely been measured in previous years (see Appendix B).

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). For context, studies described in BC MOE (1997) indicate that dissolved oxygen concentrations are generally 2–6 mg/L lower in salmonid redds than in the overlying water. Separate guidelines exist for interstitial waters, which are not considered here as only the water column was sampled.

Total Gas Pressure

Monitoring TGP was discontinued in Year 2 following evaluation of results in Year 1, and the limited potential of the Quinsam River Diversion facility to cause elevated TGP. Results from TGP monitoring in Year 1 are presented in Appendix B.

Nitrogen

Total ammonia concentrations in the Quinsam River at QUN-WQ were less than the detection limit of 5.0 µg N/L during three of the six sampling events in Year 6 (Table 20). During the June and

August sampling events, one of the duplicate samples had detectable values of 5.2 μg N/L and 18.7 μg N/L, respectively. During the October sampling event, total ammonia concentrations were detectable in both duplicates (average of 5.7 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 1986).

Nitrite concentrations were below the detection limit of 1.0 μg N/L during five of the six sampling events in Year 6 (Table 20). During the October sampling event, nitrate concentrations were detectable (average of 1.5 μg N/L), but well below the WQG-AL. 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 10.2 μg N/L (May) to 28.1 μg N/L (October) during Year 6, similar to previous years (Table 20). In oligotrophic lakes and streams, nitrate concentrations are usually lower than 100 μg N/L (Nordin and Pommen 1986).

Phosphorus

Orthophosphate concentrations were below the detection limit of 1.0 μg P/L during sampling in Year 6, similar to previous years (Table 20). 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 6 sampling period were low, similar to previous years, ranging from below MDL (<2.0 μg/L) to 4.9 μg/L (Table 20).

Table 17. Quinsam River (QUN-WQ) general water quality variables measured *in situ* during Year 6 (2019).

Year	Date	Air Temperature °C				Water Temperature °C				Conductivity µS/cm				Specific Conductivity µS/cm				pH pH units				Salinity ppt			
		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
2019	13-May	8.0	8.0	8.0	0.0	11.0	11.0	11.0	0.0	84.4	84.4	84.4	0.0	115.3	115.2	115.3	0.1	6.50	6.50	6.50	0.00	0.10	0.10	0.10	0.00
	12-Jun	18.5	18.5	18.5	0.0	18.4	18.4	18.4	0.0	128.0	128.0	128.0	0.0	146.6	146.5	146.6	0.1	7.60	7.59	7.60	0.01	0.07	0.07	0.07	0.00
	11-Jul	15.0	15.0	15.0	0.0	18.4	18.4	18.4	0.0	98.8	98.8	98.9	0.1	113.8	113.8	113.8	0.0	7.15	7.13	7.17	0.02	0.05	0.05	0.05	0.00
	12-Aug	14.0	14.0	14.0	0.0	18.8	18.8	18.8	0.0	82.8	82.8	82.8	0.0	94.6	94.6	94.6	0.0	7.42	7.41	7.42	0.01	0.04	0.04	0.04	0.00
	12-Sep	13.0	13.0	13.0	0.0	17.0	17.0	17.0	0.0	66.3	66.3	66.3	0.0	78.2	78.2	78.2	0.0	7.56	7.55	7.56	0.01	-	-	-	-
	09-Oct	5.0	5.0	5.0	0.0	8.1	8.1	8.1	0.0	91.8	91.7	91.8	0.1	135.7	135.7	135.7	0.0	7.33	7.33	7.33	0.00	0.06	0.06	0.06	0.00

[1] Average of three replicates (n=3) on each date unless otherwise indicated. Dashes (-) indicate no data were collected.

Table 18. Quinsam River (QUN-WQ) dissolved gases measured *in situ* during Year 6 (2019).

Year	Date	Oxygen Dissolved %				Oxygen Dissolved mg/L			
		Avg[1]	Min	Max	SD	Avg[1]	Min	Max	SD
2019	13-May	90.4	90.4	90.4	0.0	9.95	9.95	9.96	0.01
	12-Jun	90.9	90.9	91.0	0.1	8.54	8.53	8.55	0.01
	11-Jul	89.8	89.4	90.0	0.3	8.43	8.40	8.45	0.03
	12-Aug	91.9	91.8	92.0	0.1	8.58	8.57	8.59	0.01
	12-Sep	89.4	89.1	89.7	0.3	8.63	8.62	8.65	0.02
	09-Oct	98.4	98.3	98.5	0.1	11.64	11.64	11.65	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 19. Quinsam River (QUN-WQ) general water quality variables measured at ALS laboratories during Year 6 (2019).

Year	Date	Alkalinity, Total (as CaCO3) mg/L				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
2019	13-May	35.8	35.6	36.0	0.3	119.5	118.0	121.0	2.1	71	66	76	7	<1.2	<1.0	1.3	0.2	0.35	0.33	0.36	0.02	7.65	7.65	7.65	0.00
	12-Jun	41.5	41.5	41.5	0.0	142.0	142.0	142.0	0.0	96	94	97	2	<1.1	<1.0	1.2	0.1	0.36	0.34	0.38	0.03	7.88	7.87	7.88	0.01
	11-Jul	35.0	35.0	35.0	0.0	103.0	103.0	103.0	0.0	75	74	76	1	1.4	1.2	1.5	0.2	0.48	0.45	0.51	0.04	7.75	7.73	7.76	0.02
	12-Aug	33.7	33.4	34.0	0.4	83.6	83.4	83.8	0.3	56	56	56	0	<3.0	<3.0	<3.0	0.0	0.43	0.41	0.45	0.03	7.64	7.62	7.66	0.03
	12-Sep	31.2	31.1	31.2	0.1	78.2	77.9	78.5	0.4	62	61	62	1	<3.0	<3.0	<3.0	0.0	0.41	0.33	0.49	0.11	7.57	7.56	7.58	0.01
	09-Oct	39.2	39.1	39.3	0.1	132.0	132.0	132.0	0.0	79	77	81	3	<1.0	<1.0	<1.0	0.0	0.32	0.29	0.34	0.04	7.73	7.73	7.73	0.00

[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 20. Quinsam River (QUN-WQ) nutrient concentrations measured at ALS laboratories during Year 6 (2019).

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
2019	13-May	<5	<5	<5	0	<1	<1	<1	0	10.2	10.2	10.2	0.0	<1	<1	<1	0	<2.3	<2	2.5	0.4
	12-Jun	<5.1	<5	5.2	0.1	<1	<1	<1	0	21.3	20.8	21.8	0.7	<1	<1	<1	0	3.3	3.0	3.6	0.4
	11-Jul	<5	<5	<5	0	<1	<1	<1	0	17.9	17.3	18.5	0.8	<1	<1	<1	0	4.8	4.7	4.9	0.1
	12-Aug	<11.9	<5.0	18.7	9.7	<1	<1	<1	0	19.1	18.8	19.4	0.4	<1	<1	<1	0	<2.1	<2	2.1	0.1
	12-Sep	<5	<5	<5	0	<1	<1	<1	0	14.2	14.1	14.3	0.1	<1	<1	<1	0	3.5	3.3	3.6	0.2
	09-Oct	5.7	5.5	5.8	0.2	<1	<1	<1	0	27.1	26.1	28.1	1.4	1.5	1.5	1.5	0.0	4.4	4.3	4.4	0.1

[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.

3.1.4. 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 2019. In 2019 (January to September), monthly average water temperatures ranged between 2.0°C (February) and 18.7°C (August; Table 21).

The water temperature records for the Quinsam River show occurrences of warm water temperatures from a fisheries biology perspective, although maximum summer water temperatures were lower in Year 6 than in the previous five years. In 2019, there were 75 days (27% of record) with daily mean temperatures above 18°C, but no days with daily mean temperature above 20°C. Over the period of record between 2014 and 2018, there were 52 to 77 days per year (14% to 21%) with daily mean temperatures above 18°C, and 14 to 30 days per year (4% to 8%) with daily mean temperatures above 20°C (Table 22). There were no days in Year 6 (2019) with mean water temperature <1°C and this only occurred in 2017 (7 days).

Figure 10. Daily mean water temperatures in the Quinsam River (QUN-WQ) between May 2014 and October 2019.

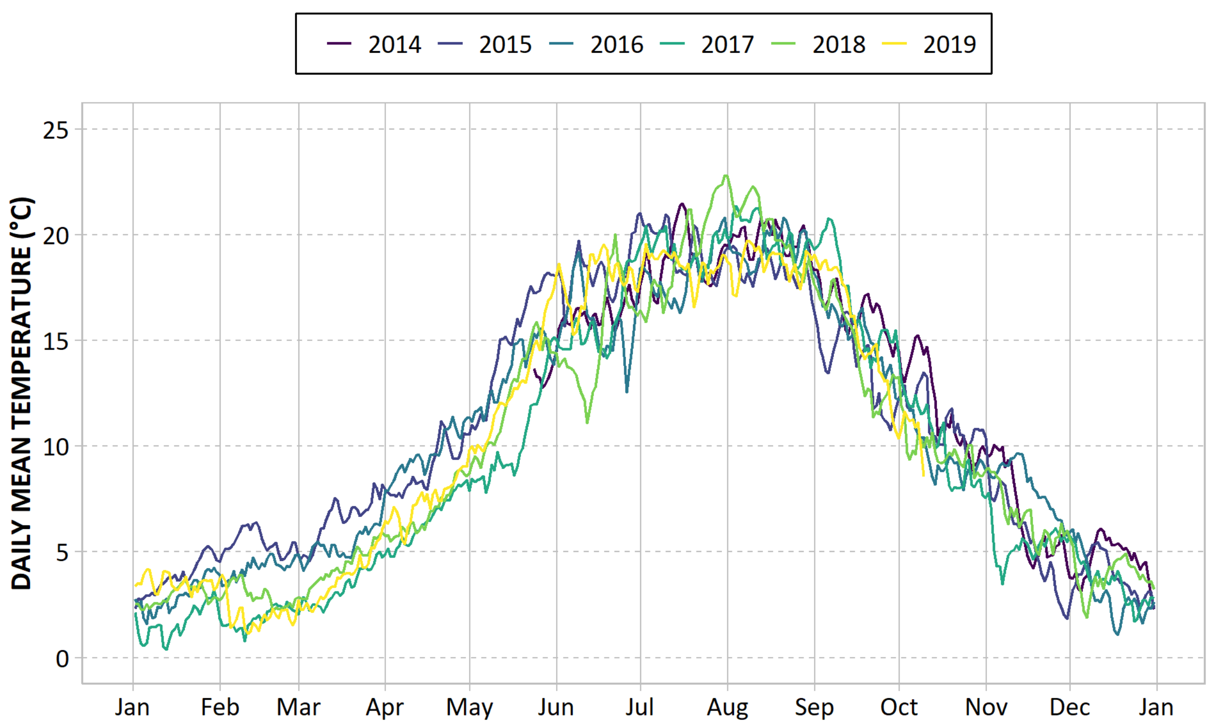


Table 21. Monthly water temperature in the Quinsam River (QUN-WQ) from 2014 to 2019. Statistics were not calculated for months with fewer than 3 weeks of observations.

Month	2014[1, 2, 3]				2015[1, 3]				2016[1, 3]				2017[1, 3]				2018[1, 3]				2019[1, 3, 4]			
	Avg	Min	Max	SD	Avg	Min	Max	SD	Avg	Min	Max	SD	Avg	Min	Max	SD	Avg	Min	Max	SD	Avg	Min	Max	SD
Jan	n/a	n/a	n/a	n/a	3.8	2.0	5.6	0.8	3.0	1.3	4.7	0.8	1.7	0.2	3.4	0.8	2.9	2.0	3.8	0.4	3.5	2.6	4.3	0.4
Feb	n/a	n/a	n/a	n/a	5.5	4.1	6.5	0.6	4.3	3.1	5.3	0.5	1.9	0.5	2.9	0.5	2.9	1.9	4.1	0.5	2.0	0.7	4.1	0.7
Mar	n/a	n/a	n/a	n/a	6.6	4.0	8.9	1.1	5.5	3.3	9.2	1.0	3.4	1.9	5.4	0.9	4.4	2.5	6.4	0.9	3.8	1.5	7.5	1.3
Apr	n/a	n/a	n/a	n/a	9.0	6.6	12.7	1.3	9.8	6.8	12.4	1.2	6.6	4.1	9.3	1.2	7.0	5.0	9.9	1.2	7.5	4.7	11.3	1.2
May	n/a	n/a	n/a	n/a	15.1	9.6	18.5	2.5	13.7	10.1	16.2	1.5	10.5	7.1	16.5	2.4	12.6	8.3	16.9	2.4	13.0	8.3	18.7	2.5
Jun	16.3	14.4	18.9	0.8	18.3	15.0	22.9	1.4	16.1	11.9	19.8	1.7	16.0	13.6	20.2	1.8	15.3	10.1	20.6	2.5	17.8	14.5	20.0	1.2
Jul	18.9	16.5	22.7	1.4	19.3	15.9	23.0	1.6	18.2	15.5	21.3	1.3	19.3	17.6	20.9	0.8	19.4	14.9	23.6	2.2	18.6	15.9	20.2	0.7
Aug	19.8	17.5	22.2	1.0	18.3	15.9	21.2	1.1	19.3	17.7	21.3	0.9	19.9	18.0	21.8	0.9	20.1	17.3	23.1	1.5	18.7	16.3	20.3	0.8
Sep	16.3	13.9	18.6	1.1	13.8	10.2	17.1	1.8	15.1	11.8	18.1	1.4	16.8	13.3	21.1	2.3	14.6	10.8	18.6	2.1	15.5	9.8	19.2	2.6
Oct	11.8	8.3	15.5	2.1	11.3	9.3	13.7	1.1	9.6	7.4	13.1	1.2	10.0	7.1	13.9	1.8	9.7	8.2	12.8	0.8	-	-	-	-
Nov	6.6	3.6	10.3	2.2	5.4	1.7	10.1	2.1	8.0	5.6	9.8	1.2	5.4	3.1	8.1	0.8	6.6	4.5	9.1	1.2	-	-	-	-
Dec	4.5	2.1	6.2	1.0	3.8	2.0	5.6	1.0	2.9	0.9	6.1	1.2	3.4	1.5	5.7	0.9	3.8	1.7	5.7	0.8	-	-	-	-

[1] "Avg", "Min", "Max" and "SD" denote the monthly average, minimum, maximum and standard deviation of water temperatures.

[2] "n/a" indicates that TidbiTs were not installed.

[3] Blue and orange shadings highlight minimum and maximum temperatures respectively.

[4] "-" indicates that TidbiT data has not yet been collected.

Table 22. Summary of the frequency of exceedances of mean daily water temperature extremes (T_{water}>18°C, T_{water}>20°C, and T_{water}<1°C) in the Quinsam River at QUN-WQ from 2014 to 2019.

Year	Record Length (days)	Days T_{water} < 1°C	Days T_{water} > 18°C	Days T_{water} > 20°C
2014	222	0	54	20
2015	365	0	69	16
2016	366	0	52	14
2017	365	7	77	25
2018	365	0	55	30
2019	281	0	75	0

Rates of Change

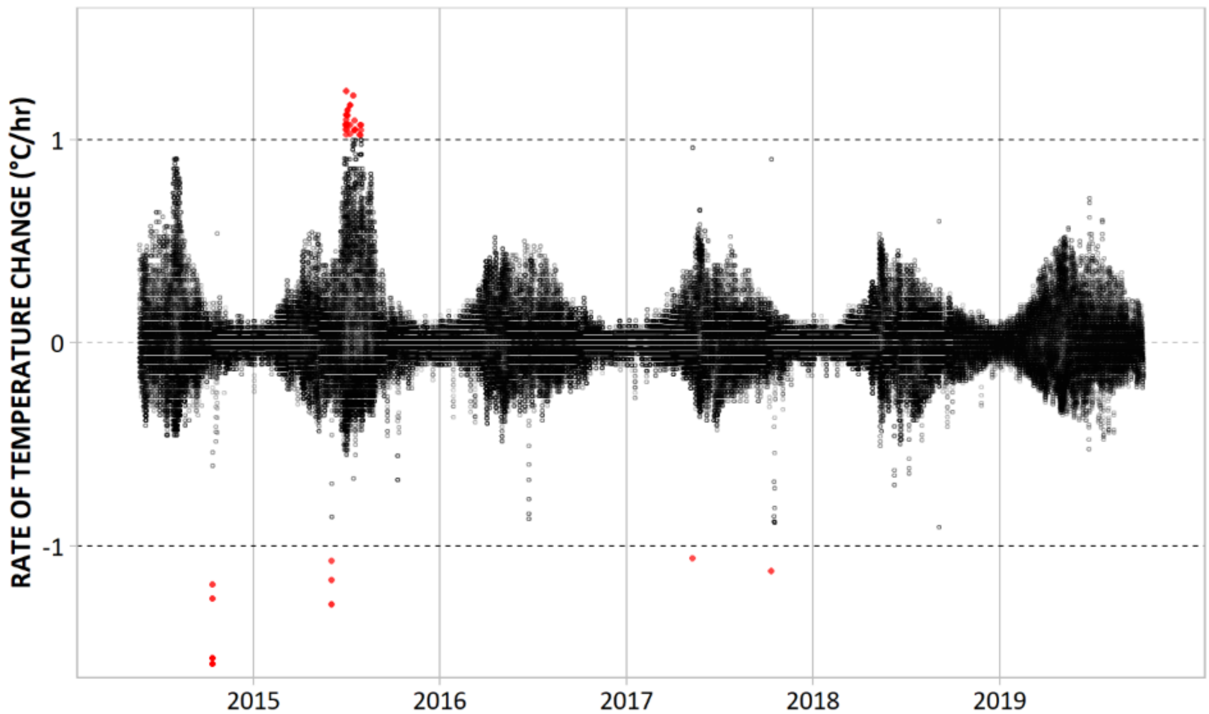
Statistics relating to rates of change of water temperature at QUN-WQ are summarized in Table 23 and presented in Figure 11. For the period of record, the hourly rates of temperature change at QUN-WQ were between -0.2°C/hr and +0.2°C/hr for at least 90% of the time (based on the 5th and 95th percentiles) 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 -1.6°C/hr (Table 23). Both these maximum values occurred prior to Year 6 (Figure 11). Rates of temperature change with magnitudes >1°C/hr occurred for 0.02% of the records. Based on our experience on other streams in BC, it is normal for hourly rates of water temperature change to exceed ±1°C for a small percentage of data points.

Table 23. Statistics for the hourly rates of change in water temperature at QUN-WQ in the Quinsam River, 2014 to 2019. The frequency of rates of change exceeding a magnitude of 1°C/hr is also shown.

Station	Start of Record	End of Record	Number of Datapoints	Occurrence of rates > 1°C/hr		Max -ve	Percentile				Max +ve
				Number	% of record		1st	5th	95th	99th	
QUN-WQ	23-May-14	9-Oct-19	188,645	37	0.020	-1.6	-0.3	-0.2	0.2	0.4	1.2

Figure 11. Hourly rate of change in 15-minute water temperature in the Quinsam River (QUN-WQ) from 2014 to 2019. Large red dots indicate rates with magnitudes exceeding ±1°C/hr.



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 10, the growing season was assumed to begin when the weekly average water temperature exceeded and remained above 5°C, and to end when the weekly average temperature dropped below 4°C (as per Coleman and Fausch 2007).

The growing season at QUN-WQ was determined for 2015 to 2018 (Years 2 to 5), which are the study years for which complete annual records exist (Table 24). The most recent growing season for which data are available was 2018 (Year 5) for which the growing season commenced on March 23rd, ended on December 2nd, covering a period of 255 days, and accumulating 3,271 degree days. This was shorter than the growing season length calculated for Year 2 (263 days) and Year 3 (265 days), but longer than for Year 4 (250 days). Growing season statistics for the 2019 growing season will be presented in the Year 7 Annual Report when all 2019 data are available.

Table 24. Growing season timing and growing degree days at QUN-WQ in the Quinsam River (2014 to 2019).

Year	Number of days with valid data	Growing Season				
		Start Date	End Date	Length (days)	Gap (days)	Degree Days[1]
2014[1]	-	-	-	-	-	-
2015	365	04-Mar-15	22-Nov-15	263	0	3,539
2016	366	17-Mar-16	06-Dec-16	265	0	3,469
2017	365	02-Apr-17	07-Dec-17	250	0	3,214
2018	365	23-Mar-18	02-Dec-18	255	0	3,271
2019[2]	-	-	-	-	-	-

[1] Growing season could not be estimated because a complete dataset over the course of the growing season is not available.

[2] Growing season will be reported once the dataset covers a complete growing season.

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 2019 are compared to optimum temperature ranges for fish species in Table 25. For each life stage, Table 25 also 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 25). 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.

Considering all years and all species/life stages, MWMxT in the Quinsam River exceeded optimum ranges by more than 1°C for an average 17.8% of the time, and were below optimum ranges by more than 1ºC for an average of 26.5% of the time (Table 25).

For Chinook Salmon, temperatures were within optimum ranges during the migration stage for all years (2014 to 2019). Temperatures for spawning were mostly within the optimum range (57.4% to 100% of the time) with instances where ranges were exceeded by more than 1ºC only occurring in 2014 and 2015. Temperatures during incubation were cooler than the optimum range at times during all years, particularly in 2016, when 52.3% of values exceeded the lower bound by more than 1°C. Water temperatures were outside the optimum range during most of the Chinook Salmon rearing period (temperatures were within the optimum range for 8.6% to 36.5% of the time). In Year 6 (2019), 36.5% of values were below the optimum rearing range and 43.8% of values above the optimum rearing range.

For Coho Salmon, temperatures were typically below the upper bound of the optimum ranges for migration, spawning, and incubation stages (except migration in 2014, where 6.5% of the temperatures 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 6 (2019), water temperatures during the Coho Salmon rearing period were above the upper bound (40.6%) more often than below the lower bound (40.3%) of the optimum temperature range.

For Pink Salmon, 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 1°C for the majority of the time (up to 83% of the spawning period in 2014). Conditions in 2019 were within the ranges observed in 2014 to 2018. During the Pink Salmon incubation period, water temperatures were within optimum ranges for the majority of time, except 2016 and 2019 when 42.6% and 45.0% of values, respectively, were within the optimum range.

For steelhead, 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 2019 were below the optimum range by more than 1°C for 75.0% to 100% of the time. In 2019, water temperatures were within the optimum bounds for 0% of the spawning stage, 9.2% of the incubation stage, and 6.5% of the rearing stage.

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 Table 25 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 25 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.

Table 25. Mean weekly maximum temperatures (MWMxT) in the Quinsam River from 2014 to 2019 compared to optimum temperature ranges for fish species present. Periodicity information is from Burt (2003).

Species	Life Stage			Year	Percent Complete (%)	MWMT (°C)		% of MWMT				
	Periodicity	Optimum Temperature Range (°C)	Duration (days)			Min	Max	Below Lower Bound by >1°C	Below Lower Bound	Between Bounds	Above Upper Bound	Above Upper Bound by >1°C
Chinook Salmon	Migration (Sep. 23 to Nov. 22)	3.3-19.0	31	2014	100%	5.2	16.5	0.0	0.0	100.0	0.0	0.0
				2015	100%	4.4	12.9	0.0	0.0	100.0	0.0	0.0
				2016	100%	7.3	14.4	0.0	0.0	100.0	0.0	0.0
				2017	100%	4.7	15.6	0.0	0.0	100.0	0.0	0.0
				2018	100%	5.8	13.5	0.0	0.0	100.0	0.0	0.0
				2019	21%	11.4	14.3	0.0	0.0	100.0	0.0	0.0
	Spawning (Oct. 01 to Nov. 30)	5.6-13.9	61	2014	100%	4.7	15.0	0.0	26.2	57.4	16.4	3.3
				2015	100%	2.8	12.9	16.4	23.0	77.0	0.0	0.0
				2016	100%	6.0	12.6	0.0	0.0	100.0	0.0	0.0
				2017	100%	4.7	14.0	0.0	26.2	72.1	1.6	0.0
				2018	100%	5.6	12.4	0.0	1.6	98.4	0.0	0.0
				2019	8%	11.4	11.5	0.0	0.0	100.0	0.0	0.0
	Incubation (Oct. 16 to Apr.	5.0-14.0	242	2014	100%	2.8	11.6	9.6	21.3	78.7	0.0	0.0
				2015	100%	2.4	12.6	25.8	49.0	51.0	0.0	0.0
				2016	100%	1.3	9.6	52.3	57.4	42.6	0.0	0.0
				2017	100%	2.6	10.2	41.6	54.3	45.7	0.0	0.0
				2018	100%	1.8	10.7	42.6	54.8	45.2	0.0	0.0
				2019	0%	-	-	-	-	-	-	-
	Rearing (Mar. 08 to Jul. 22)	10.0-15.5	365	2014	42%	13.9	21.9	0.0	0.0	8.6	91.4	86.2
				2015	100%	6.6	22.5	22.6	29.2	19.0	51.8	48.2
				2016	100%	5.4	19.3	17.5	21.9	36.5	41.6	26.3
				2017	100%	2.8	20.3	42.3	50.4	12.4	37.2	23.4
				2018	100%	4.0	21.1	34.3	40.9	27.0	32.1	26.3
				2019	100%	3.5	19.6	33.6	36.5	19.7	43.8	41.6
Coho Salmon	Migration (Sep. 16 to Dec. 31)	7.2-15.6	61	2014	100%	3.1	17.1	44.9	45.8	44.9	9.3	6.5
				2015	100%	2.8	14.9	43.9	48.6	51.4	0.0	0.0
				2016	100%	2.2	16.2	30.8	36.4	60.7	2.8	0.0
				2017	100%	2.6	16.0	55.1	56.1	41.1	2.8	0.0
				2018	100%	3.3	14.4	40.2	48.6	51.4	0.0	0.0
				2019	19%	11.4	15.4	0.0	0.0	100.0	0.0	0.0
	Spawning (Oct. 16 to Jan. 15)	4.4-12.8	39	2014	100%	2.8	11.6	10.9	28.3	71.7	0.0	0.0
				2015	100%	2.4	11.5	33.7	47.8	52.2	0.0	0.0
				2016	100%	1.3	9.6	41.3	44.6	55.4	0.0	0.0
				2017	100%	2.6	10.2	29.3	43.5	56.5	0.0	0.0
				2018	100%	3.3	10.0	3.3	38.0	62.0	0.0	0.0
				2019	0%	-	-	-	-	-	-	-
	Incubation (Oct. 16 to Dec.	4.0-13.0	197	2014	100%	3.1	11.6	0.0	6.5	93.5	0.0	0.0
				2015	100%	2.8	11.5	5.2	31.2	68.8	0.0	0.0
				2016	100%	2.2	9.6	27.3	32.5	67.5	0.0	0.0
				2017	100%	2.6	10.2	14.3	20.8	79.2	0.0	0.0
				2018	100%	3.3	10.0	0.0	16.9	83.1	0.0	0.0
				2019	0%	-	-	-	-	-	-	-
	Rearing (Jan. 01 to Dec. 31)	9.0-16.0	365	2014	60%	3.1	21.9	23.2	24.1	23.2	52.7	38.6
				2015	100%	2.8	22.5	38.4	42.7	26.3	31.0	28.5
				2016	100%	2.2	20.8	36.1	38.5	35.2	26.2	21.0
				2017	100%	1.3	21.3	47.1	53.7	19.7	26.6	23.0
				2018	100%	2.6	23.1	45.2	47.9	27.1	24.9	22.5
				2019	76%	1.8	19.8	36.3	40.3	19.1	40.6	38.1

Blue shading indicates provincial guideline exceedance of the lower bound of the optimum temperature range by more than 1°C (Oliver and Fidler 2001)
Orange shading indicates provincial guideline exceedance of the upper bound of the optimum temperature range by mor ethan 1°C (Oliver and Fidler 2001)

Table 25. Continued (2 of 2).

Species	Life Stage: Periodicity	Optimum Temperature Range (°C)	Duration (days)	Year	Percent Complete (%)	MWMT (°C) Min	MWMT (°C) Max	% of MWMT: Below Lower Bound by >1°C	Below Lower Bound	Between Bounds	Above Upper Bound	Above Upper Bound by >1°C
Pink Salmon	Migration (Aug. 01 to Oct. 15)	7.2-15.6	53	2014	100%	11.8	21.9	0.0	0.0	26.3	73.7	67.1
				2015	100%	11.0	20.9	0.0	0.0	50.0	50.0	40.8
				2016	100%	9.3	20.8	0.0	0.0	35.5	64.5	48.7
				2017	100%	10.5	21.3	0.0	0.0	35.5	64.5	59.2
				2018	100%	10.1	22.6	0.0	0.0	42.1	57.9	53.9
				2019	87%	11.4	19.8	0.0	0.0	30.3	69.7	66.7
	Spawning (Sep. 16 to Oct. 15)	7.2-12.8	61	2014	100%	11.8	17.1	0.0	0.0	10.0	90.0	83.3
				2015	100%	11.0	14.9	0.0	0.0	70.0	30.0	16.7
				2016	100%	9.3	16.2	0.0	0.0	50.0	50.0	33.3
				2017	100%	10.5	16.0	0.0	0.0	40.0	60.0	53.3
				2018	100%	10.1	14.4	0.0	0.0	63.3	36.7	6.7
				2019	67%	11.4	15.4	0.0	0.0	45.0	55.0	45.0
	Incubation (Sep. 16 to Apr. 07)	4.0-13.0	168	2014	100%	2.8	17.1	1.5	9.3	77.5	13.2	12.3
				2015	100%	2.4	14.9	9.8	24.9	72.2	2.9	2.0
				2016	100%	1.3	16.2	43.6	50.5	42.6	6.9	4.4
				2017	100%	2.6	16.0	16.7	40.2	51.0	8.8	7.4
				2018	100%	1.8	14.4	13.2	41.2	54.4	4.4	0.5
				2019	10%	11.4	15.4	0.0	0.0	45.0	55.0	45.0
Rainbow/ Steelhead Trout	Spawning (Feb. 16 to Apr. 15)	10.0-15.5	91	2014	0%	-	-	-	-	-	-	-
				2015	100%	5.3	9.4	86.4	100.0	0.0	0.0	0.0
				2016	100%	4.8	10.2	75.0	85.0	15.0	0.0	0.0
				2017	100%	2.4	6.9	100.0	100.0	0.0	0.0	0.0
				2018	100%	2.6	7.0	100.0	100.0	0.0	0.0	0.0
				2019	100%	2.1	8.3	100.0	100.0	0.0	0.0	0.0
	Incubation (Feb. 16 to Jun. 15)	10.0-12.0	137	2014	18%	13.9	16.8	0.0	0.0	0.0	100.0	100.0
				2015	100%	5.3	19.3	42.5	50.0	14.2	35.8	34.2
				2016	100%	4.8	18.6	37.2	42.1	17.4	40.5	33.9
				2017	100%	2.4	16.4	65.0	74.2	4.2	21.7	20.0
				2018	100%	2.6	16.1	55.8	63.3	6.7	30.0	26.7
				2019	100%	2.1	19.6	55.0	58.3	9.2	32.5	29.2
	Rearing (Jan. 01 to Dec. 31)	16.0-18.0	365	2014	60%	3.1	21.9	45.0	47.3	22.3	30.5	23.2
				2015	100%	2.8	22.5	65.8	69.0	4.4	26.6	18.4
				2016	100%	2.2	20.8	64.8	73.8	10.4	15.8	10.9
				2017	100%	1.3	21.3	66.3	73.4	4.4	22.2	20.3
				2018	100%	2.6	23.1	71.5	75.1	8.2	16.7	13.4
				2019	76%	1.8	19.8	57.6	59.4	6.5	34.2	16.9

Blue shading indicates provincial guideline exceedance of the lower bound of the optimum temperature range by more than 1°C (Oliver and Fidler 2001)

Orange shading indicates provincial guideline exceedance of the upper bound of the optimum temperature range by mor ethan 1°C (Oliver and Fidler 2001)

Air Temperature

Figure 12 shows the daily average air temperature for the period of record from May 2014 to October 2019. The monthly average, minimum, and maximum air temperatures are shown in Table 26. 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 33.3°C in July 2018. The maximum monthly mean air temperature (18.8°C) was in July 2015. Mean monthly air temperatures during summer 2019 were generally lower than previous years of JHTMON-8; e.g., the mean monthly air temperature during July 2019 (16.4ºC) was lower than all five previous years, while the mean monthly air temperature during August 2019 (17.0ºC) was lower than four of the previous years.

Air and water temperatures were highly correlated (Figure 13). Daily mean water temperatures typically exceeded daily mean air temperatures, which likely partly reflected the influence of warming in lakes upstream.

Figure 12. Air temperature at the Quinsam River (QUN-AT) between May 2014 and October 2019.

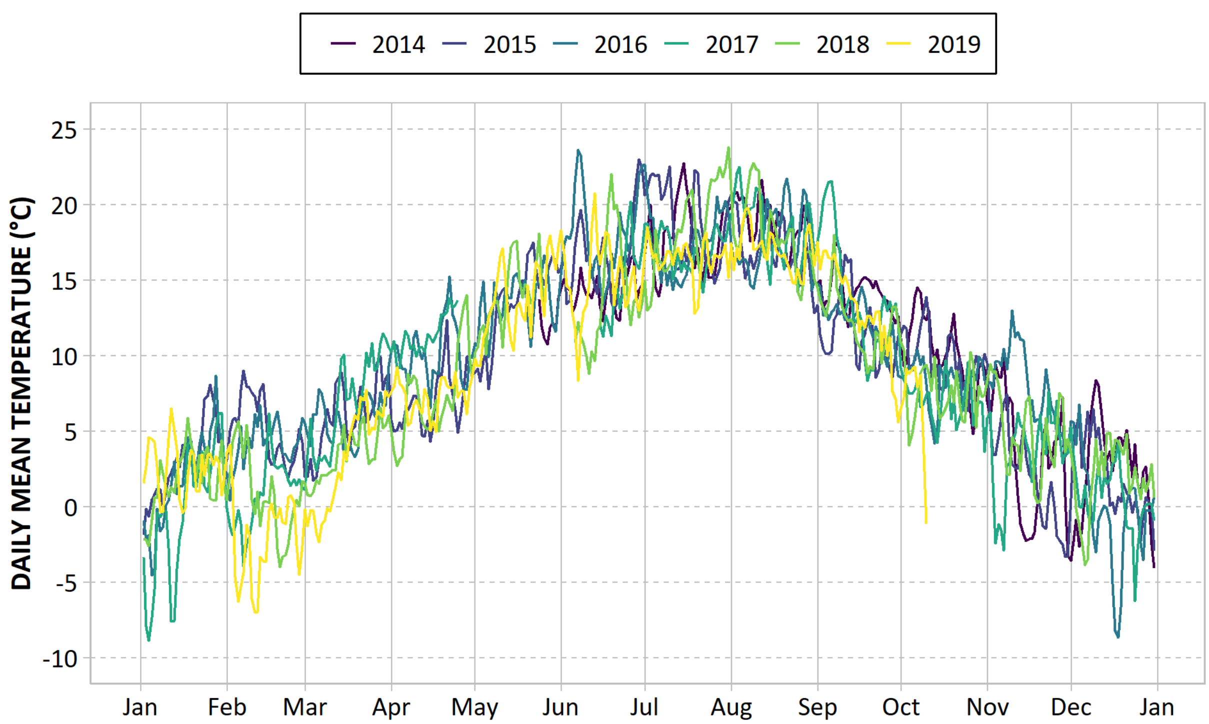


Table 26. Monthly air temperature statistics at the Quinsam River (QUN-AT) from 2014 to 2019. Statistics were not calculated for months with fewer than 3 weeks of observations.

Month	2014[1, 2, 3]				2015[1, 3]				2016[1, 3]				2017[1, 3]				2018[1, 3]				2019[1, 3, 4]			
	Avg	Min	Max	SD	Avg	Min	Max	SD	Avg	Min	Max	SD	Avg	Min	Max	SD	Avg	Min	Max	SD	Avg	Min	Max	SD
Jan	n/a	n/a	n/a	n/a	3.0	-4.0	9.1	2.6	1.8	-7.0	9.3	3.2	-0.4	-11.5	8.2	4.5	1.7	-5.4	7.3	2.3	2.6	-2.5	8.7	2.3
Feb	n/a	n/a	n/a	n/a	5.1	-1.4	10.0	2.8	4.3	-1.2	9.9	2.1	0.9	-6.9	8.5	2.6	0.9	-9.1	8.4	3.3	-2.2	-12.5	6.3	3.7
Mar	n/a	n/a	n/a	n/a	6.1	-1.8	13.8	3.2	6.1	-0.9	18.6	3.2	7.1	-0.9	14.9	3.4	3.6	-2.4	12.8	3.3	3.4	-7.7	20.1	5.5
Apr	n/a	n/a	n/a	n/a	7.1	0.0	19.7	3.7	10.2	1.8	23.7	3.8	11.4	4.8	18.3	2.1	7.3	-2.3	24.2	4.7	7.4	-0.5	18.4	4.0
May	n/a	n/a	n/a	n/a	13.7	1.6	25.5	4.7	13.9	4.0	23.9	4.3	n/a	n/a	n/a	n/a	14.0	4.0	26.7	5.1	13.9	1.4	27.2	5.4
Jun	14.4	5.9	23.0	3.4	16.9	6.9	31.7	4.8	18.3	9.3	31.0	4.0	n/a	n/a	n/a	n/a	14.1	2.2	32.6	5.0	15.0	3.2	29.0	4.6
Jul	17.9	9.2	30.9	4.3	18.8	9.4	30.6	4.9	17.1	9.7	27.1	3.4	17.0	7.2	27.4	4.1	18.5	6.0	33.3	5.6	16.4	6.3	25.9	3.4
Aug	18.6	10.1	29.5	4.2	16.9	8.8	27.3	3.9	17.6	10.0	29.4	4.2	18.4	7.8	32.0	5.0	17.9	7.3	31.2	5.2	17.0	7.8	27.9	4.0
Sep	14.2	5.8	25.3	3.8	11.5	3.4	23.4	3.4	11.9	3.7	20.2	2.9	14.0	2.4	31.0	5.4	12.1	3.0	24.6	3.7	12.8	0.2	24.0	4.2
Oct	10.1	1.9	17.3	2.7	9.8	2.8	18.4	2.7	8.3	0.5	11.7	1.9	6.9	-0.3	16.6	3.3	7.5	-0.1	16.0	3.5	-	-	-	-
Nov	3.1	-6.6	11.9	4.4	1.9	-6.8	9.2	3.3	7.8	1.5	14.5	2.7	3.6	-7.1	11.6	3.1	4.8	-2.2	11.6	3.0	-	-	-	-
Dec	2.4	-6.3	10.0	3.5	1.9	-4.8	8.4	2.8	-0.7	-10.9	8.6	3.5	0.3	-8.5	6.6	2.4	1.8	-6.8	8.4	2.7	-	-	-	-

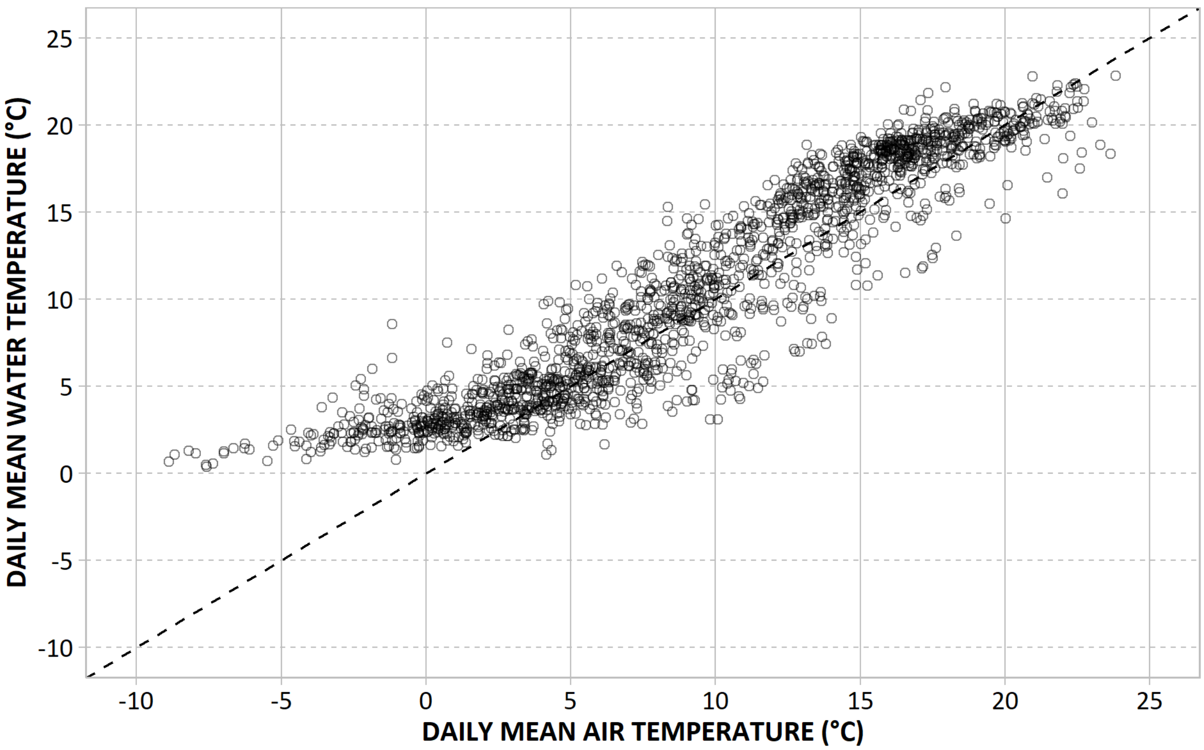
[1] "Avg", "Min", "Max" and "SD" denote the monthly average, minimum, maximum and standard deviation of air temperatures.

[2] "n/a" indicates that TidbiTs were not installed.

[3] Blue and orange shadings highlight minimum and maximum temperatures respectively.

[4] "-" indicates that TidbiT data has not yet been collected.

Figure 13. Relationship between daily average water and air temperature in the Quinsam River (QUN-AT) between May 2014 and October 2019. Dashed line denotes 1:1 line.



3.2. <u>Hydrology</u>

Quality assured data collected by the Water Survey of Canada were available until the end of 2018 (Year 5). Hydrographs for 2014–2018 at sites on the Quinsam River are presented in Figure 14 and Figure 15; hydrological metrics (Indicators of Hydrologic Alteration) for these years are presented in Table 27.

Flow measured in 2018 was within the range of previous years. For all years, discharge was low during the summer period, with minimum mean daily discharge of <0.5 m^3/s measured in the mainstem, downstream of the diversion facility (when it was not operating). It is also notable that maximum discharge was particularly high during the incubation periods for Pacific salmon species that emerged in 2015 and 2017, reflecting floods during December 2014 and November 2016.

Figure 14. Discharge measured on the Quinsam River upstream of Campbell River (Map 2) during 2014–2018.

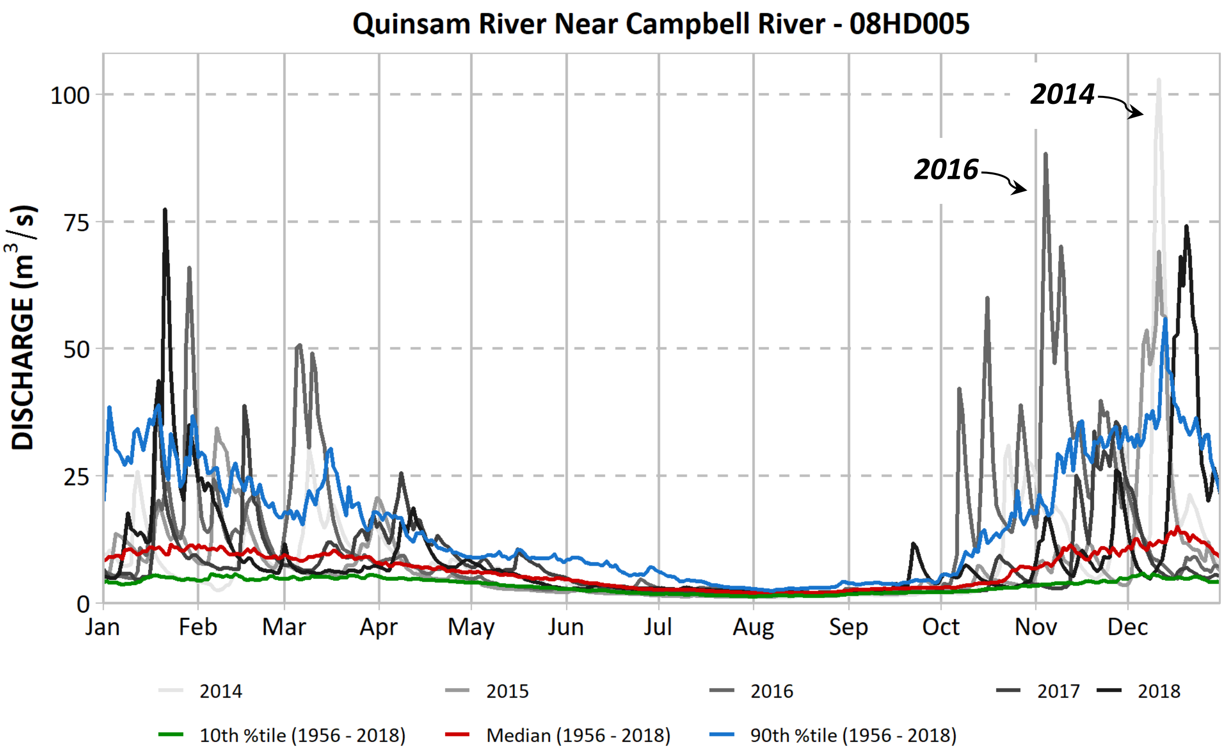


Figure 15. Discharge measured on the Quinsam River at Argonaut Bridge (Map 2) during 2014-2018.

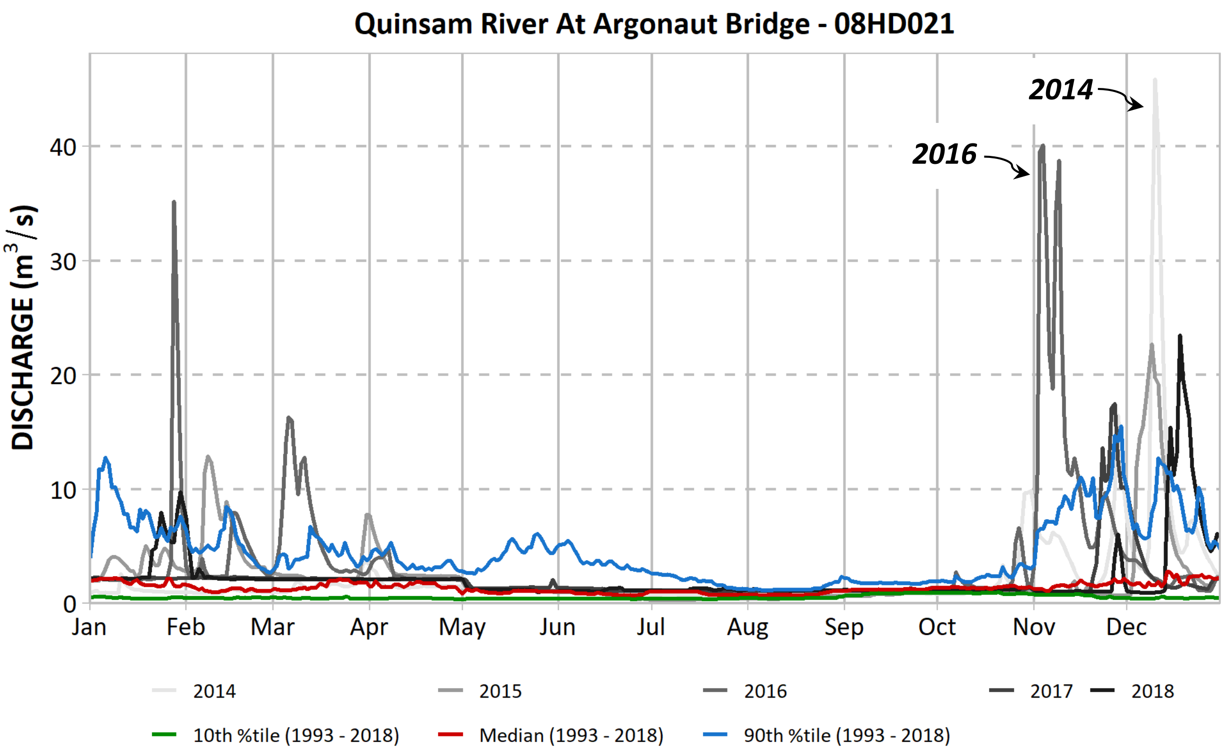


Table 27. Hydrological metrics calculated on the Quinsam River for 2014–2018. See Map 2 for hydrometric gauge locations.

Gauge	Year	Hydrological Metric (m³/s)						
		Minimum Mean Discharge (m³/s)			Maximum Discharge During Spawning and Incubation Periods[1]			
		1-Day Min.	3-Day Min.	30-Day Min.	Coho Salmon	Steelhead	Chinook Salmon	Pink Salmon
08HD021	2014	0.442	0.448	0.565	3.63	3.63	3.63	3.63
	2015	0.265	0.270	0.328	45.9	7.91	45.9	45.9
	2016	0.987	0.994	1.03	35.2	16.3	35.2	35.2
	2017	0.717	0.718	0.95	40.1	2.3	40.1	40.1
	2018	0.907	0.917	1.07	17.5	2.1	17.5	17.5
08HD005	2014	1.15	1.16	1.30	30.4	30.4	30.4	30.4
	2015	1.23	1.24	1.32	103	20.9	103	103
	2016	1.99	2.00	2.16	69.1	50.8	69.1	69.1
	2017	1.97	1.98	2.01	88.4	38.9	88.4	88.4
	2018	2.06	2.06	2.14	77.5	18.7	77.5	77.5

[1]For fall spawners, this metric was calculated based on the discharge between the start of spawning the previous year and fry emergence during the current year.

3.3. Invertebrate Drift

3.3.1. Quinsam River Invertebrate Drift

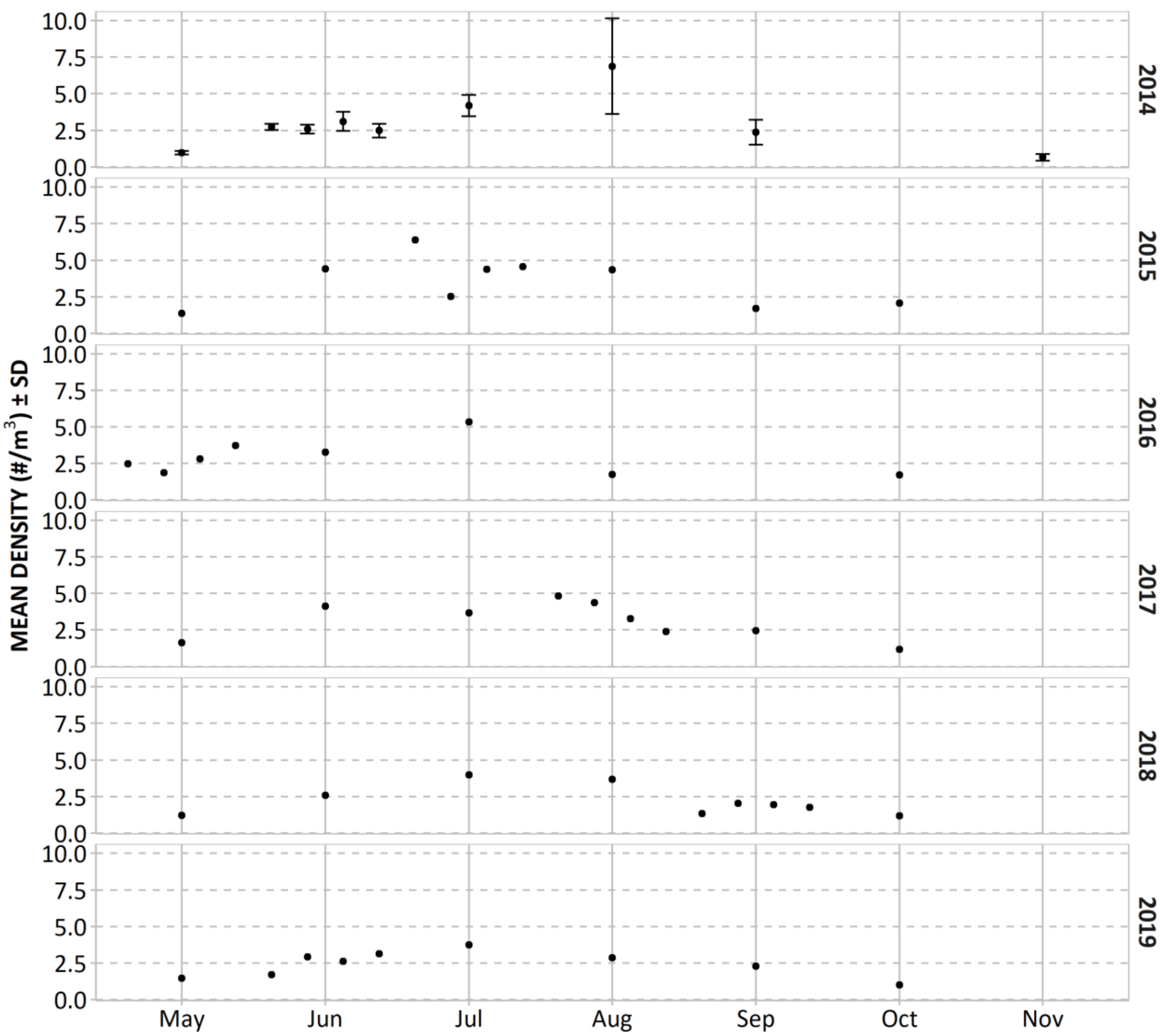
3.3.1.1. Overview

Results relating to invertebrate drift density (individuals/m^3) and biomass (mg/m^3) are provided in subsequent sections for the Quinsam River as potential indicators of changes in fish abundance. Supplementary invertebrate drift results relating to Simpson's family-level diversity index (1-λ), richness (# families), CEFI, and cluster analysis are provided in Appendix C. 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 across sampling dates in Year 6 (Figure 16). Density reached a peak of 3.74 individuals/m^3 in July 2019, with lower values observed earlier and later in the growing season (e.g., 1.48 individuals/m^3 in May; 1.00 individuals/m^3 in October; Figure 16). Density measured at weekly intervals during June ranged from 1.71 - 3.15 individuals/m^3 (Figure 16). Mean density in 2019 was within the range of values observed in previous years (0.65 – 6.88 individuals/m^3; Figure 16).

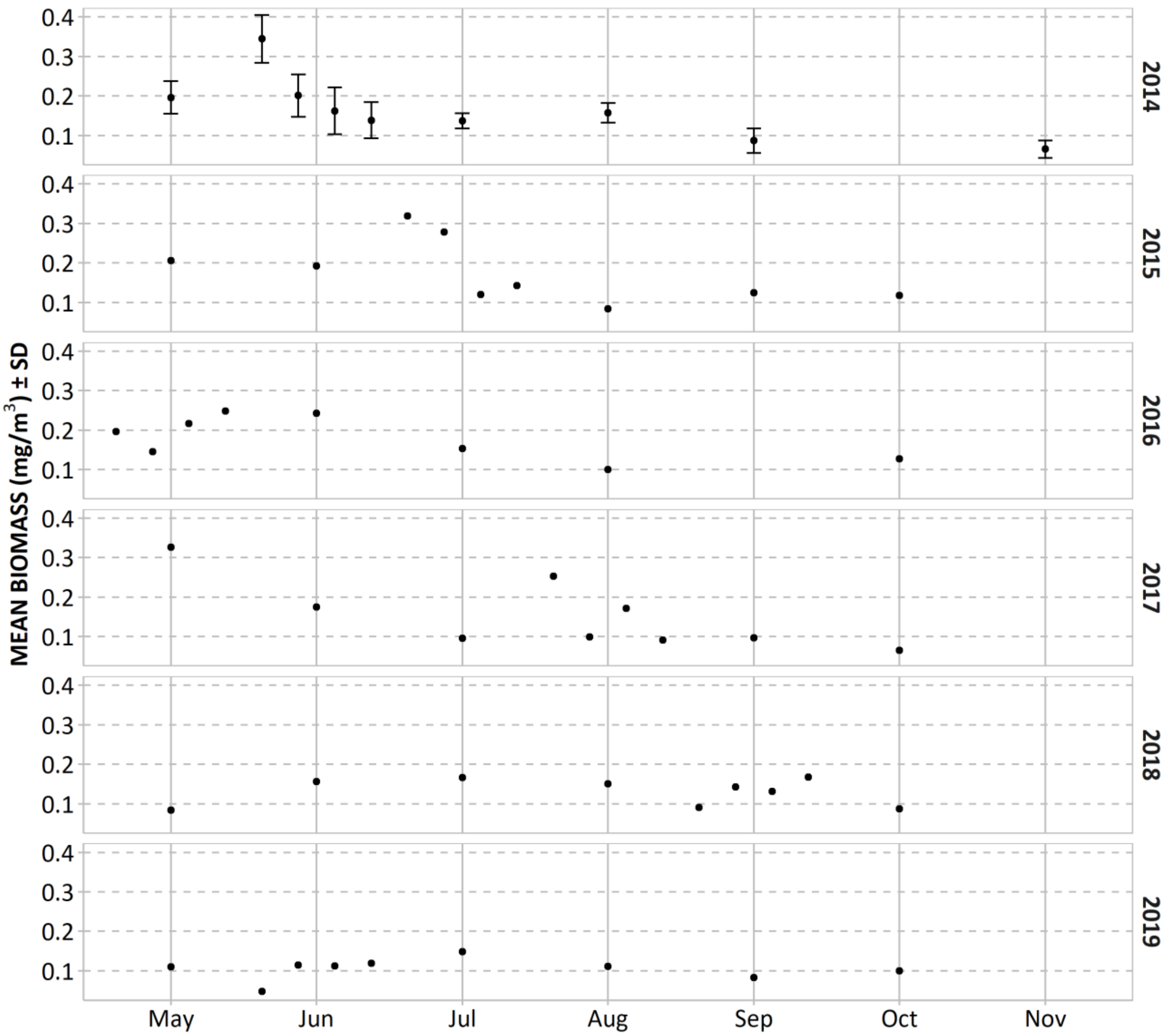
Figure 16. Drift invertebrate density (all taxa) in the Quinsam River throughout 2014 - 2019.



3.3.1.3. Biomass

Invertebrate drift biomass in the Quinsam River ranged from 0.05 – 0.15 mg/m³ in Year 6, which is lower than observed in previous years (0.06 – 0.34 mg/m³; Figure 17). Biomass was variable throughout Year 6, with the annual maximum value of 0.15 mg/m³ observed in July 2019, which was lower than maxima observed in previous years of JHTMON-8.

Figure 17. Drift invertebrate biomass (all taxa) in the Quinsam River throughout 2014 - 2019.



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 6 is provided in Table 28. Note that, in some instances, a taxonomic level higher than family is listed (e.g., Plecoptera), 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). Baetidae was the most dominant family in Year 6, as it was the dominant family on six of nine sampling dates and ranked second within the top five families contributing to biomass on two other sampling dates. True flies were also consistently present in the top five, with two or more true fly families present on seven of nine sampling dates. The contribution to biomass of individual mayfly families ranged from 3.1% to 64.0% while individual true fly families ranged from 4.0% to 43.2%.

Other taxa sometimes present in the top five included Caddisflies (Limnephilidae and Philopotamidae), true bugs (Aphididae), stoneflies (unspecified Plecoptera families), mites (Torrenticolidae), beetles (Cantharidae, Chrysomelidae, Elmidae, and Coccinellidae) and spiders (Araneae).

A summary of the top five families contributing to biomass across all JHTMON-8 years in the Quinsam River is provided in Table 29. These results show consistencies in the top five families across years, with Baetidae comprising the top family in five of six years and present in all six years along with two other families (Chironomidae and Simuliidae). In all years, these three families comprised 37.2–49.5% of the biomass. Ephemeroptera and Diptera have been shown to be important invertebrate taxa for juvenile salmonids (Johnson and Ringler 1980, Rader 1997), and for the most part, dominated the top five families during each sampling date in Year 6 (2019) and other years.

Table 28. Top five families contributing to invertebrate drift biomass (all taxa) in the Quinsam River in Year 6. Names in parentheses represent taxa higher than families in instances where family level classifications were unavailable.

QUN-IV	13-May-19	QUN-IV	6-Jun-19	QUN-IV	12-Jun-19	QUN-IV	20-Jun-19	Key
Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	True Flies
								Mayflies
Baetidae	64.0	Baetidae	23.3	Baetidae	21.8	(Plecoptera)	16.9	Caddisflies
Simuliidae	6.8	Chironomidae	14.4	Philopotamidae	16.7	Baetidae	16.7	True Bugs
Ameletidae	5.4	Limnephilidae	11.9	Simuliidae	12.8	Simuliidae	10.1	Stoneflies
Chironomidae	4.6	Torrenticolidae	5.6	(Araneae1)	7.0	Cantharidae	6.9	Spiders
Leptophlebiidae	3.1	Simuliidae	4.1	Chironomidae	6.7	Chironomidae	6.2	Beetles
Sum	83.9	Sum	59.4	Sum	64.9	Sum	56.8	Mites

QUN-IV	27-Jun-19	QUN-IV	11-Jul-19	QUN-IV	12-Aug-19	QUN-IV	12-Sep-19	QUN-IV	9-Oct-19
Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass	Family	% of Total Biomass
Baetidae	33.4	Baetidae	24.3	Baetidae	44.0	Chironomidae	22.0	Simuliidae	43.2
Torrenticolidae	11.2	Torrenticolidae	10.1	Heptageniidae	22.5	Baetidae	19.5	Torrenticolidae	30.9
Simuliidae	8.5	Simuliidae	9.9	Chironomidae	11.6	Simuliidae	14.3	Chrysomelidae	4.8
(Araneae1)	6.4	Chironomidae	7.8	Aphididae	6.0	Coccinellidae	8.1	Chironomidae	4.0
Elmidae	5.3	Dolichopodidae	6.7	Simuliidae	5.2	Aphididae	7.4	(Plecoptera)	2.9
Sum	64.8	Sum	58.9	Sum	89.3	Sum	71.3	Sum	85.8

Table 29. Annual top five families contributing to invertebrate drift biomass (all taxa) in the Quinsam River throughout Years 1 to 6. Names in parentheses represent taxa higher than families in instances where family level classifications were unavailable.

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

QUN-IV Family	2017 % of Total Biomass	QUN-IV Family	2018 % of Total Biomass	QUN-IV Family	2019 % of Total Biomass
Baetidae	18.0	Baetidae	21.3	Baetidae	28.3
Chironomidae	12.0	Simuliidae	12.6	Simuliidae	12.8
Simuliidae	9.4	Chironomidae	12.1	Chironomidae	8.4
Empididae	8.6	Hydropsychidae	6.0	Torrenticolidae	7.8
Bibionidae	5.7	(Araneae)	3.8	Heptageniidae	3.1
Sum	53.8	Sum	55.9	Sum	60.4

Key
True Flies
Mayflies
Caddisflies
True Bugs
Stoneflies
Spiders
Beetles
Mites

3.3.2. Comparison of Kick Net and Drift Net Sampling Methods

Invertebrates collected using kick net sampling were almost exclusively aquatic taxa (99.6–100%) in the Quinsam River whereas drift sampling captured 64.2–79.3% aquatic invertebrates (based on biomass; Table 30). 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 are more likely to have entered the stream from terrestrial or semi-aquatic (riparian) habitats.

The contribution of individual families to invertebrate biomass differed between the two sampling methods (Table 31). In the Quinsam River, two groups (true flies and mayflies) accounted for the majority of the biomass in drift net samples and most of the top five families comprised these taxa on all dates, whereas a wider range of families were present during kick sampling including Hydropsychidae (caddisflies), Gomphidae (dragonflies), Astacidae (crayfish), and Lumbricidae (earthworm). Overall, the taxa present in the kick net samples were more diverse within and among sampling dates than taxa present in drift net samples.

Table 30. 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

Table 31. 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.

Date	Driftnet		Kicknet	
	Family	% of Biomass	Family	% of Biomass
9/16/2015	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
9/13/2017	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
9/12/2018	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
9/12/2019	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

Key
True Bugs
Aquatic Worm
Mites
True Flies
Mayflies
Caddisflies
Crustacean
Dragonflies
Stoneflies
Beetles
Earthworm

4. DISCUSSION

4.1. Status

JHTMON-8 is at an interim stage and analysis 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, describes key results, and provides an overview of the planned approach to test the hypothesis, including how work undertaken in previous years will be used in the analysis.

4.2. $H_0$1: Annual population abundance does not vary with time (i.e., years) over the course of the Monitor

This hypothesis focuses on juvenile fish (BC Hydro 2018a). 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 7 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 (2014–2019), variability in annual outmigration data provided by DFO has been greatest for wild Chinook Salmon (~600 to ~114,000 fry) and lower for wild Coho Salmon (~22,000 to ~47,000 smolts) and steelhead (~3,000 to ~11,000 smolts) (Figure 6).

Work undertaken in Year 5 to compile, digitize, and analyze juvenile fish outmigration data collected at the Quinsam Hatchery fence prior to JHTMON-8 (since the 1970s; Figure 7) will substantially increase the statistical power of analysis to quantify variability in juvenile fish abundance in the Quinsam River. 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, spawner-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 (discussed further below in Section 4.7 in relation to H_06). These spawner-recruitment relationships will be used to test the other hypotheses discussed below; specifically, analysis will be undertaken to quantify whether variability in factors corresponding to each hypothesis (e.g., WUA for H_02) explains variability in the spawner recruitment relationships.

In Year 6, we completed a review of capture efficiency estimates calculated based on juvenile mark-recapture experiments conducted at the Quinsam Hatchery salmon counting fence (Section 3.1.2.2). Based on our review, we propose to modify the analysis methods for the JHTMON-8 study period to increase the accuracy of the results; however, we welcome feedback from DFO or BC Hydro regarding this proposed change.

4.3. $H_0$2: Annual population abundance is not correlated with annual habitat availability as measured by Weighted Usable Area (WUA)

WUA (in m^2) provides an index of habitat availability calculated using relationships developed between flow and the area of different habitats, accounting of differences in habitat suitability across different flows (Lewis *et al.* 2004). In Year 5, we quantified WUA for different life stages of JHTMON-8 priority species. This analysis 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 spawner-recruitment relationships (discussed above) and then test whether variability in WUA explains variability in the spawner-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 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 spawner-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.

4.4. $H_0$3: Annual population abundance is not correlated with water quality

$H_0$3 focuses on juvenile fish. Year 6 water quality results were generally consistent with results for Year 1 through Year 5. Results from JHTMON-8 to date show that the Quinsam River is fairly typical of streams in coastal BC watersheds with low nutrient concentrations (oligotrophic), near-neutral pH, and low turbidity during baseflow. Results show that 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, although maximum summer water temperatures were lower in Year 6 than in previous years of JHTMON-8. For example, MWMxT during the rearing period for stream-resident species and life stages (e.g., juvenile Coho Salmon and steelhead) was lower in Year 6 (19.8ºC) than during Year 1 to 5 (20.8–23.1ºC), although MWMxT measured in all six years still exceeded the upper limit of the optimum temperature ranges for the rearing life stage of species that include juvenile Coho Salmon (upper limit of 16ºC) and Rainbow Trout (upper limit of 18ºC). As observed in previous years, concentrations of DO less than the provincial WQG-AL for the protection of buried embryos/alevins were recorded in Year 6 during reported incubation periods (Burt 2003) for resident Rainbow Trout and steelhead. Measurements in Year 6 also indicated that DO concentrations were below the WQG-AL range during the start of the Pink Salmon incubation period.

Analysis to test $H_0$3 will be undertaken separately for individual species and water quality variables. Analysis will initially focus on the ten-year period of the monitor, 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 (Dinn *et al.* 2016). 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. In Year 4, an initial screening analysis of the water quality variables was completed. This 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 nonetheless be considered. The Year 4 screening analysis generally showed that interannual variability in many of the water quality variables was low, which may limit the power of the final analysis to quantify effects of water quality (if present) on fish abundance. As an alternate line of evidence, it will therefore also be important to 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.

4.5. $H_0$4: Annual population abundance is not correlated with the occurrence of flood events

As part of JHTMON-8, data collected by the Water Survey of Canada have been collated and analyzed to quantify hydrologic variability in the Quinsam River. Analysis of data collected for the first five years of JHTMON-8 shows that the largest flood event occurred in December 2014, when flow at the mouth of the Quinsam River briefly peaked at just over 100 m^3/s (Figure 14). Particularly high flows also occurred in November 2016, when flow at the mouth of the Quinsam River reached approximately 85 m^3/s (Figure 14). For all years, discharge was low during the summer low-flow period, with minimum mean daily discharge of <1.0 m^3/s measured in the Quinsam River during each year in the summer (when the diversion facility was not operating).

This hypothesis will be tested by quantifying high flow metrics separately for each watershed based on discharge measured at gauges maintained by the Water Survey of Canada. Relationships between the occurrence of floods and juvenile fish abundance will then be analyzed. Further, 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 following a review conducted 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. We plan to consider additional metrics in future years; e.g., that quantify the duration of high flows.

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.

4.6. $H_0$5: Annual population abundance is not correlated with food availability as measured by aquatic invertebrate sampling

Invertebrate drift data have now been collected for six growing seasons for the Quinsam River. There are no clear differences in invertebrate drift biomass among years, although data indicate that invertebrate drift biomass was lower in 2019 (Year 6) relative to previous years (average 2019 biomass was ~37% lower than the average of all data collected in previous years) (Figure 17). The reason for this is uncertain; however, low flow in the Quinsam River during spring 2019[5] caused by low precipitation may have been a contributing factor as antecedent low flow conditions can suppress macroinvertebrate biomass in streams (reviewed by Rolls *et al*. 2012).

Otherwise, results show that invertebrate drift biomass generally tends to decline during the growing season (Figure 17), while analysis of similarity in the invertebrate assemblages shows consistent trends among years, with distinct communities present early in the growing season (May and June) relative to later in the growing season (Appendix C). The potential effect of diversion operations on changes in community structure has not been analyzed but the seasonal patterns in invertebrate community that were observed are consistent with those that we would expect in unregulated streams. Therefore, we expect that the broad seasonal trends are at least largely driven by natural changes (phenology) in invertebrate community composition.

These seasonal trends have potential implications for juvenile salmonid productivity, because 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 (e.g., due to a decrease in the biomass of taxa preferred by salmonids), which could theoretically affect juvenile growth and abundance.

Relationships between invertebrate drift and fish abundance will be examined in Year 10 (2023). To test $H_0$5, we plan to examine whether variability in invertebrate drift biomass explains variability in species-specific spawner recruitment curves for JHTMON-8 priority species. If robust spawner recruitment curves cannot be established (due to weak or no relationships between adult and juvenile fish), then we plan to use juvenile fish abundance as the dependent variable in the analysis. $H_0$5 would be rejected if invertebrate biomass is shown to be a statistically significant predictor of juvenile fish abundance, although it will be necessary to then evaluate the effect size to infer biological significance. Furthermore, we plan to evaluate invertebrate density as a secondary measure, or proxy, of food abundance; however, consistent with the TOR (BC Hydro 2018a), we expect to use invertebrate biomass as the primary measure of food availability because it is a direct measure of the energy available for fish to consume.

[5] Based on reviewing discharge data for Water Survey of Canada gauge 08HD005.

If strong relationships are detected between fish abundance and invertebrate biomass/density, 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.

Results so far show marked within-year (seasonal) variability in invertebrate drift biomass but variability in invertebrate drift biomass among years is generally low (Figure 17). 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 in 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 premised on the assumption that invertebrate drift measured at a single index site is representative of conditions experienced by fish in the wider watershed.

4.7. $H_0$6: Annual smolt abundance is not correlated with the number of adult returns (Quinsam River)

We propose to test this hypothesis by constructing spawner-recruitment relationships to quantify the relationship between the abundance of adult fish and the subsequent recruitment of juvenile fish each year. This hypothesis will therefore be tested using juvenile and adult fish abundance data. This analysis will use the juvenile abundance data collected at the Quinsam Hatchery salmon counting fence and the adult escapement data collected by DFO. Compilation of the historical juvenile abundance dataset for the Quinsam River in Year 5 (Abell *et al.* 2019) provides the potential to substantially increase the duration of the dataset that can be analyzed to test this analysis, thereby increasing statistical power.

In Year 7 (2020), we plan to construct interim spawner-recruitment relationships before final analysis is completed in Year 10 (2023). The final spawner-recruitment relationships will be used in Year 10 to test $H_0$6; i.e., to confirm whether the abundance of outmigrating juveniles is correlated with the abundance of corresponding previous adult returns. Spawner-recruitment relationships can then be analyzed as part of analysis to test the remaining hypotheses; i.e., to quantify whether variability in the environmental factors that have been identified can explain variability in the spawner-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).

At a minimum, we propose to test $H_0$6 separately for Chinook Salmon, Coho Salmon and Pink Salmon. Quantitative analysis is not proposed to test $H_0$6 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. ADDITIONAL TASKS FOR YEAR 7 (2020) AND SUBSEQUENT YEARS

Each year, we have undertaken additional analysis tasks to streamline final hypothesis testing in Year 10. Additional tasks proposed for the remaining years of JHTMON-8 are summarized in Table 32.

Table 32. Additional tasks planned for the remainder of JHTMON-8.

Year Number (Year)	Task	Hypothesis
6 (2019)	Reviewed capture efficiency estimates calculated based on juvenile mark-recapture experiments conducted at the Quinsam Hatchery salmon counting fence (Appendix A, this report)	$H_0$1, $H_0$6
7 (2020)	• Updated JHTMON-8 outmigration estimates based on the outcomes of our review • Construct and review initial spawner-recruitment relationships (to be used in Year 10)	$H_0$1, $H_0$6
8 (2021)	Complete initial hypothesis testing for one hypothesis ($H_0$3) to demonstrate proof of concept for proposed analysis approach	$H_0$3
9 (2022)	Prepare predictor variables for final analysis, including WUA estimates	All
10 (2023)	Complete final analysis to test hypotheses and address management questions	All

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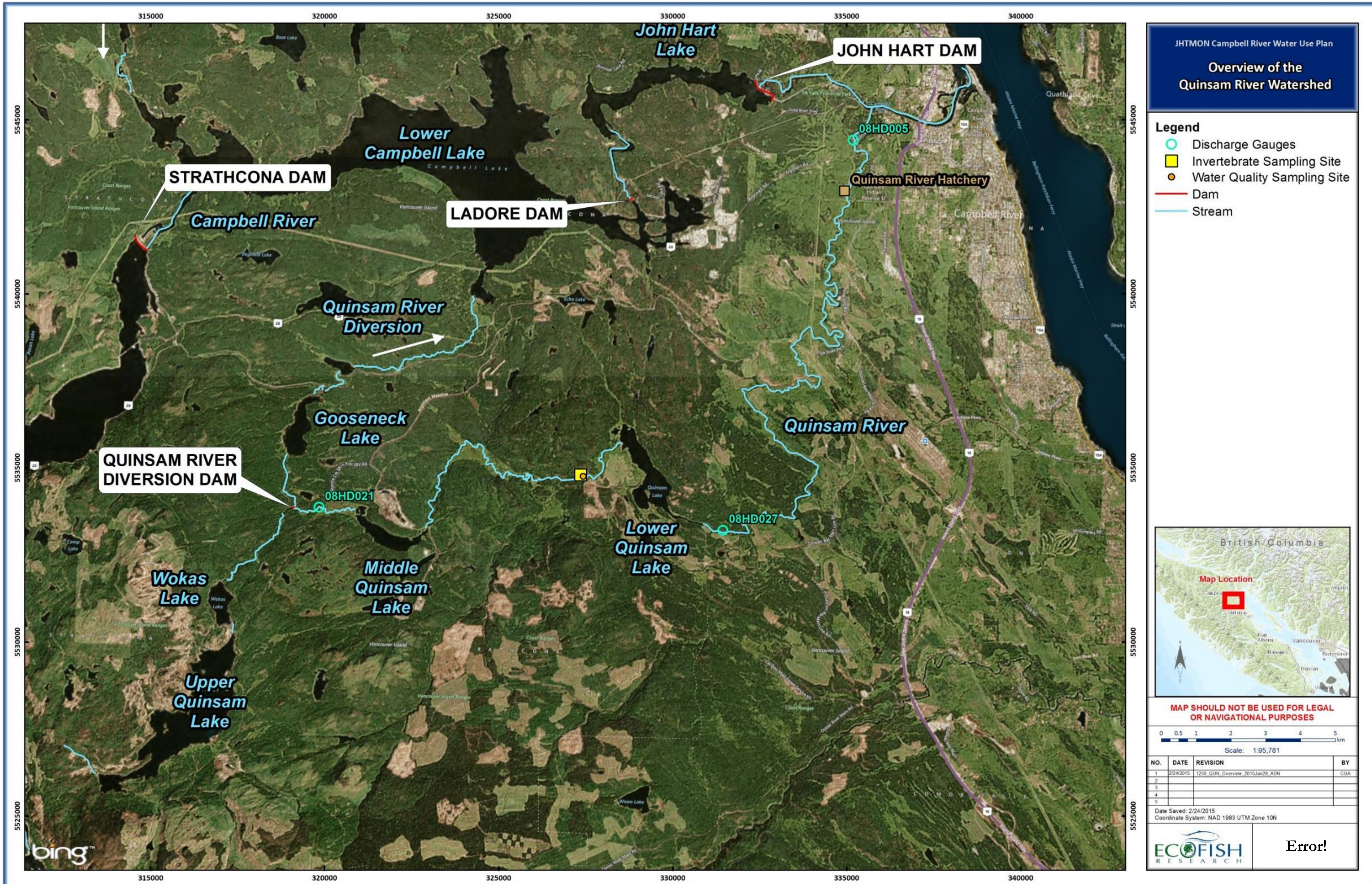
PROJECT MAPS



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APPENDICES

Appendix A. Review of Capture Efficiency Estimates

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1. BACKGROUND

During data review, DFO Quinsam Hatchery staff noted there was high uncertainty regarding some capture efficiency values estimated in Year 6 (Fortkamp pers. comm. 2019). Specifically, DFO noted there was high variability in capture efficiency estimates calculated using mark-recapture experiments undertaken with Coho Salmon smolts in Year 6. DFO hypothesized that this variability was due to experimental error, such as trap malfunction, and that using these anomalous capture efficiency estimates without appropriate adjustment would inflate uncertainty in total outmigration estimates of JHTMON-8 priority species (Chinook Salmon, Coho Salmon, and steelhead).

DFO's communication prompted a review of Year 6 and historical capture efficiency estimates as part of data JHTMON-8 QA/QC procedures. This was informed by a phone conference between Ecofish staff and DFO Quinsam Hatchery staff on December 19, 2019, as well as associated e-mail discussions between DFO and Ecofish. The objective of the review was to examine variability in capture efficiency estimates to inform how to best use capture efficiency estimates in JHTMON-8 analysis to maximize the accuracy of the results. In particular, an aim was to understand whether apparent outlier values were likely caused by experimental error, or whether capture efficiency estimates were highly variable overall and such values more likely reflect variability in environmental factors (e.g., flow) that strongly influence capture efficiency. A related aim was to understand whether it was appropriate to substitute suspected erroneous values with a long-term mean value.

2. METHODS

As part of the review, capture efficiency estimates calculated based on mark-recapture experiments were compiled, with estimates omitted that corresponded to experiments that were flagged by DFO as having high experimental error. A plot of capture efficiency estimates for the JHTMON-8 study period to date (2014–2019) was jointly evaluated by DFO and Ecofish. The group then discussed the potential for variability in capture efficiency estimates to be affected by differences in sampling method and the characteristics of the area in the vicinity of the counting fence. In particular, DFO identified that the presence of either tire or wooden booms approximately 40 m upstream from the fence in all years prior to 2015 (Figure 1) may have affected capture efficiency estimates, although the potential causal mechanism is uncertain. These booms were permanently removed during or following high flows in fall/winter 2014.

Based on these discussions, the following hypotheses were identified regarding variability in capture efficiency estimates:

i. There are intra-seasonal trends in capture efficiency estimates for fry and smolts.

ii. There is a long-term declining trend in capture efficiency estimates, possibly due to increasing predation of marked fish.

iii. Removal of booms in late 2014 affected capture efficiency estimates for fry and smolts.

To assess the existence of intra-seasonal and inter-annual trends, we fitted generalized additive models (Wood 2006) where the life-stage-specific (fry or smolt) capture efficiency was expressed as:

$$\textit{Capture efficiency} \sim s(Year) + s(Day\ of\ Year) + \varepsilon; \quad \varepsilon \sim N(\mu, \sigma^2)$$

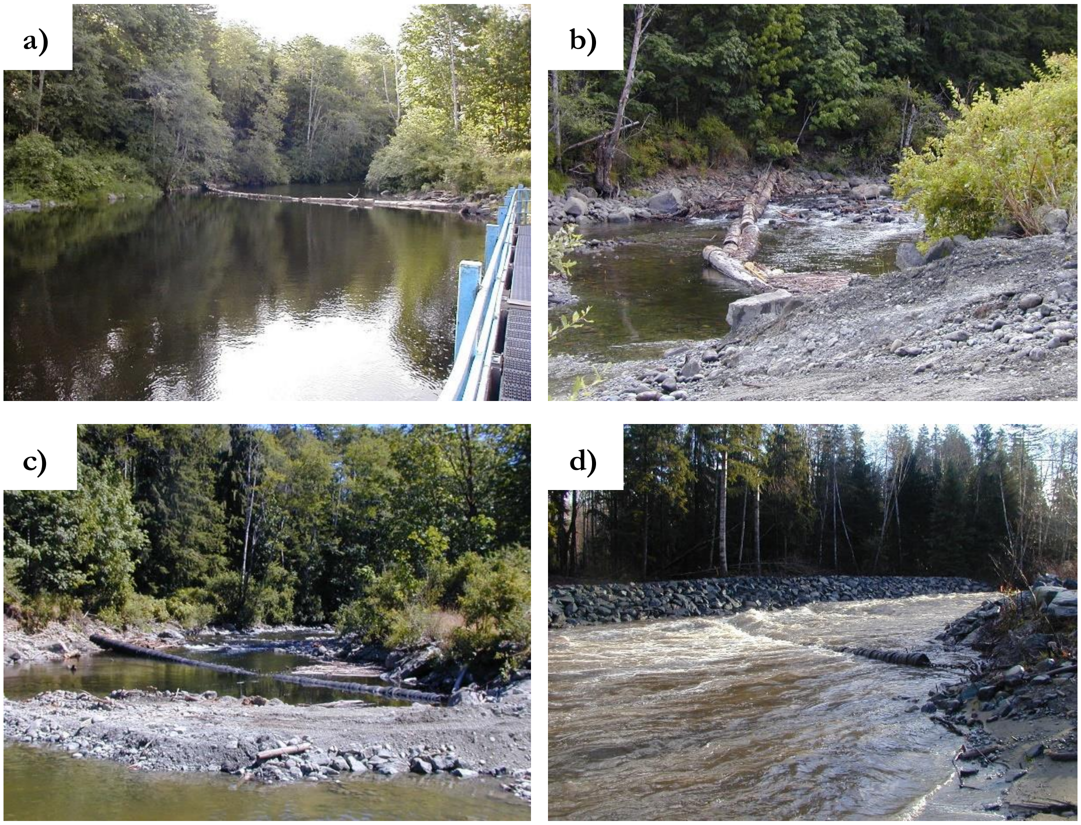
where the term *s(Year)* is a smooth term that represents the inter-annual trend and the term *s(Day of Year)* is a smooth term that represents the intra-annual trend in capture efficiency estimates. Models for fry and smolts were fitted separately.

To assess the effect of boom presence on capture efficiencies, we fitted a general linear model of the form:

$$\textit{Capture efficiency} \sim Boom + Stage + \varepsilon; \varepsilon \sim N(\mu, \sigma^2)$$

where the term *Boom* represents the presence of a boom (i.e., either pre or post 2015), and the term *Stage* represents life stage (i.e., either smolt of fry).

Figure 1. Booms present approximately 40 m upstream of the counting fence prior to 2015. a) wooden boom, photograph taken in 2000, b) wooden boom, photograph taken in 2008, c) tire boom, photograph taken in 2011, d) tire boom (date unknown).



3. RESULTS

3.1. Analysis of Variability

Review of capture efficiency estimates calculated based on mark-recapture experiments during the JHTMON-8 period (2014–2019) showed that smolt capture efficiency estimates were more variable than fry capture efficiency estimates (Figure 2), reflecting the smaller sample size used for the smolt mark-recapture experiments. Clear seasonal trends were not present in the capture efficiency estimates for the JHTMON-8 period. Average capture efficiency estimates were higher for smolts than fry, with capture efficiency estimates for the JHTMON-8 period generally well-described by a life stage-specific mean capture efficiency (0.026 for fry, and 0.070 for smolts; Figure 2).

For the longer period of 1996–2019 (when digital records are currently available), analysis of capture efficiency estimates addressed the hypotheses in Section 1**Error! Reference source not found.** r elating to 1) intra-seasonal trends; 2), inter-annual trends; and 3) an influence of removing booms in late 2014. We did not detect significant intra-seasonal variability for either life stage (p-value$_{fry}$ = 0.90, p-value$_{smolt}$ = 0.14); i.e., there was no indication that capture efficiency estimates consistently increased or decreased through the spring period. Further, there was no significant inter-annual trend in the capture efficiency estimates for smolts (p-value = 0.16) (Figure 3), but there was a significant decreasing trend in the capture efficiency estimates for fry (p-value = 0.02) (Figure 4). However, evaluation of Figure 4 indicated that the significant trend is largely due to lower estimates of capture efficiency starting in 2015, which is the first year that monitoring was undertaken following removal of the booms (Figure 1). This was supported by analysis that showed that the presence of booms had a statistically significant effect on capture efficiencies for both life stages (p-value$_{boom}$ = 0.016; p-value$_{stage}$ < 0.0000001). The removal of the booms seemingly resulted in lower capture efficiencies on average; mean capture efficiency for fry decreased from 0.037 to 0.024, whereas mean capture efficiency for smolts decreased from 0.084 to 0.071 (Figure 5).

Figure 2. Life-stage-specific capture efficiency estimates calculated based on mark-recapture experiments at the Quinsam Hatchery salmon counting fence during the JHTMON-8 period (2014–2019). Horizontal lines represent life stage-specific (fry and smolt) means and shaded regions represent 95% confidence intervals for the estimates of the means.

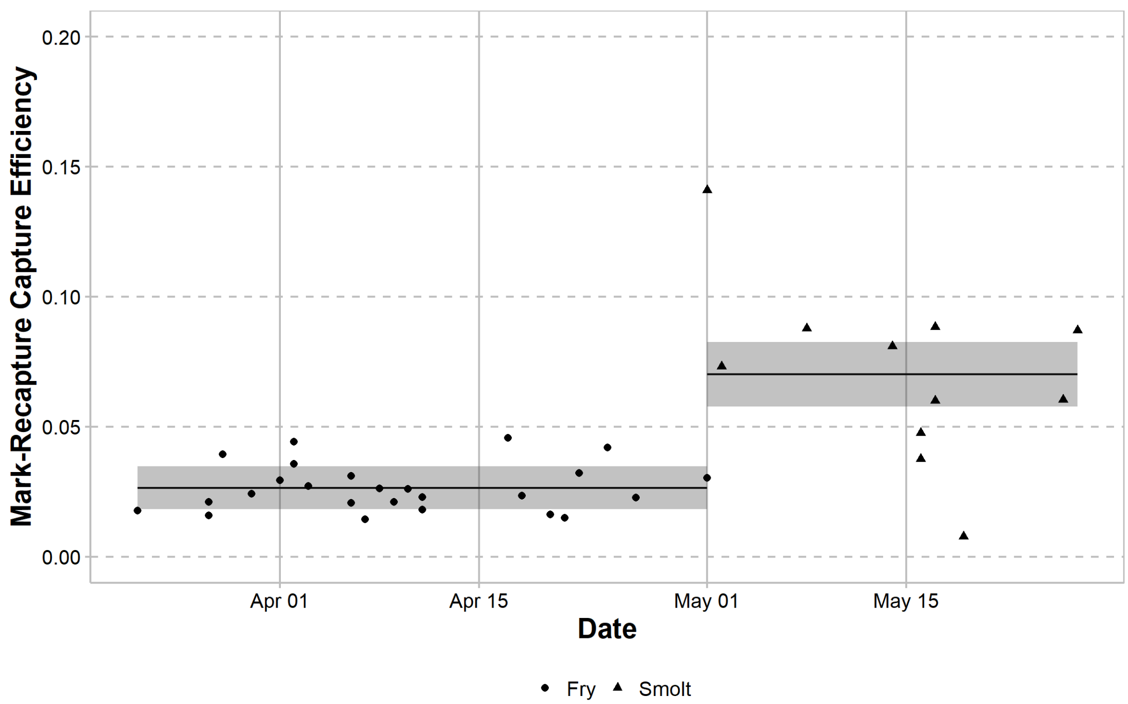


Figure 3. Capture efficiency estimates based on mark-recapture experiments using smolts, 1996–2019. Symbols represent annual experiment number to show intra-annual variability.

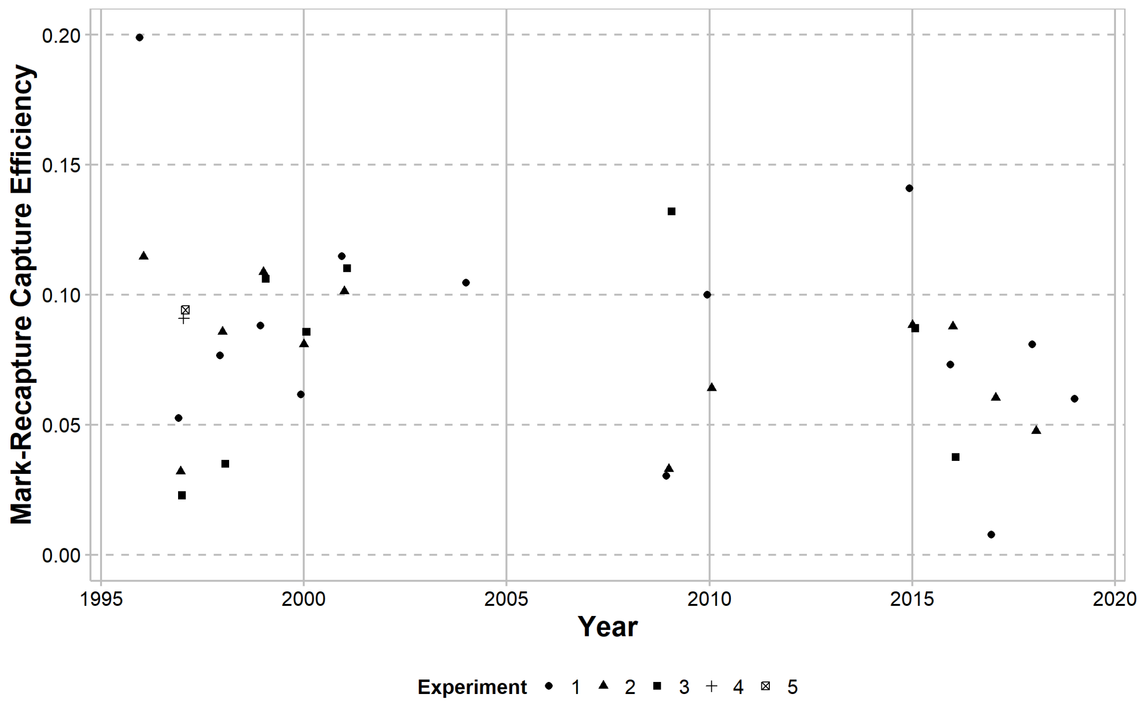


Figure 4. **Capture efficiency estimates based on mark-recapture experiments using fry, 1996–2019. Symbols represent annual experiment number to show intra-annual variability. The solid line represents mean capture efficiency and the shaded region represents the 95% confidence interval for the estimate of the mean.**

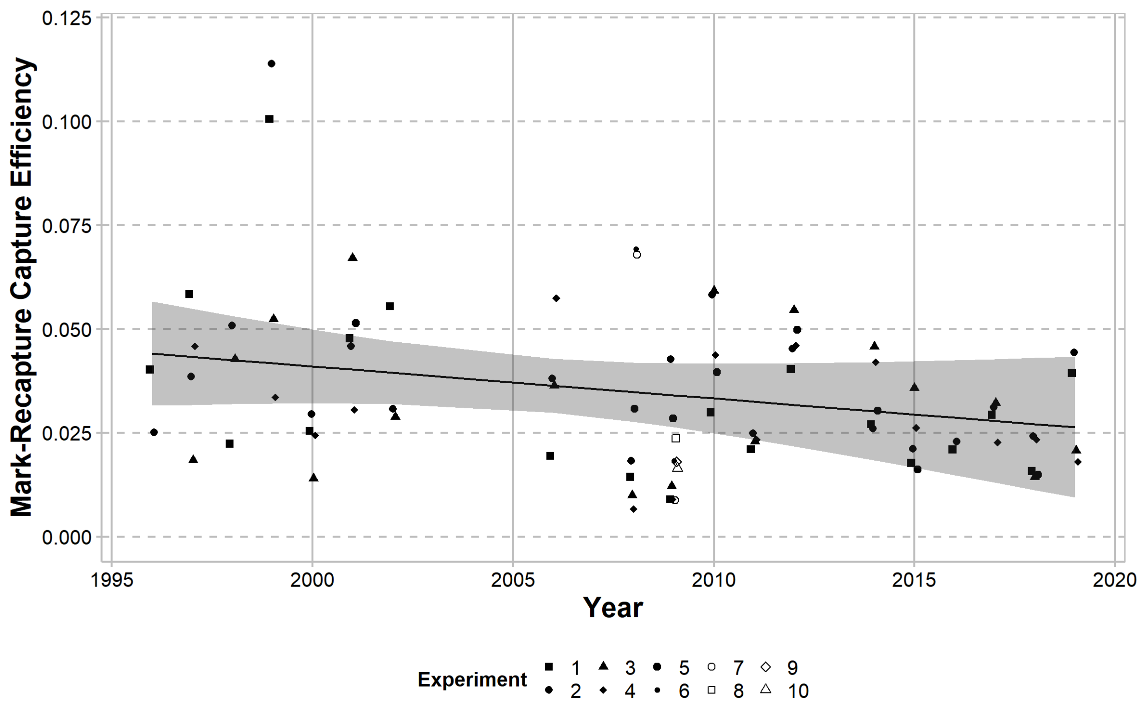
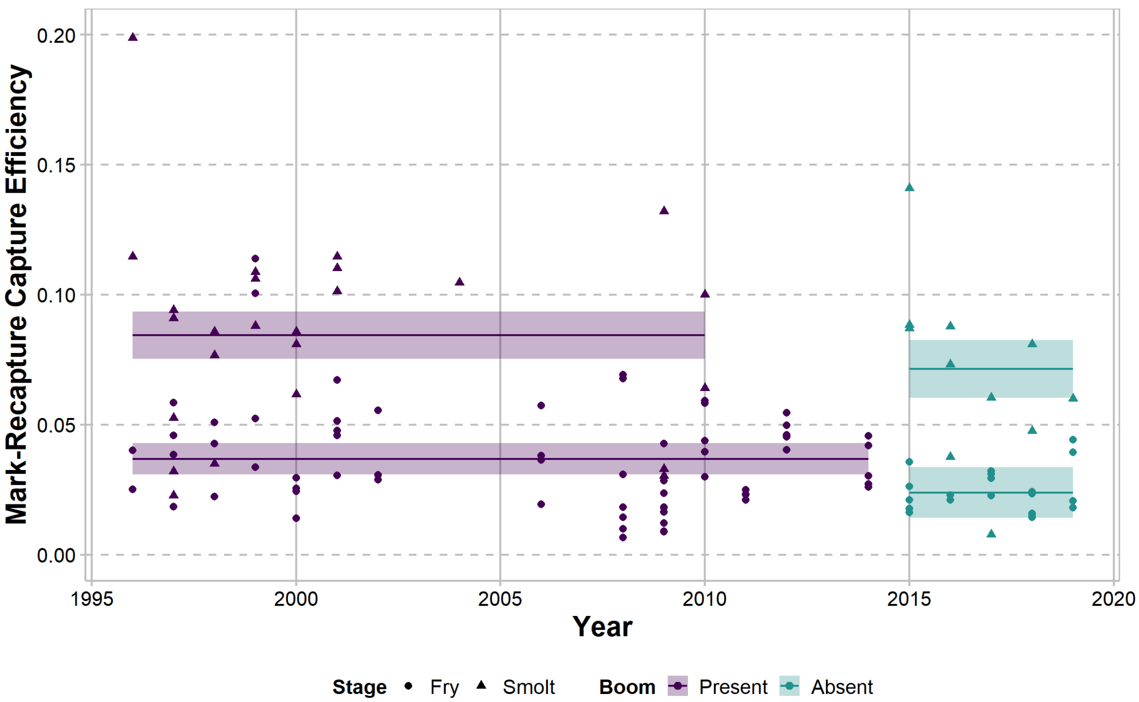


Figure 5. Life-stage-specific capture efficiency estimates based on mark-recapture experiments at the Quinsam Hatchery salmon counting fence, 1996–2019. Symbols represent life stages and colors represent the presence/absence of booms upstream of the counting fence. The horizontal lines represent mean capture efficiencies and the shaded regions represent the 95% confidence intervals for the estimates of the means.



3.2. Options to Use Capture Efficiency Estimates to Adjust Counts

Differences between the capture efficiency estimates for fry and smolts reflect both the differences in catchability between the life stages, and differences due to the way the traps are operated during the fry and smolt migration periods. For individual life stages, variability in capture efficiency estimates reflects observation error (variability introduced by differences in how experiments are undertaken) and process error (variability that is desirable to capture and is caused by differences in environmental factors among sampling events; e.g., flow or the precise configuration of traps). Regarding observation error, DFO noted that experimental details have varied slightly throughout the monitoring period; e.g., the timing of experimental releases has varied between midday, mid-afternoon and the evening, with variability also in the duration of holding times after using anaesthetic for marking[1].

When applying capture efficiency estimates to adjust count data, it is desirable to use estimates that best reflect process error and minimize bias due to observation error. Accounting for process error is supported by using capture efficiency estimates that are specific to the time of sampling (e.g., based on the most recent mark-recapture experiment), whereas minimizing bias due to observation error can be achieved by averaging capture efficiency estimates (e.g., using data for multiple years) to "smooth out" the influence of erroneous values. Potential options to use capture efficiency estimates in JHTMON-8 analysis include:

a) *Status quo*: apply the capture efficiency estimate from the most recent experiment but discard capture efficiency estimates from failed experiments and replace with estimates based on professional judgement. This option is prone to introducing bias by including occasional experimental results with high observation error, and provides no alternative for years when mark-recapture experiments were unsuccessful.

b) *Life-stage and period-specific means*: apply the mean values shown in Figure 5. This option would disregard inter- or intra-annual variability, but would provide a consistent approach for years when mark-recapture experiments were unsuccessful. Note that the results of mark-recapture experiments conducted in an individual year would therefore only have a small influence on the capture efficiency estimates used that year.

c) *Life-stage-specific annual means*: in years when mark-recapture experiments are deemed successful (i.e., no clear source of experimental error is identified), and there was a minimum number of experiments conducted (proposed sample sizes of $n \geq 3$ for fry and $n \geq 2$ for smolts), apply an annual mean capture efficiency that is specific to either fry or smolts (Figure 6, Table 1). When those criteria are not met, apply stage- and period-specific means, as per Figure 5. This option would account for inter-annual variability, although it would disregard intra-annual variability; i.e., variability within the spring sampling period. Based on inspection of Figure 5, this approach seems reasonable as variability among years generally seems greater than variability within years (although there are several exceptions).

For the final JHTMON-8 analysis, we propose to adopt option c described above to revise outmigration estimates for the JHTMON-8 study period because it is expected to yield the most accurate results relative to other options identified. However, we welcome feedback from DFO or BC Hydro regarding this proposed change. We expect this change would have minor effects on observed inter-annual trends in estimates of total outmigration of priority species. Changes to

[1] DFO is considering revising their data collection sheets to better record such details.

the magnitude of annual outmigration estimates could be larger although, on average, changes are expected to improve the accuracy of these estimates.

Figure 6. Life-stage-specific capture efficiency estimates of the Quinsam Hatchery salmon counting fence, years 1996-2019. Symbols represent experiments. Solid circles represent annual means, and vertical lines represent 1 SD.

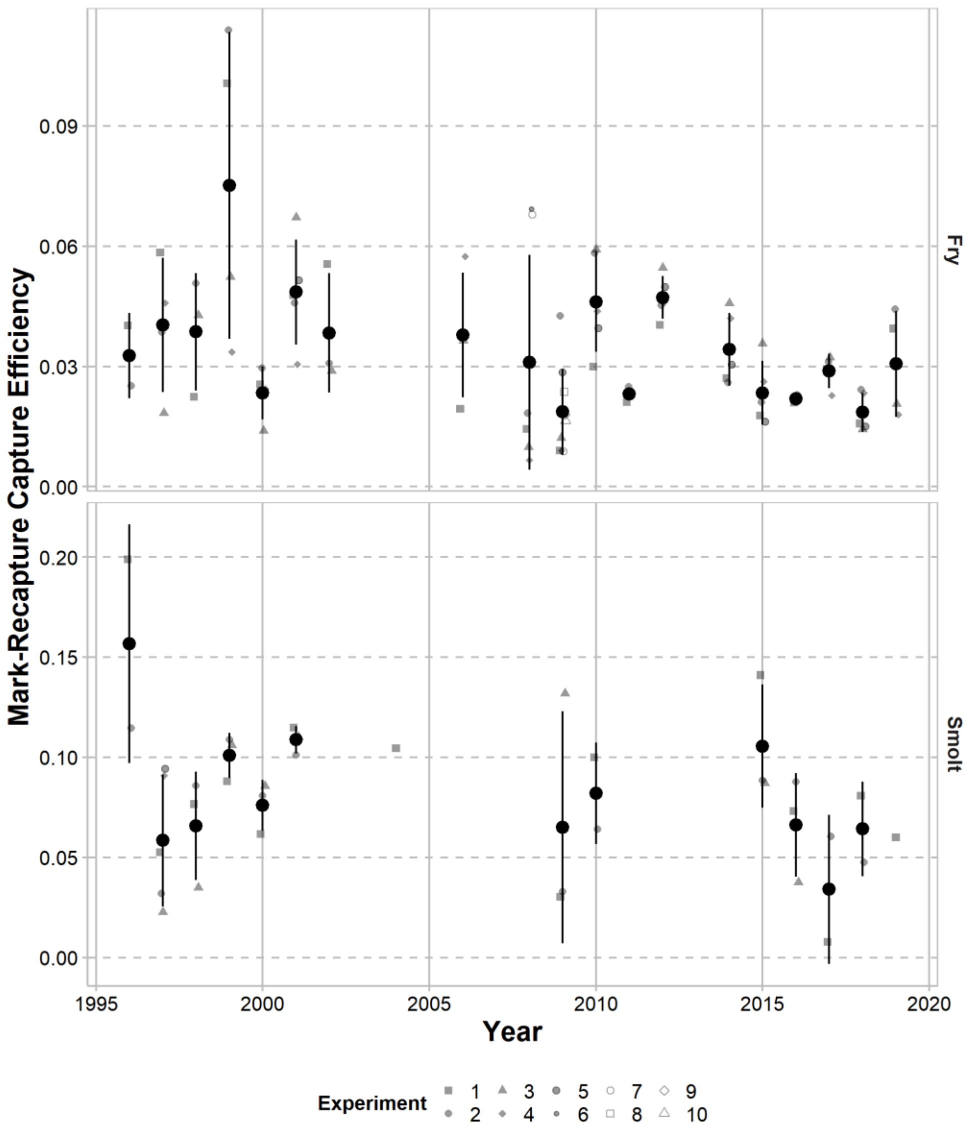


Table 1. Annual life-stage-specific mean capture efficiencies of the Quinsam Hatchery salmon counting fence, standard deviation, and number of mark-recapture experiments.

Year	Fry			Smolt		
	n[1]	Capture Efficiency		n	Capture Efficiency	
		Mean	SD		Mean	SD
1996	2	0.033	0.011	2	0.157	0.060
1997	4	0.040	0.017	5	0.058	0.033
1998	3	0.039	0.015	3	0.066	0.027
1999	4	0.075	0.038	3	0.101	0.011
2000	4	0.023	0.007	3	0.076	0.013
2001	5	0.049	0.013	3	0.109	0.007
2002	3	0.038	0.015	0	n/a	n/a
2003	0	n/a	n/a	0	n/a	n/a
2004	0	n/a	n/a	1	0.105	n/a
2005	0	n/a	n/a	0	n/a	n/a
2006	4	0.038	0.016	0	n/a	n/a
2007	0	n/a	n/a	0	n/a	n/a
2008	7	0.031	0.027	0	n/a	n/a
2009	10	0.019	0.011	3	0.065	0.058
2010	5	0.046	0.013	2	0.082	0.025
2011	4	0.023	0.002	0	n/a	n/a
2012	5	0.047	0.005	0	n/a	n/a
2013	0	n/a	n/a	0	n/a	n/a
2014	5	0.034	0.009	0	n/a	n/a
2015	5	0.023	0.008	3	0.105	0.031
2016	2	0.022	0.001	3	0.066	0.026
2017	4	0.029	0.004	2	0.034	0.037
2018	5	0.019	0.005	2	0.064	0.024
2019	4	0.031	0.013	1	0.060	n/a

[1] n, number of mark-recapture experiments

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Appendix B. Water Quality Guidelines, Typical Parameter Values, Previous Results, and Quality Control Results Summary

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1. WATER QUALITY GUIDELINES AND TYPICAL PARAMETER VALUES

Table 1. Water quality guidelines for the protection of aquatic life in British Columbia for conductivity, pH, alkalinity, and nutrients.

Parameter	Unit	BC Guideline for the Protection of Aquatic Life[1]	Guideline Reference
Specific Conductivity	µS/cm	No provincial or federal guidelines	n/a
pH	pH units	When baseline values are between 6.5 and 9 there is no restriction on changes within this range (lethal effects observed below 4.5 and above 9.5)	MOE (1991)
Alkalinity	mg/L	No provincial or federal guidelines. However, waterbodies with <10 mg/L are highly sensitive to acidic inputs, 10 to 20 mg/L are moderately sensitive to acidic inputs, > 20 mg/L have a low sensitivity to acidic inputs	n/a
Total Ammonia (N)	µg/L	Dependent on pH and temperature, too numerous to present, lowest maximum allowable concentration of 680 µg/L occurs at a pH of 9 and water temperature of 8°C, lowest maximum average 30 day concentration of 102 µg/L occurs at a pH of 9 and water temperature of 20°C	Nordin and Pommen (2009)
Nitrite (N)	µg/L	The lowest maximum allowable concentration occurs when chloride is ≤ 2 mg/L; instantaneous maximum allowable concentration is 60 µg/L and a maximum 30 day average of 20 µg/L is allowed when chloride is ≤ 2 mg/L	Nordin and Pommen (2009)
Nitrate (N)	µg/L	The 30 day average concentration to protect freshwater aquatic life is 3,000 µg/L[2] and the maximum concentration is 32.8 mg/L	Nordin and Pommen (2009)
Orthophosphate	µg/L	No provincial or federal guidelines	n/a
Total Phosphorus (P)	µg/L	Trigger ranges that would signify a change in the trophic classification: <4 µg/L: ultra-oligotrophic, 4-10 µg/L: oligotrophic, 10-20 µg/L: mesotrophic, 20-35 µg/L: meso-eutrophic, 35-100 µg/L: eutrophic, > 100 µg/L: hyper-eutrophic	CCME (2004)

[1] Guideline for total phosphorus is a federal guideline; provincial guidelines do not exist

[2] The 30-d average (chronic) concentration is based on 5 weekly samples collected within a 30-day period

Table 2. Total suspended sediments and turbidity guidelines for the protection of aquatic life in British Columbia.

Period	British Columbia[1] Suspended Sediment and Turbidity Guidelines for the Protection of Aquatic Life	
	Total Suspended Sediments (mg/L)	Turbidity (NTU)
Clear Flow Period (less than 25 mg/L or less than 8 NTU)	"Induced suspended sediment concentrations should not exceed background levels by more than 25 mg/L during any 24-hour period (hourly sampling preferred). For sediment inputs that last between 24 hours and 30 days (daily sampling preferred), the average suspended sediment concentration should not exceed background by more than 5 mg/L."	"Induced turbidity should not exceed background levels by more than 8 NTU during any 24-hour period (hourly sampling preferred). For sediment inputs that last between 24 hours and 30 days (daily sampling preferred) the mean turbidity should not exceed background by more than 2 NTU."
Turbid Flow Period (greater than or equal to 25 mg/L or greater than or equal to 8 NTU)	"Induced suspended sediment concentrations should not exceed background levels by more than 10 mg/L at any time when background levels are between 25 and 100 mg/L. When background exceeds 100 mg/L, suspended sediments should not be increased by more than 10% of the measured background level at any one time."	"Induced turbidity should not exceed background levels by more than 5 NTU at any time when background turbidity is between 8 and 50 NTU. When background exceeds 50 NTU, turbidity should not be increased by more than 10% of the measured background level at any one time."

[1] reproduced from Singleton (2001)

Table 3. Dissolved oxygen guidelines for the protection of aquatic life in British Columbia.

BC Guidelines for the Protection of Aquatic Life (MOE 2019)			
	Life Stages Other Than Buried Embryo/Alevin	**Buried Embryo/Alevin[1]**	**Buried Embryo/Alevin[1]**
Dissolved Oxygen Concentration	Water column mg/L O_2	Water column mg/L O_2	Interstitial Water mg/L O_2
Instantaneous minimum[2]	5	9	6
30-day mean[3]	8	11	8

[1] For the buried embryo / alevin life stages these are in-stream concentrations from spawning to the point of yolk sac absorption or 30 days post-hatch for fish; the water column concentrations recommended to achieve interstitial dissolved oxygen values when the latter are unavailable. Interstitial oxygen measurements would supersede water column measurements in comparing to criteria.

[2] The instantaneous minimum level is to be maintained at all times.

[3] The mean is based on at least five approximately evenly spaced samples. If a diurnal cycle exists in the water body, measurements should be taken when oxygen levels are lowest (usually early morning).

Table 4. Total gas pressure guidelines for the protection of aquatic life in British Columbia.

Water Depth	**Water Use**	**Maximum Allowable ΔP (Excess Gas Pressure) for the Protection of Aquatic Life in BC[1]**
> 1 m	Freshwater	76 mm Hg regardless of pO_2 levels
< 1 m	Shallow Water/Hatchery Environments	24 mm Hg is the most conservative form (assuming water column depth = 0 m)[2]
All depths	Background Levels Higher than BC WQG	No increase in ΔP or %TGP

[1] Adapted from Fidler and Miller (1994) and BC WQG Summary Report (MOE 2019).

[2] Derived from equation: $\Delta P_{\text{initiation of swim bladder overinflation}} = 73.89 * \text{water depth (m)} + 0.15 * pO_2$, where pO_2 = 157 mm Hg (i.e., sea level, normoxic condition) (Fidler and Miller 1994).

Table 5. Typical values for water quality parameters in British Columbia waters.

Parameter	Unit	Typical range in BC streams and rivers	Reference
Specific Conductivity	µS/cm	The typical value in coastal British Columbia streams is 100 µS/cm	RISC (1998)
pH	pH units	Natural fresh waters have a pH range from 4 to 10, lakes tend to have a pH ≥ 7.0 and coastal streams commonly have pH values of 5.5 to 6.5	RISC (1998)
Alkalinity	mg/L	Natural waters almost always have concentrations less than 500 mg/L, with waters in coastal BC typically ranging from 0 to 10 mg/L; waters in interior BC can have values greater than 100 mg/L	RISC (1998)
Total Suspended Solids	mg/L	In British Columbia natural concentrations of suspended solids vary extensively from waterbody to waterbody and can have large variation within a day and among seasons	Singleton (1985) in Caux *et al.* (1997)
Turbidity	NTU	In British Columbia natural concentrations of suspended solids vary extensively from waterbody to waterbody and can have large variation within a day and among seasons	Singleton (1985) in Caux *et al.* (1997)
Dissolved Oxygen	mg/L	In BC surface waters are generally well aerated and have DO concentrations > 10 mg/L	MOE (1997)
Dissolved Oxygen	% saturation	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)
ΔP (Total Gas Pressure - Barometric Pressure)	mm Hg	In British Columbia, dissolved gas supersaturation is a natural feature of many waters with ΔP commonly being between 50 – 80 mm Hg. (We often see values between -10 and 60)	Fidler and Miller (1994)
Total Ammonia (N)	µg/L	<100 µg/L for waters not affected by waste discharges	Nordin and Pommen (2009)
Nitrite (N)	µg/L	Due to its unstable nature, nitrite concentrations are very low, typically present in surface waters at concentrations of <1 µg/L	RISC (1998)
Nitrate (N)	µg/L	In oligotrophic 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)
Orthophosphate (P)	µg/L	Coastal BC streams have concentrations <1 µg/L	Slaney and Ward (1993); Ashley and Slaney (1997)
Total Phosphorus (P)	µg/L	Oligotrophic 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.	CCME (2004)

2. 2014 TO 2019 WATER QUALITY IN THE QUINSAM RIVER

Table 6. Quinsam River (QUN-WQ) general water quality variables measured *in situ* during Years 1 to 6 (2014 to 2019).

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
2014	23-May	-	-	-	-	73.3	73.3	73.3	0.0	95.6	95.6	95.6	0.0	12.8	12.8	12.8	0.0	7.38	7.38	7.39	0.01
	18-Jun	14	14	14	0	121.5	121.5	121.6	0.1	143.1	143.1	143.1	0.0	17.1	17.1	17.1	0.0	7.58	7.57	7.58	0.01
	22-Jul	16	16	16	0	127.5	127.5	127.5	0.0	148.1	148.1	148.1	0.0	17.7	17.7	17.7	0.0	7.36	7.36	7.36	0.00
	19-Aug	19	19	19	0	138.2	138.1	138.3	0.1	152.3	152.2	152.4	0.1	20.2	20.2	20.2	0.0	7.38	7.36	7.43	0.04
	24-Sep	14	14	14	0	91.2	91.2	91.3	0.1	109.9	109.9	109.9	0.0	16.1	16.1	16.1	0.0	7.30	7.23	7.36	0.07
	04-Nov	7	7	7	0	48.9	48.9	48.9	0.0	69.4	69.4	69.4	0.0	9.6	9.6	9.6	0.0	7.01	7.01	7.02	0.01
2015	12-May	14	14	14	0	114.6	114.6	114.6	0.0	144.4	144.4	144.5	0.1	14.2	14.2	14.2	0.0	7.68	7.68	7.68	0.00
	17-Jun	15	15	15	0	121.9	121.9	121.9	0.0	98.1	14.0	140.2	72.8	18.2	18.2	18.2	0.0	7.71	7.71	7.71	0.00
	23-Jul	17	17	17	0	161.6	161.6	161.7	0.1	190.7	190.7	190.7	0.0	17.0	17.0	17.0	0.0	7.49	7.49	7.49	0.00
	13-Aug	17	17	17	0	173.2	173.1	173.2	0.1	197.7	197.6	197.7	0.1	18.5	18.5	18.5	0.0	7.41	7.40	7.41	0.01
	16-Sep	12	12	12	0	147.1	147.1	147.1	0.0	185.7	185.7	185.7	0.0	14.1	14.1	14.1	0.0	7.50	7.50	7.50	0.00
	14-Oct	11	11	11	0	92.9	92.9	92.9	0.0	131.9	131.8	131.9	0.1	9.5	9.5	9.6	0.1	7.52	7.50	7.54	0.02
2016	18-May	12	12	12	0	119.1	119.1	119.2	0.1	150.1	150.0	150.2	0.1	14.7	14.7	14.7	0.0	7.18	7.16	7.20	0.02
	15-Jun	9	9	9	0	112.1	112.0	112.1	0.1	143.5	143.4	143.6	0.1	14.0	14.0	14.0	0.0	6.86	6.86	6.87	0.01
	13-Jul	15	15	15	0	125.5	125.4	125.6	0.1	154.2	154.1	154.4	0.2	15.7	15.7	15.7	0.0	10.52	10.46	10.59	0.07
	17-Aug	19	19	19	0	139.4	139.4	139.4	0.0	157.4	157.4	157.4	0.0	19.3	19.3	19.3	0.0	7.25	7.24	7.25	0.01
	14-Sep	12	12	12	0	138.5	138.5	138.5	0.0	172.6	172.6	172.7	0.1	15.1	15.1	15.1	0.0	7.40	7.39	7.40	0.01
	12-Oct	5	5	5	0	115.2	114.9	115.5	0.3	175.9	175.5	176.1	0.3	7.7	7.7	7.7	0.0	15.86	15.86	15.86	0.00
2017	10-May	7	7	7	0	73.3	73.3	73.3	0.0	105.7	105.7	105.8	0.1	8.9	8.9	8.9	0.0	7.58	7.58	7.58	0.00
	14-Jun	9	9	9	0	99.3	99.3	99.3	0.0	124.1	124.1	124.1	0.0	15.0	15.0	15.0	0.0	7.47	7.46	7.47	0.01
	12-Jul	17	17	17	0	140.4	140.4	140.4	0.0	158.2	158.2	158.2	0.0	19.4	19.4	19.4	0.0	7.08	7.05	7.10	0.03
	09-Aug	13	13	13	0	149.8	149.8	149.8	0.0	162.7	162.6	162.7	0.1	21.1	21.1	21.1	0.0	7.17	7.17	7.17	0.00
	13-Sep	8	8	8	0	137.6	137.6	137.6	0.0	166.8	166.8	166.9	0.1	16.2	16.2	16.2	0.0	7.21	7.20	7.22	0.01
	11-Oct	2	2	2	0	128.9	128.8	128.9	0.1	178.0	178.0	178.1	0.1	11.2	11.2	11.2	0.0	7.21	7.17	7.24	0.04
2018	10-May	9	9	9	0	66.7	66.6	66.8	0.1	95.9	95.8	96.0	0.1	9.7	9.7	9.7	0.0	6.02	5.92	6.11	0.10
	05-Jun	8	8	8	0	118.5	118.5	118.5	0.0	153.4	153.3	153.4	0.1	13.6	13.6	13.6	0.0	6.58	6.57	6.58	0.01
	04-Jul	12	12	12	0	116.1	116.1	116.1	0.0	139.0	139.0	139.0	0.0	16.8	16.8	16.8	0.0	7.59	7.59	7.59	0.00
	09-Aug	14	14	14	0	129.9	129.8	129.9	0.1	137.4	137.3	137.4	0.1	22.1	22.1	22.1	0.0	7.05	7.04	7.06	0.01
	12-Sep	10	10	10	0	91.0	91.0	91.0	0.0	112.8	112.8	112.8	0.0	15.3	15.3	15.3	0.0	7.69	7.69	7.70	0.01
	05-Oct	5	5	5	0	79.3	79.3	79.4	0.1	112.5	112.4	112.6	0.1	9.5	9.5	9.5	0.0	-	-	-	-

[1] Average of three replicates (n=3) on each date unless otherwise indicated. A single data listed under Avg. indicates n=1.

Dashes (-) mean that no data were collected.

Table 6. Continued (2 of 2).

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
2019	13-May	8.0	8.0	8.0	0.0	84.4	84.4	84.4	0.0	115.3	115.2	115.3	0.1	11.0	11.0	11.0	0.0	6.50	6.50	6.50	0.00
	12-Jun	18.5	18.5	18.5	0.0	128.0	128.0	128.0	0.0	146.6	146.5	146.6	0.1	18.4	18.4	18.4	0.0	7.60	7.59	7.60	0.01
	11-Jul	15.0	15.0	15.0	0.0	98.8	98.8	98.9	0.1	113.8	113.8	113.8	0.0	18.4	18.4	18.4	0.0	7.15	7.13	7.17	0.02
	12-Aug	14.0	14.0	14.0	0.0	82.8	82.8	82.8	0.0	94.6	94.6	94.6	0.0	18.8	18.8	18.8	0.0	7.42	7.41	7.42	0.01
	12-Sep	13.0	13.0	13.0	0.0	66.3	66.3	66.3	0.0	78.2	78.2	78.2	0.0	17.0	17.0	17.0	0.0	7.56	7.55	7.56	0.01
	09-Oct	5.0	5.0	5.0	0.0	91.8	91.7	91.8	0.1	135.7	135.7	135.7	0.0	8.1	8.1	8.1	0.0	7.33	7.33	7.33	0.00

[1] Average of three replicates (n=3) on each date unless otherwise indicated. A single data listed under Avg. indicates n=1.

Dashes (-) mean that no data were collected.

Table 7. Quinsam River (QUN-WQ) dissolved gases measured *in situ* during Years 1 to 6 (2014 to 2019).

Year	Date	Barometric Pressure mm Hg				Oxygen Dissolved %				Oxygen Dissolved mg/L				TGP %				TGP mm Hg				ΔP mm Hg			
		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
2014	23-May	744	743	745	1	101.8	101.4	102.6	0.7	10.74	10.69	10.82	0.07	100	100	100	0	744	744	745	1	0	0	1	1
	18-Jun	748	748	749	1	91.3	90.9	91.9	0.5	8.84	8.80	8.87	0.04	101	101	101	0	755	753	757	2	7	5	8	2
	22-Jul	747	747	748	1	95.8	95.8	95.9	0.1	9.13	9.12	9.13	0.01	101	101	101	0	753	753	753	0	6	5	6	1
	19-Aug	745	744	745	1	77.9	77.7	78.3	0.3	7.01	6.99	7.03	0.02	99	99	99	0	735	735	735	0	-10	-10	-9	1
	24-Sep	753	752	753	1	91.7	90.1	92.7	1.4	8.78	8.53	8.91	0.21	98	98	98	0	739	739	740	1	-13	-14	-13	1
	04-Nov	761	761	762	1	88.5	88.4	88.5	0.1	9.95	9.94	9.96	0.01	99	99	99	0	755	755	755	0	-6	-7	-6	1
2015	12-May	741	741	741	0	96.2	96.2	96.3	0.1	9.89	9.88	9.89	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	17-Jun	-	-	-	-	83.7	83.6	83.9	0.2	7.90	7.89	7.91	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	23-Jul	744	744	744	0	84.2	84.1	84.4	0.2	8.14	8.13	8.14	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	13-Aug	746	746	746	0	84.2	84.1	84.4	0.2	7.89	7.88	7.91	0.02	-	-	-	-	-	-	-	-	-	-	-	-
	16-Sep	743	743	743	0	78.1	77.8	78.5	0.4	8.03	8.00	8.05	0.03	-	-	-	-	-	-	-	-	-	-	-	-
	14-Oct	754	754	754	0	87.0	86.8	87.3	0.3	9.88	9.87	9.89	0.01	-	-	-	-	-	-	-	-	-	-	-	-
2016	18-May	747	747	747	0	81.9	81.7	82.0	0.2	8.30	8.30	8.30	0.00	-	-	-	-	-	-	-	-	-	-	-	-
	15-Jun	744	744	744	0	80.0	79.9	80.2	0.2	8.23	8.22	8.24	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	13-Jul	757	757	757	0	79.4	79.3	79.5	0.1	7.89	7.87	7.92	0.03	-	-	-	-	-	-	-	-	-	-	-	-
	17-Aug	749	749	749	0	84.4	84.1	84.6	0.3	7.77	7.75	7.79	0.02	-	-	-	-	-	-	-	-	-	-	-	-
	14-Sep	747	747	747	0	81.0	80.9	81.2	0.2	8.16	8.15	8.17	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	12-Oct	747	747	747	0	98.0	97.6	98.5	0.5	11.70	11.63	11.75	0.06	-	-	-	-	-	-	-	-	-	-	-	-

[1] Average of three replicates (n=3) on each date unless otherwise indicated. A single data listed under Avg. indicates n=1.

Blue shading indicates that the more conservative provincal guideline (DO instantaneous minium of 9 mg/L) for the protection of buried embryo/alevin has not been achieved. Note that the guideline for life stages other than buried embryo/alevin is met (DO instantaneous minimum of 5 mg/L).

Dashes (-) mean that no data were collected.

Table 7 Continued (2 of 2).

Year	Date	Barometric Pressure mm Hg				Oxygen Dissolved %				Oxygen Dissolved mg/L				TGP %				TGP mm Hg				ΔP mm Hg			
		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
2017	10-May	742	742	742	0	76.9	76.6	77.3	0.4	8.94	8.92	8.96	0.02	-	-	-	-	-	-	-	-	-	-	-	-
	14-Jun	752	752	752	0	89.6	89.5	89.7	0.1	9.03	9.01	9.05	0.02	-	-	-	-	-	-	-	-	-	-	-	-
	12-Jul	749	749	749	0	87.1	87.0	87.1	0.1	8.02	8.01	8.03	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	09-Aug	748	748	748	0	80.0	79.5	80.3	0.5	7.13	7.13	7.13	0.00	-	-	-	-	-	-	-	-	-	-	-	-
	13-Sep	749	749	749	0	83.7	83.5	83.8	0.2	8.21	8.20	8.22	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	11-Oct	751	751	751	0	91.6	91.6	91.7	0.1	10.05	10.04	10.06	0.01	-	-	-	-	-	-	-	-	-	-	-	-
2018	10-May	748	748	748	0	96.5	95.8	97.0	0.6	10.99	10.97	11.02	0.03	-	-	-	-	-	-	-	-	-	-	-	-
	05-Jun	744	743	744	0	85.3	85.2	85.4	0.1	8.86	8.85	8.87	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	04-Jul	753	753	753	0	82.4	82.2	82.6	0.2	7.99	7.97	8.02	0.03	-	-	-	-	-	-	-	-	-	-	-	-
	09-Aug	-	-	-	-	90.7	90.0	91.9	1.0	8.25	7.85	8.87	0.55	-	-	-	-	-	-	-	-	-	-	-	-
	12-Sep	744	744	744	0	93.8	92.1	95.7	1.8	9.41	9.24	9.62	0.19	-	-	-	-	-	-	-	-	-	-	-	-
	05-Oct	-	-	-	-	84.8	84.1	85.9	1.0	9.75	9.65	9.80	0.08	-	-	-	-	-	-	-	-	-	-	-	-
2019	13-May	-	-	-	-	90.4	90.4	90.4	0.0	9.95	9.95	9.96	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	12-Jun	745.9	745.8	745.9	0.1	90.9	90.9	91.0	0.1	8.54	8.53	8.55	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	11-Jul	754.1	754.1	754.2	0.1	89.8	89.4	90.0	0.3	8.43	8.40	8.45	0.03	-	-	-	-	-	-	-	-	-	-	-	-
	12-Aug	754.2	754.2	754.3	0.1	91.9	91.8	92.0	0.1	8.58	8.57	8.59	0.01	-	-	-	-	-	-	-	-	-	-	-	-
	12-Sep	753.0	753.0	753.0	0.0	89.4	89.1	89.7	0.3	8.63	8.62	8.65	0.02	-	-	-	-	-	-	-	-	-	-	-	-
	09-Oct	755.8	755.8	755.8	0.0	98.4	98.3	98.5	0.1	11.64	11.64	11.65	0.01	-	-	-	-	-	-	-	-	-	-	-	-

[1] Average of three replicates (n=3) on each date unless otherwise indicated. A single data listed under Avg. indicates n=1.

Blue shading indicates that the more conservative provincal guideline (DO instantaneous minium of 9 mg/L) for the protection of buried embryo/alevin has not been achieved. Note that the guideline for life stages other than buried embryo/alevin is met (DO instantaneous minimum of 5 mg/L).

Dashes (-) mean that no data were collected.

Table 8. Quinsam River (QUN-WQ) general water quality variables measured at ALS laboratories during Years 1 to 6 (2014 to 2019).

Year	Date	Alkalinity, Total (as CaCO3) mg/L				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
2014	23-May	31.7	31.5	31.8	0.2	94.8	94.1	95.4	0.9	69	68	70	1	<1	<1	<1	0.0	0.59	0.52	0.65	0.09	7.77	7.77	7.77	0.00
	18-Jun	41.0	40.8	41.1	0.2	139.5	139.0	140.0	0.7	96	96	96	0	<1	<1	<1	0.0	0.42	0.40	0.44	0.03	7.87	7.87	7.87	0.00
	22-Jul	42.4	42.4	42.4	0.0	140.0	139.0	141.0	1.4	103	101	105	3	<1	<1	<1	0.0	0.46	0.44	0.47	0.02	7.73	7.65	7.81	0.11
	19-Aug	42.1	41.9	42.3	0.3	156.0	146.0	166.0	14.1	96	95	96	1	<1	<1	<1	0.0	0.70	0.47	0.93	0.33	7.81	7.57	8.05	0.34
	24-Sep	35.0	35.0	35.0	0.0	109.0	109.0	109.0	0.0	71	67	74	5	<1	<1	<1	0.0	0.56	0.50	0.62	0.08	7.55	7.52	7.58	0.04
	04-Nov	23.7	23.5	23.8	0.2	71.3	70.7	71.8	0.8	59	53	64	8	<1	<1	<1	0.0	0.74	0.71	0.77	0.04	7.61	7.59	7.63	0.03
2015	12-May	40.8	40.6	41.0	0.3	143.0	143.0	143.0	0.0	91	89	93	3	<1	<1	<1	0.0	0.38	0.37	0.39	0.01	7.79	7.78	7.80	0.01
	17-Jun	43.9	43.8	43.9	0.1	157.0	157.0	157.0	0.0	97	94	100	4	<1	<1	<1	0.0	0.41	0.40	0.42	0.01	7.91	7.90	7.92	0.01
	23-Jul	52.9	51.7	54.0	1.6	206.0	206.0	206.0	0.0	120	120	120	0	<1	<1	<1	0.0	0.49	0.49	0.49	0.00	8.00	7.99	8.01	0.01
	13-Aug	48.8	48.0	49.6	1.1	175.0	173.0	177.0	2.8	124	120	127	5	<1	<1	<1	0.0	0.36	0.30	0.42	0.08	7.78	7.70	7.85	0.11
	16-Sep	46.2	46.0	46.3	0.2	178.0	177.0	179.0	1.4	145	116	173	40	<1	<1	<1	0.0	0.40	0.38	0.42	0.03	7.94	7.94	7.94	0.00
	14-Oct	34.0	33.9	34.1	0.1	130.0	129.0	131.0	1.4	94	92	96	3	<1	<1	1.6	0.4	0.47	0.40	0.53	0.09	7.55	7.52	7.58	0.04
2016	18-May	35.4	35.1	35.6	0.4	131.5	131.0	132.0	0.7	85	85	85	0	<1	<1	<1	0.0	0.49	0.38	0.59	0.15	7.83	7.80	7.86	0.04
	15-Jun	34.3	33.9	34.7	0.6	130.5	130.0	131.0	0.7	87	86	88	1	<1	<1	<1	0.0	0.45	0.44	0.46	0.01	7.78	7.77	7.78	0.01
	13-Jul	36.6	36.5	36.7	0.1	110.0	109.0	111.0	1.4	70	67	72	4	<1	<1	1.5	0.4	1.17	1.14	1.19	0.04	7.68	7.67	7.68	0.01
	17-Aug	35.5	35.4	35.5	0.1	137.5	137.0	138.0	0.7	87	86	88	1	<1	<1	1.1	0.1	0.46	0.44	0.47	0.02	7.51	7.50	7.51	0.01
	14-Sep	35.3	35.1	35.4	0.2	139.0	139.0	139.0	0.0	84	83	84	1	<1	<1	<1	0.0	0.46	0.45	0.46	0.01	7.71	7.70	7.72	0.01
	12-Oct	30.6	30.4	30.8	0.3	118.5	114.0	123.0	6.4	83	81	84	2	<1	<1	<1	0.0	0.72	0.72	0.72	0.00	7.70	7.69	7.71	0.01
2017	10-May	32.4	32.2	32.6	0.3	105.5	104.0	107.0	2.1	90	72	107	25	2.1	1.7	2.4	0.5	0.59	0.55	0.62	0.05	7.71	7.69	7.72	0.02
	14-Jun	41.1	41.1	41.1	0.0	145.5	145.0	146.0	0.7	99	95	102	5	<1	<1	<1	0.0	0.54	0.53	0.54	0.01	7.94	7.93	7.94	0.01
	12-Jul	44.3	43.5	45.0	1.1	148.0	147.0	149.0	1.4	93	92	94	1	1.4	1.3	1.4	0.1	0.57	0.53	0.61	0.06	7.91	7.89	7.93	0.03
	09-Aug	43.8	43.7	43.9	0.1	161.0	160.0	162.0	1.4	102	101	103	1	<1	<1	<1	0.0	0.61	0.54	0.68	0.10	7.80	7.79	7.80	0.01
	13-Sep	43.2	42.7	43.7	0.7	162.0	162.0	162.0	0.0	103	98	107	6	<1	<1	<1	0.0	0.46	0.44	0.47	0.02	7.91	7.91	7.91	0.00
	11-Oct	45.4	45.1	45.6	0.4	169.0	169.0	169.0	0.0	127	125	128	2	<1	<1	<1	0.0	0.41	0.41	0.41	0.00	7.63	7.62	7.63	0.01
2018	10-May	27.8	27.8	27.8	0.0	93.2	92.7	93.6	0.6	70	69	70	1	<1	<1	<1	0.0	0.46	0.43	0.48	0.04	7.59	7.57	7.60	0.02
	05-Jun	41.3	40.9	41.7	0.6	149.5	149.0	150.0	0.7	97	96	98	1	1.4	1.1	1.6	0.4	0.48	0.45	0.50	0.04	7.85	7.84	7.85	0.01
	04-Jul	38.7	38.4	39.0	0.4	132.5	132.0	133.0	0.7	93	87	98	8	1.4	1.3	1.5	0.1	0.58	0.54	0.62	0.06	7.78	7.76	7.79	0.02
	09-Aug	41.2	41.1	41.2	0.1	132.0	132.0	132.0	0.0	88	88	88	0	<1	<1	1.1	0.1	0.64	0.52	0.75	0.16	7.84	7.84	7.84	0.00
	12-Sep	37.0	36.8	37.1	0.2	110.0	110.0	110.0	0.0	78	73	82	6	<3	<3	<3	0.0	0.38	0.32	0.43	0.08	7.81	7.80	7.82	0.01
	05-Oct	31.0	30.9	31.0	0.1	105.5	104.0	107.0	2.1	78	77	78	1	<1	<1	1.3	0.2	0.44	0.43	0.44	0.01	7.65	7.61	7.68	0.05
2019	13-May	35.8	35.6	36.0	0.3	119.5	118.0	121.0	2.1	71	66	76	7	<1.2	<1.0	1.3	0.2	0.35	0.33	0.36	0.02	7.65	7.65	7.65	0.00
	12-Jun	41.5	41.5	41.5	0.0	142.0	142.0	142.0	0.0	96	94	97	2	<1.1	<1.0	1.2	0.1	0.36	0.34	0.38	0.03	7.88	7.87	7.88	0.01
	11-Jul	35.0	35.0	35.0	0.0	103.0	103.0	103.0	0.0	75	74	76	1	1.4	1.2	1.5	0.2	0.48	0.45	0.51	0.04	7.75	7.73	7.76	0.02
	12-Aug	33.7	33.4	34.0	0.4	83.6	83.4	83.8	0.3	56	56	56	0	<3	<3	<3	0.0	0.43	0.41	0.45	0.03	7.64	7.62	7.66	0.03
	12-Sep	31.2	31.1	31.2	0.1	78.2	77.9	78.5	0.4	62	61	62	1	<3	<3	<3	0.0	0.41	0.33	0.49	0.11	7.57	7.56	7.58	0.01
	09-Oct	39.2	39.1	39.3	0.1	132.0	132.0	132.0	0.0	79	77	81	3	<1	<1	<1	0.0	0.32	0.29	0.34	0.04	7.73	7.73	7.73	0.00

[1] Average of two replicates (n=2) on each date unless otherwise indicated. A single data listed under Avg. indicates n=1.

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

Table 9. Quinsam River (QUN-WQ) low level nutrients measured at ALS laboratories during Years 1 to 6 (2014 to 2019).

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
2014	23-May	<5	<5	<5	0	<1	<1	<1	0	14	14	14	0	<1	<1	<1	0	4	4	4	0
	18-Jun	<5	<5	<5	0	<1	<1	<1	0	30	29	30	1	<1	<1	<1	0	3	3	3	0
	22-Jul	<5	<5	<5	0	<1	<1	<1	0	32	31	32	0	<1	<1	<1	0	3	3	3	0
	19-Aug	<5	<5	5	0	<1	<1	<1	0	17	17	17	0	<1	<1	<1	0	5	5	5	0
	24-Sep	<5	<5	<5	0	<1	<1	<1	0	21	21	22	1	<1	<1	<1	0	4	4	5	0
	04-Nov	5	5	5	0	<1	<1	<1	0	25	24	25	1	<1	<1	<1	0	4	3	4	1
2015	12-May	<5	<5	<5	0	<1	<1	<1	0	23	23	23	0	<1	<1	<1	0	3	3	3	1
	17-Jun	<5	<5	<5	0	<1	<1	<1	0	24	24	24	0	<1	<1	<1	0	<2	<2	<2	0
	23-Jul	<5	<5	<5	0	<1	<1	<1	0	30	29	31	1	<1	<1	<1	0	<2	<2	2	0
	13-Aug	<5	<5	<5	0	<1	<1	<1	0	41	41	41	0	<1	<1	<1	0	<2	<2	<2	0
	16-Sep	<5	<5	<5	0	<1	<1	<1	0	14	14	14	0	<1	<1	<1	0	<2	<2	2	0
	14-Oct	9	9	9	0	<1	<1	<1	0	36	36	36	0	<1	<1	<1	0	5	4	5	0
2016	18-May	<5	<5	<5	0	<1	<1	<1	0	16	16	16	0	<1	<1	<1	0	3	3	4	1
	15-Jun	<5	<5	<5	0	1	1	2	0	15	14	16	1	<1	<1	<1	0	3	3	4	1
	13-Jul	<5	<5	<5	0	<1	<1	<1	0	17	16	17	1	<1	<1	<1	0	5	4	5	0
	17-Aug	<5	<5	<5	0	<1	<1	<1	0	24	24	24	0	<1	<1	<1	0	4	3	5	1
	14-Sep	<5	<5	<5	0	<1	<1	<1	0	18	18	19	0	<1	<1	<1	0	3	3	3	0
	12-Oct	10	9	10	0	<1	<1	<1	0	39	39	39	0	<1	<1	<1	0	5	5	6	0
2017	10-May	<5	<5	<5	0	<1	<1	<1	0	14	13	14	1	<1	<1	<1	0	<2	<2	<2	0
	14-Jun	<5	<5	<5	0	<1	<1	<1	0	18	18	18	0	<1	<1	<1	0	<2	<2	<2	0
	12-Jul	<5	<5	<5	0	<1	<1	<1	0	20	20	21	0	<1	<1	<1	0	3	2	3	1
	09-Aug	<5	<5	<5	0	<1	<1	<1	0	18	18	19	1	<1	<1	<1	0	2	2	3	0
	13-Sep	<5	<5	<5	0	<1	<1	<1	0	12	12	13	0	<1	<1	<1	0	<2	<2	2	0
	11-Oct	24	23	25	1	<1	<1	<1	0	47	47	48	1	<1	<1	<1	0	4	4	4	0
2018	10-May	<5	<5	<5	0	<1	<1	<1	0	9.6	8.5	10.6	1.5	<1	<1	<1	0	2.7	2.6	2.7	0.1
	05-Jun	<5	<5	5.4	0.3	<1	<1	<1	0	16.6	16.2	16.9	0.5	<1	<1	<1	0	3.1	2.9	3.3	0.3
	04-Jul	<5	<5	<5	0	<1	<1	<1	0	13.5	13.1	13.9	0.6	<1	<1	<1	0	5.5	4.9	6.0	0.8
	09-Aug	<5	<5	<5	0	<1	<1	<1	0	21.6	21.5	21.6	0.1	<1	<1	<1	0	3.9	3.7	4.0	0.2
	12-Sep	<5	<5	<5	0	<1	<1	<1	0	30.4	30.2	30.5	0.2	<1	<1	<1	0	3.3	3.1	3.5	0.3
	05-Oct	16.8	16.7	16.9	0.1	<1	<1	<1	0	21.6	21.3	21.8	0.4	<1	<1	<1	0	4.7	4.2	5.2	0.7

[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 9. Continued (2 of 2).

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
2019	13-May	<5	<5	<5	0	<1	<1	<1	0	10.2	10.2	10.2	0.0	<1	<1	<1	0	<2.3	<2	2.5	0.4
	12-Jun	<5.1	<5	5.2	0.1	<1	<1	<1	0	21.3	20.8	21.8	0.7	<1	<1	<1	0	3.3	3.0	3.6	0.4
	11-Jul	<5	<5	<5	0	<1	<1	<1	0	17.9	17.3	18.5	0.8	<1	<1	<1	0	4.8	4.7	4.9	0.1
	12-Aug	<11.9	<5.0	18.7	9.7	<1	<1	<1	0	19.1	18.8	19.4	0.4	<1	<1	<1	0	<2.1	<2	2.1	0.1
	12-Sep	<5	<5	<5	0	<1	<1	<1	0	14.2	14.1	14.3	0.1	<1	<1	<1	0	3.5	3.3	3.6	0.2
	09-Oct	5.7	5.5	5.8	0.2	<1	<1	<1	0	27.1	26.1	28.1	1.4	1.5	1.5	1.5	0.0	4.4	4.3	4.4	0.1

[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. QUALITY CONTROL/QUALITY ASSURANCE

Table 10. Hold time exceedances for water samples analyzed by ALS laboratories recorded during 2014 to 2019.

Description[1]	Sampling Date	Recommended Hold Time (days)	Actual Hold Time (days)
Anions and Nutrients			
Nitrite in Water by Ion Chromatography	19-Aug-14	3	8
Nitrate in Water by IC (Low Level)	10-May-18	3	5

[1] All samples for all sites and sample dates exceeded the recommended hold time for pH of 0.25 hours

Table 11. Results of field blank and trip blanks for water samples analysed by ALS laboratories, 2014 to 2019.

Year	Date	Type of Sample	Alkalinity, Total (as CaCO3) mg/L	Ammonia, Total (as N) µg/L	Conductivity µS/cm	Orthophosphate (as P) µg/L	Nitrate (as N) µg/L	Nitrite (as N) µg/L	Total Dissolved Solids mg/L	Total Phosphorus (P) µg/L	Total Suspended Solids mg/L	Turbidity NTU	pH pH units
2014	23-May	Field Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.60
		Trip Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.64
	18-Jun	Field Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.47
		Trip Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.45
	22-Jul	Field Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.69
		Trip Blank	<2.0	2.71	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.76
	19-Aug	Field Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.91
		Trip Blank	<2.0	38.7	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	6.17
	24-Sep	Field Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.45
		Trip Blank	<2.0	55.1	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.41
	04-Nov	Field Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.70
		Trip Blank	<2.0	99.5	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.75
2015	12-May	Field Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.84
		Trip Blank	<2.0	11.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.80
	17-Jun	Field Blank	<2.0	<5.0	3.2	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	6.22
		Trip Blank	<2.0	58.5	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.91
2016	18-May	Field Blank	<2.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.62
		Trip Blank	<2.0	5.90	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.58
2018	10-May	Field Blank	<1.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.53
		Trip Blank	<1.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.3
2019	13-May	Field Blank	<1.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.41
		Trip Blank	<1.0	<5.0	<2.0	<1.0	<5.0	<1.0	<10	<2.0	<1.0	<0.10	5.38

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Appendix C. Supplementary Results of Analysis of Invertebrate Drift Data

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1. QUINSAM RIVER INVERTEBRATE DRIFT RESULTS (SUMMARY TABLE)

Table 1. Quinsam River invertebrate drift mean density, biomass, Simpson's diversity index (family level), richness and Canadian Ecological Flow Index (CEFI). Each drift net was analyzed separately in 2014 for density, biomass and CEFI, while nets were combined into one sample per site for biodiversity metrics (Family richness, Simpson's diversity) and for all metrics in subsequent years.

All Taxa (Aquatic, Semi-Aquatic, and Terrestrial)													
Year	Date	# of Replicates	Density (#/m^3)[1]			Biomass (mg/m^3)[1]			CEFI Index[†1]			Simpson's Diversity Index (1-λ)[2]	Richness (# of Families)[2]
			Mean	S.D.	C.V.	Mean	S.D.	C.V.	Mean	S.D.	C.V.	Mean	Mean
2014	23-May	5	0.96	0.12	12.6	0.20	0.04	21.2	0.38	0.01	2.9	0.84	66
	04-Jun	5	2.73	0.22	8.1	0.34	0.06	17.5	0.36	0.02	4.5	0.78	66
	12-Jun	5	2.57	0.31	12.0	0.20	0.05	26.9	0.36	0.01	2.4	0.74	65
	18-Jun	5	3.11	0.65	20.9	0.16	0.06	36.8	0.36	0.01	1.6	0.76	63
	27-Jun	5	2.48	0.46	18.7	0.14	0.05	33.2	0.35	0.01	2.1	0.81	70
	22-Jul	5	4.19	0.73	17.5	0.14	0.02	14.1	0.36	0.00	0.6	0.82	60
	19-Aug	5	6.88	3.27	47.5	0.16	0.02	15.7	0.35	0.01	1.9	0.66	59
	24-Sep	5	2.36	0.85	35.9	0.09	0.03	35.6	0.32	0.01	3.4	0.81	52
	04-Nov	5	0.65	0.22	33.3	0.07	0.02	33.5	0.33	0.01	1.6	0.92	80
2015	12-May	1	1.38	-	-	0.21	-	-	0.35	-	-	0.78	52
	17-Jun	1	4.41	-	-	0.19	-	-	0.34	-	-	0.65	50
	09-Jul	1	6.38	-	-	0.32	-	-	0.34	-	-	0.74	61
	16-Jul	1	2.52	-	-	0.28	-	-	0.35	-	-	0.81	73
	23-Jul	1	4.38	-	-	0.12	-	-	0.33	-	-	0.76	53
	29-Jul	1	4.57	-	-	0.14	-	-	0.34	-	-	0.64	39
	13-Aug	1	4.34	-	-	0.08	-	-	0.31	-	-	0.78	42
	16-Sep	1	1.71	-	-	0.12	-	-	0.35	-	-	0.79	33
	14-Oct	1	2.06	-	-	0.12	-	-	0.34	-	-	0.87	50
2016	04-May	1	2.49	-	-	0.20	-	-	0.36	-	-	0.78	38
	11-May	1	1.87	-	-	0.15	-	-	0.36	-	-	0.79	43
	18-May	1	2.82	-	-	0.22	-	-	0.35	-	-	0.78	48
	25-May	1	3.72	-	-	0.25	-	-	0.34	-	-	0.82	59
	15-Jun	1	3.25	-	-	0.24	-	-	0.33	-	-	0.82	40
	13-Jul	1	5.33	-	-	0.15	-	-	0.31	-	-	0.66	41
	17-Aug	1	1.76	-	-	0.10	-	-	0.33	-	-	0.77	53
	12-Oct	1	1.71	-	-	0.13	-	-	0.35	-	-	0.92	53
2017	10-May	1	1.63	-	-	0.33	-	-	0.36	-	-	0.85	44
	14-Jun	1	4.13	-	-	0.18	-	-	0.37	-	-	0.71	28
	12-Jul	1	3.66	-	-	0.10	-	-	0.35	-	-	0.76	39
	09-Aug	1	4.84	-	-	0.25	-	-	0.34	-	-	0.75	46
	16-Aug	1	4.37	-	-	0.10	-	-	0.34	-	-	0.68	33
	23-Aug	1	3.29	-	-	0.17	-	-	0.33	-	-	0.81	40
	31-Aug	1	2.38	-	-	0.09	-	-	0.35	-	-	0.77	45
	13-Sep	1	2.46	-	-	0.10	-	-	0.34	-	-	0.80	31
	11-Oct	1	1.18	-	-	0.06	-	-	0.34	-	-	0.83	30
2018	10-May	1	1.21	-	-	0.08	-	-	0.35	-	-	0.74	32
	05-Jun	1	2.58	-	-	0.16	-	-	0.32	-	-	0.69	35
	04-Jul	1	3.97	-	-	0.17	-	-	0.34	-	-	0.78	40
	09-Aug	1	3.67	-	-	0.15	-	-	0.34	-	-	0.85	47
	04-Sep	1	1.35	-	-	0.09	-	-	0.36	-	-	0.84	46
	12-Sep	1	2.04	-	-	0.14	-	-	0.37	-	-	0.84	35
	21-Sep	1	1.94	-	-	0.13	-	-	0.33	-	-	0.91	28
	26-Sep	1	1.76	-	-	0.17	-	-	0.36	-	-	0.90	56
	05-Oct	1	1.19	-	-	0.09	-	-	0.35	-	-	0.89	47

† Calculation considers only aquatic taxa

1 Replicates were averaged where applicable prior to calculating metric

2 Net data were combined into a single sample for the site prior to calculating metric

Table 1. Continued (2 of 2).

All Taxa (Aquatic, Semi-Aquatic, and Terrestrial)													
Year	Date	# of Replicates	Density (#/m^3)[1]			Biomass (mg/m^3)[1]			CEFI Index[†1]			Simpson's Diversity Index (1-λ)[2]	Richness (# of Families)[2]
			Mean	S.D.	C.V.	Mean	S.D.	C.V.	Mean	S.D.	C.V.	Mean	Mean
2019	13-May	1	1.48	-	-	0.11	-	-	0.40	-	-	0.56	29
	06-Jun	1	1.71	-	-	0.05	-	-	0.34	-	-	0.87	49
	12-Jun	1	2.92	-	-	0.12	-	-	0.35	-	-	0.81	33
	20-Jun	1	2.61	-	-	0.11	-	-	0.34	-	-	0.86	39
	27-Jun	1	3.15	-	-	0.12	-	-	0.33	-	-	0.86	40
	11-Jul	1	3.74	-	-	0.15	-	-	0.34	-	-	0.88	36
	12-Aug	1	2.87	-	-	0.11	-	-	0.34	-	-	0.77	23
	12-Sep	1	2.27	-	-	0.08	-	-	0.34	-	-	0.79	31
	09-Oct	1	1.00	-	-	0.10	-	-	0.38	-	-	0.63	35

† Calculation considers only aquatic taxa

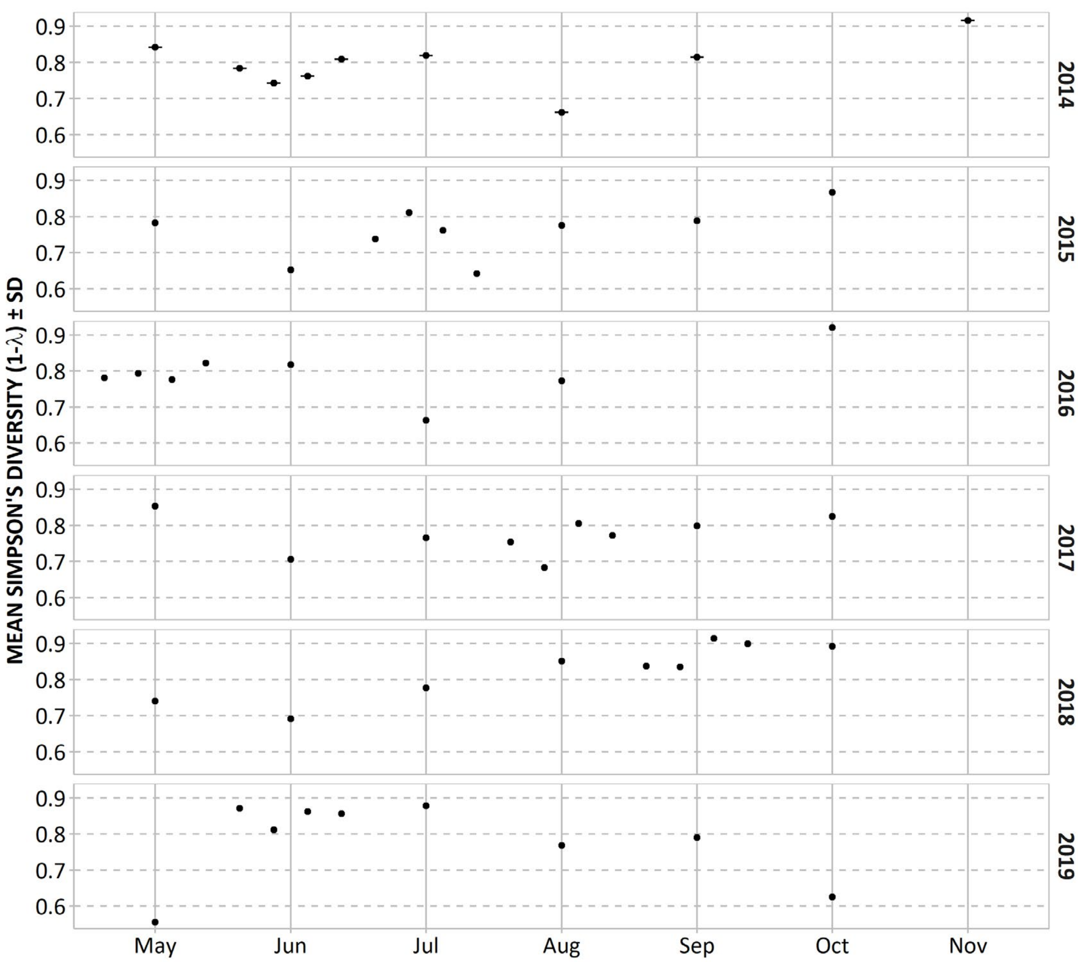
1 Replicates were averaged where applicable prior to calculating metric

2 Net data were combined into a single sample for the site prior to calculating metric

2. SIMPSON'S FAMILY LEVEL DIVERSITY (1-Λ)

Mean Simpson's family level diversity values varied throughout Year 6 with greatest values in June and July and the lowest values at the start (May) and end (October)of the monitoring period (Table 1, Figure 1). Mean Simpson's diversity ranged from a minimum of 0.56 on May 13, 2019 to a maximum of 0.88 on July 11, 2019. The values measured in May and October were lower than the range of values measured in previous years (0.64 – 0.93).

Figure 1. Drift invertebrate Simpson's Diversity (all taxa) in the Quinsam River throughout 2014 - 2019.

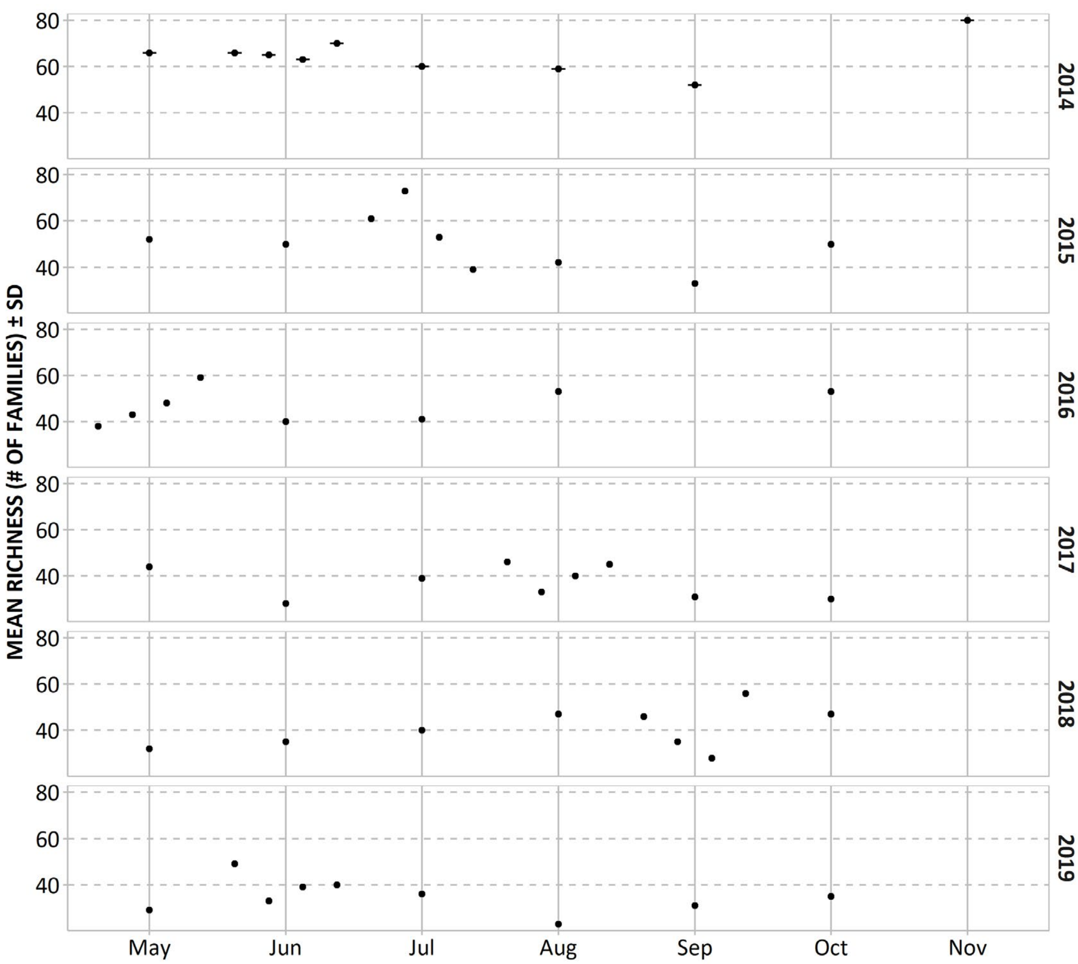


3. RICHNESS (# OF FAMILIES)

Mean family richness measured in Year 6 ranged from 23 to 49 families across sampling dates (Table 1, Figure 2). There was no clear seasonal pattern, with the lowest value observed on August 12, 2019 and the highest value observed on June 6, 2019.

On average, family richness in Year 6 was lower than previous years (Figure 2) and the value observed in August 2019 was the lowest measured to date during JHTMON-8.

Figure 2. Drift invertebrate family richness (all taxa) in the Quinsam River throughout 2014 - 2019.

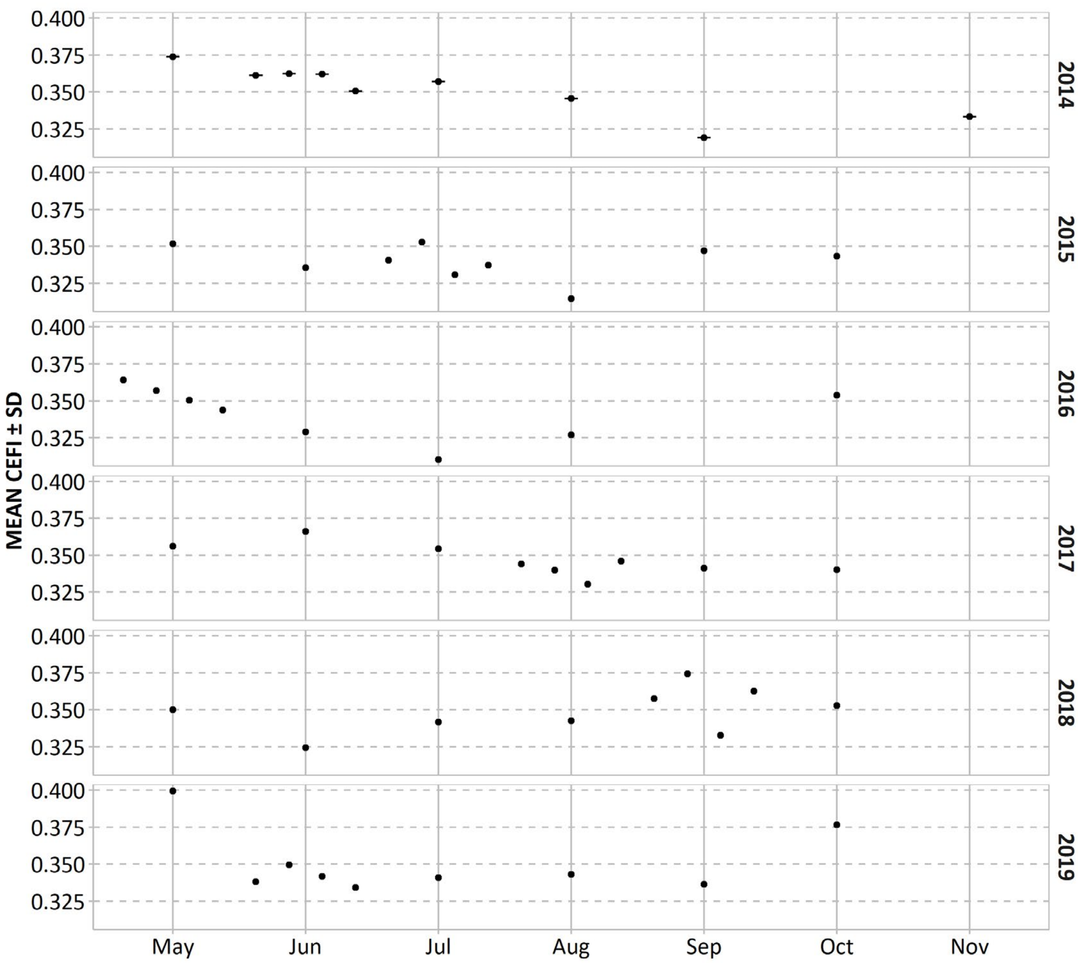


4. CANADIAN ECOLOGICAL FLOW INDEX

The CEFI results for Year 6 are in the range 0.33 – 0.40, which are above the 0.25 threshold of low CEFI values (Armanini *et al.* 2011). In Year 6, the lowest CEFI value occurred in June while the maximum value was observed in May. These results showed no clear seasonal patterns throughout Year 6.

The Year 6 results had one CEFI value greater than the range observed in previous years (0.31 - 0.38), with a value of 0.40 in May 2019 (Table 1, Figure 3). This value indicates that, on this date, the sample comprised a relatively high proportion of taxa that are adapted to high current velocity (Armanini *et al.* 2011).

Figure 3. CEFI index values for drift invertebrates (aquatic taxa) in the Quinsam River throughout 2014 – 2019.



5. CLUSTER ANALYSIS

The results of the cluster analysis (based on density data) are provided in a dendrogram (Figure 4). This dendrogram was derived from density data (using the highest available taxonomic resolution) on each sample date. Black lines indicate branching of groups with a dissimilar community composition at a 5% significance level (SIMPROF test); red lines denote groups that are not significantly different in their community composition at a 5% significance level (SIMPROF test).

The dendrogram shows that the invertebrate communities in the Quinsam River first divide into groups based on seasonality at a low similarly threshold (<50); i.e., these groups are the most different from each other. Here, the data generally cluster into three groups representing the early season (May, June), mid-season (July, August) and late season (September, October) invertebrate communities. Within these groups, clustering is primarily explained by temporal proximity as samples from the same month commonly cluster together, followed by other samples from that year, and then samples from other years. As such, the dendrogram shows that the largest differences in similarity are driven by broad seasonal patterns, while smaller differences result from temporal separation. Samples collected at weekly intervals during individual months (rotated each year) are generally similar; this indicates that single samples collected during individual months are representative of that specific month.

The MDS of the Bray Curtis similarity matrices provides an alternative visualization of these results with an ordination plot (Figure 5). The resulting MDS has a stress value of 0.21, where stress values ≤0.1 correspond to a good ordination with negligible possibility of a misleading interpretation, while stress values between 0.1 and 0.2 provide a useful two-dimensional MDS representation as long as there is agreement between the dendrogram (Figure 4) and the MDS plot (Figure 5) (Clarke and Warwick 2001). The relationships displayed by the MDS plot support those described above in relation to the dendrogram, with distinction among the samples collected during different periods in the growing season, even when results for multiple years are considered.

Figure 4. Quinsam River cluster analysis results on the Bray-Curtis similarity matrix.

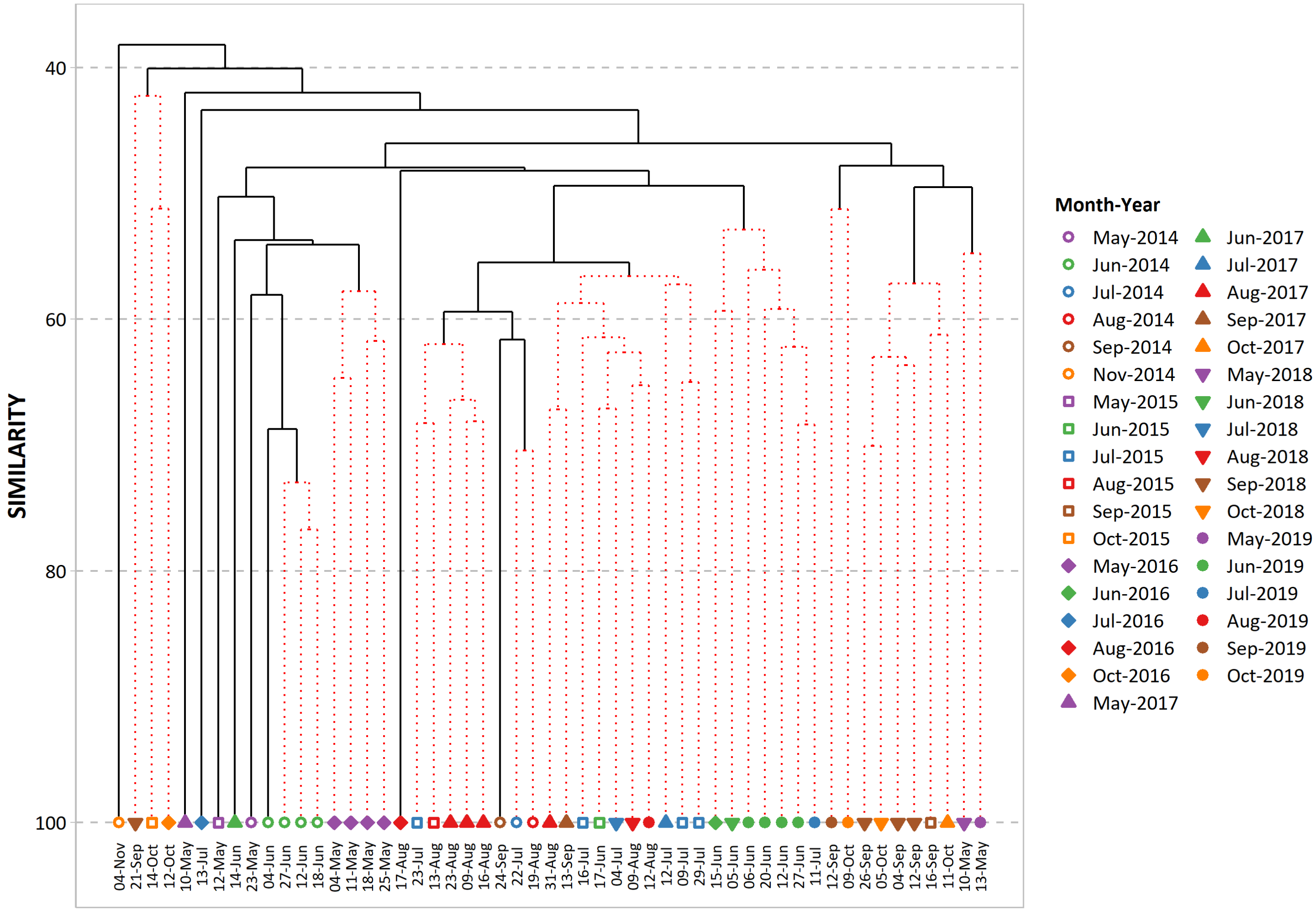
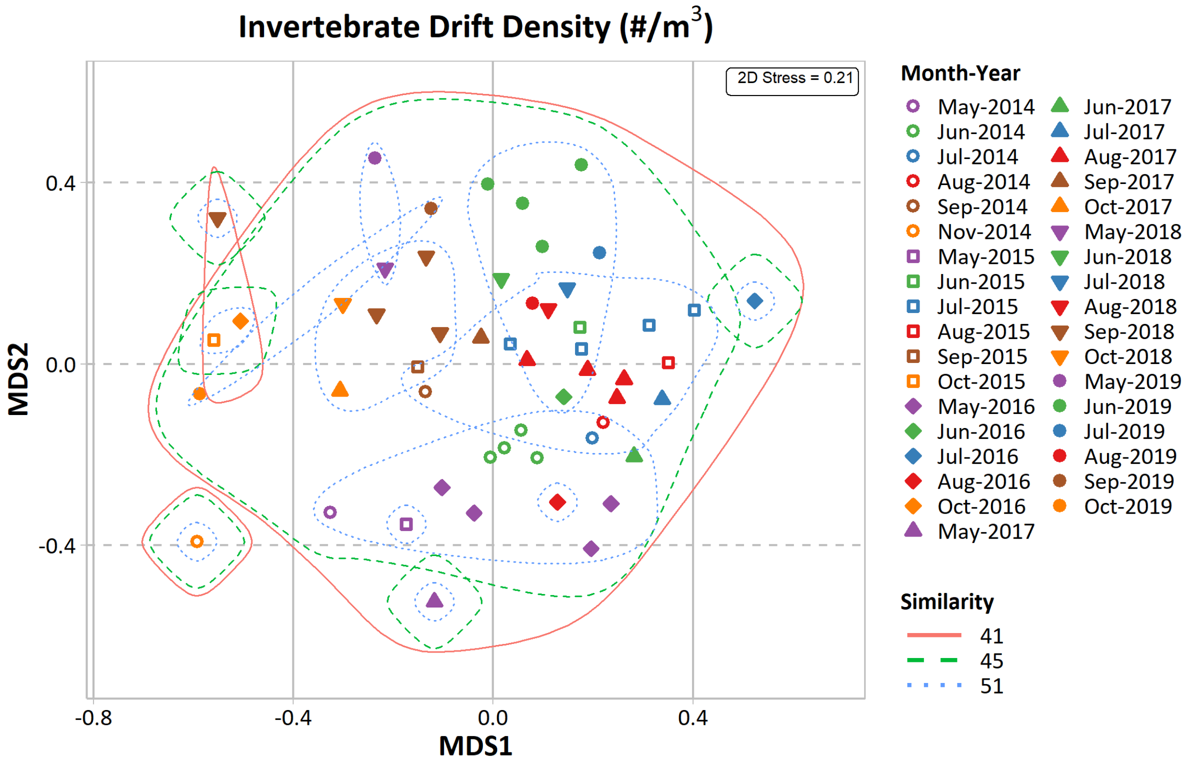


Figure 5. Quinsam River non-metric multi-dimensional scaling ordination plot by date.



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Clarke, K.R., and R.M. Warwick. 2001. Change in marine communities: an approach to statistical analysis and interpretation, 2nd edition. Plymouth, UK: PRIMER-E.