Cheakamus River Project Water Use Plan

Cheakamus River Juvenile Salmon Outmigration Enumeration Assessment Compendium Report

Final Data Report: 2001-2019

Reference: CMSMON1a

Cheakamus River Juvenile Salmon Outmigration Assessment

Study Period: 2001-2019

Prepared for:

BC Hydro

6911 Southpoint Dr, 11th floor

Burnaby, BC V3N 4X8

Prepared by:

Stephanie Lingard, Annika Putt, Katrina Cook and Caroline Melville InStream Fisheries Research, Inc.

1211A Enterprise Way

Squamish, BC V8B 0E8

T: +1 (604) 892-4615

October 22, 2019

Executive Summary

Cheakamus River juvenile salmon abundance was estimated from 2001 to 2019 under CMSMON1a of the Cheakamus River Water Use Plan (WUP) to answer two management questions:

> MQ1. What is the relationship between discharge and juvenile salmonid production, productivity, and habitat capacity of the mainstem and major side-channels of the Cheakamus River?
>
> MQ2. Did juvenile salmonid production, productivity, or habitat capacity change following implementation of the WUP flow regime?

This report focuses on the effects of flow regime on abundance of juvenile Pink, Coho, and Chinook Salmon. The monitoring program examined the effect of two flow treatments. The Interim Flow Agreement (IFA), in place from 2000 to 2006, aimed to approximate a natural hydrograph, with 45% of the previous days inflows being released into the Cheakamus River. The WUP flow treatment, in place from 2007 to 2019, consisted of a set of minimum flows to be maintained throughout the year.

From 2001 to 2019, juvenile salmon were captured in the mainstem Cheakamus River using Rotary Screw Traps (RSTs), and in side channels using fyke nets and weir-style fish fences. Mark-recapture methods were used to estimate weekly abundance for both side channels and mainstem habitats. Coho Salmon abundance estimates were only generated for years 2001 through 2017; the 2018 and 2019 monitoring periods were protracted and did not capture the entire Coho Salmon smolt migration. Chum Salmon and Steelhead Trout data were used in other monitoring programs (CMSMON1b and CMSMON3, respectively) and are thus not presented.

Abundances were highly variable over the monitoring period. Chinook Salmon abundance ranged from 17,000 to 870,000 (2001 to 2019; n = 18), while Coho Salmon abundance ranged from 69,000 to 150,000 (2001 to 2017; n = 17). Pink Salmon, present in odd-years in the Cheakamus River, ranged from abundances of 82,000 to 29,000,000. Comparative abundances from side channel and mainstem river traps indicated the majority (> 60%) of all juvenile salmonids captured originated in the mainstem Cheakamus River.

To address MQ1, a suite of discharge and temperature variables from the Cheakamus River were compiled and assessed as predictors of abundance for juvenile Coho, Pink, and Chinook salmon. Variables were selected based on a set of a priori hypotheses. These hypotheses guided the calculation of monthly variables from raw temperature and discharge data according to relevant life history stages. For Chinook Salmon, linear regression analyses indicated a positive relationship between juvenile abundance

and discharge during the adult spawning window in August (i.e., discharge of 40 $m^3 \cdot s^{-1}$ maintained through-out August). For Pink, and to some extent Coho, our results indicated a positive relationship between discharage in late winter and juvenile abundance. Results from linear models also suggested extreme temperatures and discharge events, which are expected to increase with climate change, are likely to have negavtive effects on salmon populations in the Cheakamus River. Most noteably, Chinook Salmon appear to require more water and cooler temperatures during their July migration and August spawning window.

A statistical comparison of means (T-tests) compared abundances of Chinook and Coho between the two flow treatments to address MQ2, but no significant differences were detected. Pink Salmon were not included in this analysis given limited data. However, statistical power of t-tests comparing salmon abundance was predicted to be low in a prior power analysis done in 2003 due to the small number of observations under each flow treatment (n <= 10 *per* treatment) (Parnell et al., 2003). Generally, a threshold of 0.80 statistical power is acceptedas an appropriate trade off between the risk of type 2 errors and sample size requirments, as a larger sampe size is needed to acheive higher statistical power (Cohen 1992). However, some staticians question whether the 0.80 threshold is high enough given the potential consequences of failing to detect population effects in fields such as ecology and psychology (Di Stefano 2003; Field and Hole 2003). Given that none of the comparisons in this monitor were predicted to reach the 0.80 power threshold for a 50% change in abundnace, we could not conclusively determine if the WUP had an effect on salmon populations and answer MQ2.

We also examined whether the environmental variables identified as being predictive of abundance in the regression analysis differed significantly beween the IFA and WUP flow regimes. Only minimum February discharge was significantly different between IFA and WUP flow treatments.

The results and conclusions of this monitor should be interpreted with caution given the considerable limitations of the dataset. For example, with the project scope limited to monitoring a single life stage, relationships detected may be confounded by factors from other life stages not accounted for, most notably adult spawner abundance. That is, annual variability in juvenile abundance may simply reflect changes in adult spawner abundance, rather than a true effect of environmental change. The lack of power in CMSMON1a is especially concerning given a flood event in 2003 and a chemical spill in 2005, extreme events that could strongly influence regression relationships. Indeed, resulting linear relationships were often driven by a small number of highly influential data points. It is clear that additional data is necessary to accurately assess these relationships. To increase confidence and statistical power, additional years of data collection as well as expantion of project scope to include, for example, fish behaviour and survival between life stages, is required to adequately determine the effects of dam operation on salmon in the Cheakamus River.

Table of Contents

List of Tables

List of Figures

Glossary of Terms

AUC	Area-Under-The-Curve
BTSPAS	Bayesian Time-Stratified Spline Model
DFO	Fisheries and Oceans Canada
IFA	Instream Flow Agreement
IFO	Interim Flow Order
NVOS	North Vancouver Outdoor School
RST	Rotary Screw Trap
SD	Standard Deviation
TH	Tenderfoot Hatchery
VIE	Visual Implant Elastomer
WSC	Water Survey of Canada
WUP	Water Use Plan
YOY	Young-Of-The-Year

1.0 INTRODUCTION

1.1 Background

The Cheakamus River, located in the south coast of British Columbia, is important ecologically and culturally to multiple stakeholder groups and the Squamish Nation. The Squamish Nation harvests salmon in the Cheakamus River for food, social and ceremonial purposes and the watershed provides opportunities for commercial anglers, raft guiding outfitters, and recreation.

The Cheakamus River was dammed for power generation and flood control in 1957. The 28 m high and 680 m long dam impounds the Cheakamus River and creates Daisy Lake Reservoir, which has a water storage capacity of 55,000,000 m^3. Water is diverted from Daisy Lake via an 11 km tunnel through Cloudburst Mountain to a powerhouse on the Squamish River (Figure 1). The maximum capacity of the diversion through Cloudburst to the Squamish River is 60 $m^3 s^{-1}$.

Prior to 1997, the water licence for the Cheakamus Generation Project specified that water must be released for fish. Post construction, minimum flows of 320 cubic feet per second (~9.5 m^3s^{-1}) between April and December and 200 cubic feet per second (~5.6 m^3s^{-1}) year-round were recommended by Fisheries and Oceans Canada (DFO). However, there was no legal requirement for BC Hydro to meet these recommended minimum flows (Mattison et al. 2014). In 1997, DFO issued an Interim Flow Order (IFO) with specific minimum flows for the Cheakamus River. An Instream Flow Agreement (IFA) resulting from the order was implemented in 1999. The IFA specified that the greatest of either 5 $m^3 s^{-1}$ or 45% of the previous days' inflows to the lake be released from Daisy Dam (within a daily range of 37% to 52% and within 45% of the previous 7 days' average) (BC Hydro, 2005).

Uncertainties regarding the effects of the IFA on salmonid populations were identified in 1999 during the water use planning process (BC Hydro, 2005) and monitoring studies were initiated in the spring of 2000 to address the key uncertainties. In 2005, a matrix of minimum discharges was presented to the Water Comptroller in the Cheakamus River Water Use Plan (WUP) (BC Hydro, 2005). The WUP describes discharge rules for the Cheakamus River designed to balance environmental, social and economic values. An objective of the Cheakamus River WUP is to maximize the productivity of wild fish populations. The changes made to the IFA during the creation of the WUP flow structure were based on expected benefits to wild fish populations resulting from increases in available fish habitat (BC Hydro 2005). The new flow order (hereafter, WUP) for the Cheakamus River was approved by the Water Comptroller and implemented on February 26, 2006.

At the time of implementation, effects of the WUP flow regime on fish populations were uncertain. To assess the relationship between fish habitat and fish production, a study using rotary screw traps (RSTs) and mark-recapture methods to monitor juvenile salmonid production began in the spring of 2000 (Melville & McCubbing 2001) and continued annually to 2017 following the terms of reference for Monitor 1a (hereafter, CMSMON1a). Following the end of the mandated WUP monitoring period, an additional two years of data collection were approved under CMSMON1a to provide juvenile Chum Salmon abundances for CMSMON1b (Middleton et al. 2019). The scope of CMSMON1a in the two additional years was limited to the period when Chum Salmon fry were migrating, therefore, Coho Salmon abundance was not derived for 2018 and 2019. The objectives of this updated final report are to synthesize the entire dataset and use it to address the management questions of CMSMON1a.

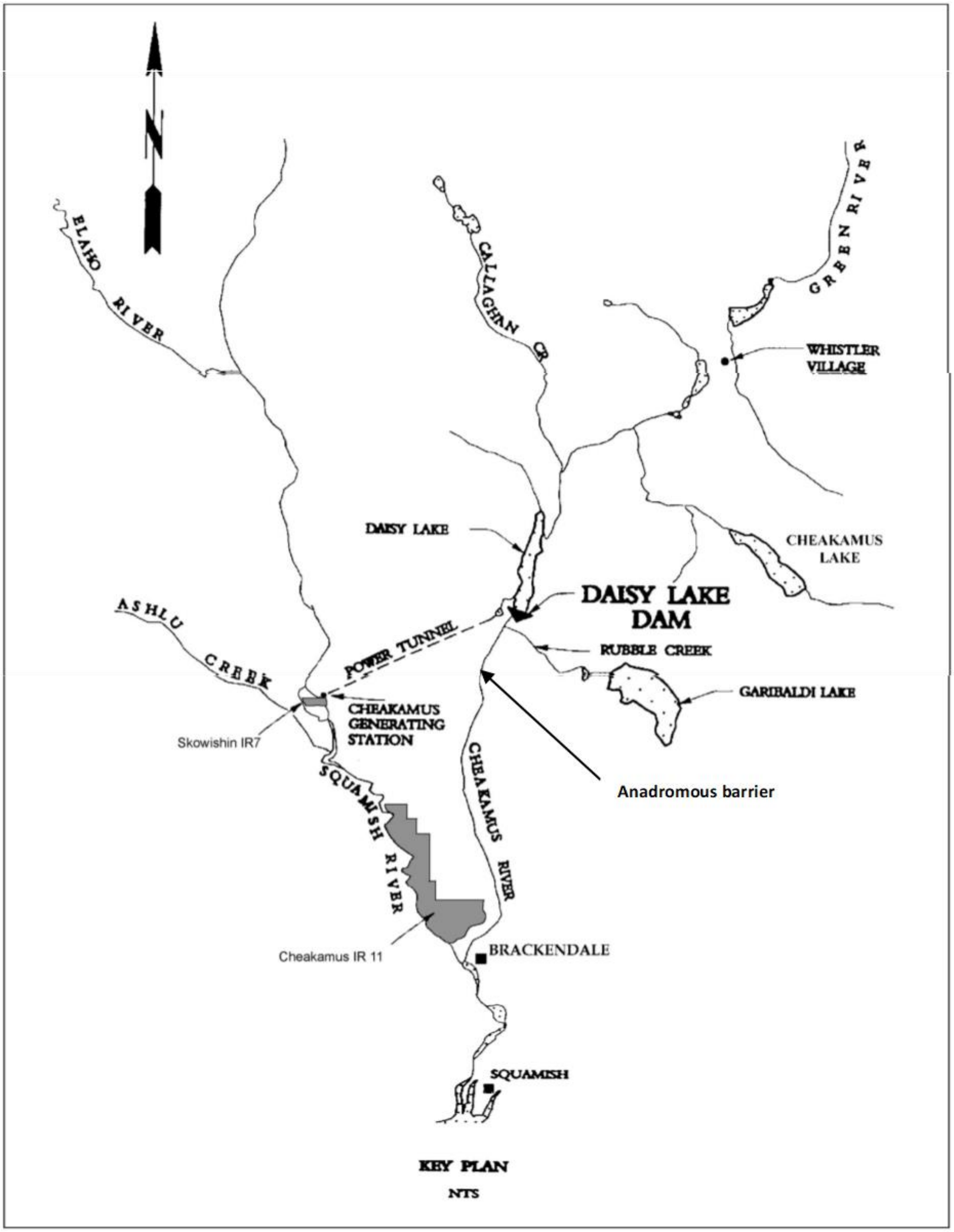

Figure 1. Map of the Cheakamus River and Daisy Generation Project in southwestern British Columbia.

1.2 Management Questions

CMSMON1a aims to assess the effects of the prescribed WUP flows below Daisy Dam on juvenile salmonid production and productivity in the Cheakamus River. The two management questions for CMSMON1a are:

> MQ1. What is the relationship between discharge and juvenile salmonid production, productivity, and habitat capacity of the mainstem and major side-channels of the Cheakamus River?
>
> MQ2. Did juvenile salmonid production, productivity, or habitat capacity change following implementation of the WUP flow regime?

The focus of CMSMON1a is to answer MQ1 and MQ2 for Pink (*Oncorhynchus gorbuscha*), Coho (*O. kisutch*) and Chinook (*O. tshawytscha*) salmon juveniles. Chum Salmon and Steelhead Trout (*O. mykiss*) are discussed in detail in CMSMON1b (Middleton et al. 2018) and CMSMON3 (Korman and Schick, 2019), respectively. Productivity refers to the number of juveniles produced per spawner, while habitat capacity is defined as the asymptote of the spawner recruit curve (BC Hydro 2006). Because spawner abundance was not collected for Pink, Coho and Chinook salmon, juvenile fish abundance (production) was the response metric used to assess the management questions in CMSMON1a.

1.3 Salmon Life-History Characteristics in the Cheakamus River

Salmonids are present in the Cheakamus River year-round. Pink Salmon are odd-year dominant and adults are present in August and September. Juveniles begin emerging in January and migrate downstream between February and May as young-of-the-year (YOY).

Both summer and fall spawning Chinook Salmon populations are present in the watershed. Adults begin entering the Cheakamus River in June, with spawning occurring in August for the summer population and between late September and mid-October for the fall population. Chinook Salmon juveniles express a diversity of life histories resulting from complex trade-offs between genetic and environmental factors (Volk et al., 2010; Bourret et al., 2016). YOY Chinook Salmon start to emerge in November. Between February and May, an unknown portion of the population emigrate shortly after emergence as YOY fry (35 to 50 mm). However, some juveniles will remain for several months and emigrate as large YOY (60 to 80 mm) between May and September (sub-yearling), while others overwinter in the Cheakamus River and migrate the following spring as yearling (or ‘stream-type’) smolts.

Adult Coho Salmon return to the Cheakamus River between October and January and spawning occurs between December and February. Juveniles start to emerge in March and typically remain in the freshwater environment for a year before migration to marine environments as yearling smolts

(Sandercock, 1991). An unknown proportion of the Cheakamus population migrate as YOY to rear in estuarine and marine environments (Lingard, 2015), but recent research from Washington, Oregon and Alaska suggests YOY emigrants contribute significantly to adult Coho returns (Bennett et al., 2015; Koski, 2009).

Wild salmon populations in the Cheakamus River are supplemented by hatchery production from the Tenderfoot Creek Hatchery (operated by DFO). Over the duration of CMSMON1a, the Tenderfoot Creek Hatchery has enhanced Pink, Chum, Coho, and Chinook salmon and Steelhead trout populations in the Cheakamus River. Hatchery production methods and release totals have varied among years. See Lingard et al., (2016) for more detailed information on hatchery releases.

Many factors in both freshwater and marine environments have the potential to affect juvenile salmonid abundance including river discharge (e.g., maximum and minimum discharges and ramping rates [Zimmerman et al., 2001; Connor and Pflug 2004; Zeug et al., 2014; Rebenack et al., 2015]), water temperature (Beer et al., 2001, Murray and McPhail 1988), marine productivity (Hinch et al., 1995; Beamish et al., 2004), predation, and natural and man-made barriers. In the Cheakamus River, discharge is controlled by flow releases at Daisy Dam, and management of Daisy Dam influences discharge and temperature in mainstem habitat where salmon spawn, incubate, rear, and migrate. This report focuses on the effects of environmental factors (and therefore management actions) on juvenile salmon abundance and productivity.

1.4 Flood of 2003 and 2005 Caustic Soda Spill

Two noteworthy events occurred in the Cheakamus River during the monitoring period that should be mentioned due to their impacts on fish populations. First, a rain-on-snow event in October 2003 resulted in a 100-year flood during which Cheakamus River discharge exceeded the rating curve for the WSC Gauge at Brackendale (08GA043), reaching a recorded maximum of 709 $m^3 s^{-1}$ on October 19, 2003.

Second, on August 5, 2005, a CN train de-railed at river kilometer (rkm) 19 and spilled 41,000 liters of caustic soda (NaOH) into the Cheakamus River. This event had significant impacts on fish populations and are documented in detail in McCubbing et al., (2006). Effects of these impacts are considered in the context of the results of this monitoring program.

2.0 METHODS

2.1 Site Description

The Cheakamus River is a major tributary of the Squamish River Watershed, entering approximately 20 km north of Howe Sound (Figure 1). The Cheakamus Watershed covers an area of 1,010 km^2 in the coastal mountain range of southwestern British Columbia and is glacially fed. Annual water temperatures in the anadromous reach of the watershed range from 0.5 to 15 ºC. The Cheakamus River typically includes low flow periods (15-20 $m^3 s^{-1}$) in winter (December to March) and late summer/early fall (August to September), and high flow periods resulting from spring snow melt (April to July) and fall storm events (October to November).

Daisy Dam is located on the Cheakamus River approximately 26 km upstream of the confluence with the Squamish River and impounds Daisy Lake Reservoir. A natural barrier to anadromous fish migration exists 9 km downstream of Daisy Dam at rkm 17, below which the Cheakamus River supports populations of anadromous salmon and trout. Ten species of salmonids are present: Pink, Coho, Chum, Chinook, Sockeye and Kokanee (*O. nerka*) salmon as well as Rainbow and Steelhead Trout, Cutthroat Trout (*O. clarkii*), Bull Trout (*Salvelinus confluen*tus), and Dolly Varden (*S. malma*).

The mainstem habitat in the Cheakamus River is complimented by a large area of man-made restoration channels which are fed either by groundwater or surface water diverted from the mainstem river (Figure 2). The first restoration channel in the Cheakamus River was built in 1982 at the property now known as the Cheakamus Center. In the 1990s and early 2000s, a network of restoration channels was expanded as part of the Dave Marshall Salmon Reserve. Additional channels have been built upstream and downstream of the Cheakamus Center and include Mykiss Channel, BC 49 Channel, BC Rail Channel, Dave’s Pond and Moody’s Channel. In addition to the constructed channels, large woody debris structures were placed in the mainstem Cheakamus River to increase habitat complexity (Harper and Wilson 2007).

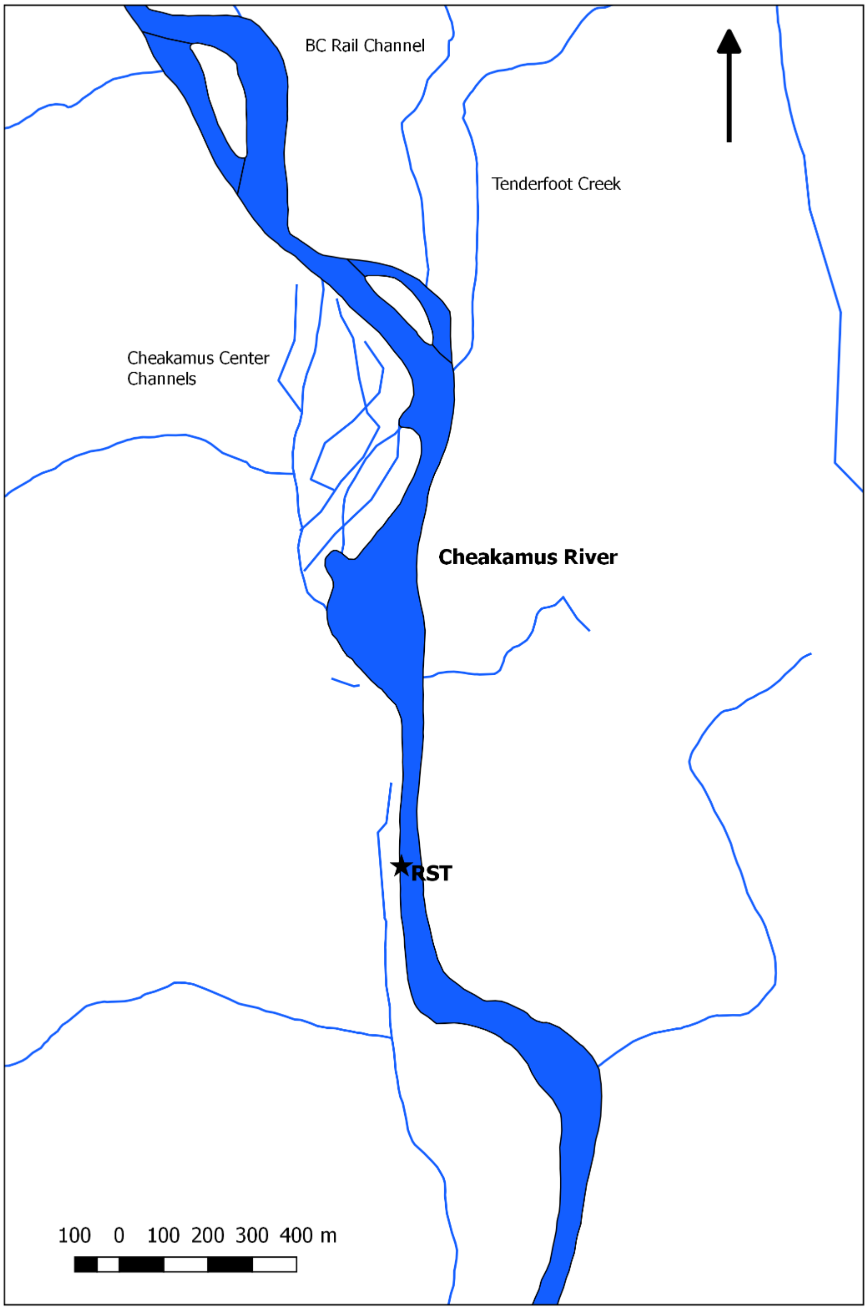

Figure 2. Map of the study area including the Cheakamus River and major side-channels.

2.2 Juvenile Abundance Estimation

This section briefly summarizes methods used to capture and enumerate juvenile salmon during CMSMON1a. Detailed methods of abundance estimation can be found in Lingard et al. (2016), or in previous annual reports at: (https://www.bchydro.com/about/sustainability/conservation/water_use_planning/lower_mainland/cheakamus.html).

2.2.1 Trapping Sites and Fish Capture Methods

Juvenile fish in the mainstem were enumerated by two six-foot RSTs operated adjacent to the Cheakamus Center property at rkm 5.5 (10U 0489141:5518035, Figure 2). Traps were operated annually, typically between February 15 and June 15[1]. Fyke nets were operated in both groundwater and river-augmented (flow through) side-channels in the Cheakamus Center complex, the BC Rail Channel and Tenderfoot Creek (Figure 2). Fence traps, spanning the entire channel, were installed on the Cheakamus Center and BC Rail side-channels to capture yearling Coho Salmon (1+) and Steelhead trout (2+ & 3+) smolts from April 1 to June 15 each year (Figure 2).

2.2.2 Mark-Recapture Abundance Estimation

A modified Petersen mark-recapture model was used to generate abundance estimates for juvenile salmon in the Cheakamus River. In traditional Petersen methods, data pooling between sampling events (or strata) is often required in the event of sparse data. Pooling strata assumes homogeneity in capture probabilities, which is often violated due to varying river discharge and capture effort throughout the run. When heterogeneity is present, pooled Petersen estimators can substantially underestimate uncertainty in abundance estimates. A Bayesian Time-Stratified Spline Model (BTSPAS) was used to estimate annual fish abundance (Bonner & Schwarz, 2011). The BTSPAS model is a modified Petersen mark-recapture model that estimates weekly abundance using splines to model the general shape of the run. The Bayesian hierarchical method shares information on catchability among strata when data are sparse (Bonner and Schwarz 2011). See Bonner and Schwarz (2011) for a detailed explanation of the model and its development.

Abundance estimates were generated for weekly strata for both the RSTs and side-channel fyke nets. Weekly strata for YOY Chinook, Chum and Pink Salmon ran from Tuesday to Monday. Fish captured

[1]Trapping dates varied across years due to environmental factors (discharge events) and, to some extent, increased understanding of juvenile salmon outmigration patterns. For details on annual trapping dates, see annual reports at: https://www.bchydro.com/about/sustainability/conservation/water_use_planning/lower_mainland/cheakamus.html

between Monday and Thursday were marked with a biological stain and released upstream of the RSTs or fyke net. Fish were not marked between Friday and Sunday to allow the mark group to move past the trap before the next strata began (Lingard et al., 2016).

Weekly strata for Coho Salmon and Steelhead trout smolts ran from Monday to Sunday. Fish captured at the Cheakamus Center fence were used as the mark group for the RSTs. Fish were marked daily using Visual Implant Elastomer (VIE) tags and caudal fin clips. Each stratum was assigned a unique mark. Smolts were held in a holding box until dusk. Coho and Steelhead captured at the Cheakamus Center and the BC Rail fence were considered the entire catch for each channel and were counted daily.

Estimates generated from the RSTs represented the combined mainstem and side-channel estimate; those from side-channel traps were subtracted from the RST estimate to determine comparative production from side-channel and mainstem habitat. Hatchery production totals were not included in the population estimates generated from this study.

2.4 Assessing Relationships Between Environmental Variables and Salmon Abundance

We used linear regression modelling to assess potential relationships between various measures of Cheakamus River water temperature and discharge, herein referred to as ‘environmental variables’, and abundance of Pink, Coho, and Chinook Salmon from 2001 to 2019. Discharge was measured hourly throughout the duration of the monitoring period by the WSC Gauge at Brackendale (10U 0489186:5518291), located 100 m upstream of the RST site (Figure 2). Hourly water temperature was collected using an Onset TidbiT v2 data logger (UTBI-001) installed at the RST site. Water temperature was collected from approximately February 15 to June 15 from 2001 to 2006 and year-round beginning in 2007.

2.4.1 Calculation and Selection of Environmental Variables

We reviewed published literature to develop a series of a priori hypotheses relating temperature and discharge in the Cheakamus River to juvenile salmon abundance (Table 1). These hypotheses guided the calculation of variables from raw temperature and discharge data according to relevant life stage or time period for each species (see life stages in Table 2). Within each time period of interest, minimums, maximums, variances and cumulative values were calculated from daily temperature and discharge data. Cumulative discharge was calculated by summing the average daily discharge over a given period of interest. For each species, variables were calculated by individual month within a life history period.

Additional specific discharge metrics were calculated based on either the a priori hypotheses or management implications. The number of days over WUP-specified minimum flows in each month and ramping rates (6-hour rate of change in discharge) for periods of both increasing and decreasing discharge during the emergent and early-rearing life history period of each species (i.e., when expected fork lengths are < 50 mm) were calculated from discharge data. Authorized ramp rates for Cheakamus River currently range from 13 m^3s^{-1} per hour to 13m^3s^{-1} per 10 minutes depending on river stage (Province of British Columbia 2006), which is higher than ramping rates recommended by the DFO. Additionally, because the WUP allows for a decrease in discharge from 38 to 20 $m^3 s^{-1}$ on August 15 during the Chinook and Pink spawning period, discharge metrics were also calculated from August 1 to 14 and August 15 to 31 to test whether this discretionary decrease influences juvenile fish abundance. Although not an a priori hypothesis, this discharge category was included in the analysis due to its management significance. These discharge metrics were also calculated for each species by individual month within a life history period.

Systematic methods guided a process to subset the suite of calculated variables for each species to those most associated with abundance, as detailed in Figure 3. First, all variables were assessed for normality using a Shapiro-Wilk test ($p < 0.05$). Those failing to meet the criteria for normality were log-transformed and re-tested. Only untransformed and log-transformed variables with normal distributions were retained and assessed as predictors of abundance in individual univariate linear regression models.

Variables that significantly predicted abundance in linear regressions were critically assessed using professional judgement. Although less systematic, this process synthesised the data and ensured that presented regression models were biologically relevant. Significant variables were grouped into themes based on Pearson’s correlation coefficients, classification of similar variable categories (e.g., seasons), and consensus-based expert judgement of the Cheakamus River system (e.g. known to be relevant during a specific life history). Selection of the final regression models for each species was context dependent. Where information from multiple correlated predictors was redundant, only the most significant was selected. However, multiple correlated predictors were presented if from distinct life history time periods, if they had known biological significance, or had unique management implications. This process was particularly important for Chinook salmon, that had the largest number of variables remaining following the systematic selection criteria.

Table 1. A priori hypotheses developed for variable selection in linear modelling of Cheakamus River environmental variables and juvenile salmon abundance

Variable	Salmon Life history period	Hypothesis	References
minimum discharge	adult spawning period	Minimum discharge during adult spawning influences adult migration conditions and habitat availability for spawners	Webb et al., 2001 Cheakamus 2D Model
minimum discharge	incubation / rearing / migration	Minimum discharge during juvenile incubation, rearing and migration influences available habitat area	Cheakamus 2D model
discharge variance	adult spawning period	Variability in discharge affects migration timing and behavior in adult salmon	Tetzlaff et al., 2005, 2008; Smith, et al., 1994
discharge variance	incubation / rearing	Variability in discharge during incubation and rearing affects juvenile abundance through stranding related mortality, reduced habitat stability, and early emigration.	Bradford et al., 1997; Freeman et al., 2001; Rebenack et al., 2015; Irvine 1986
discharge variance	migration	Variability in discharge during migration affects migration conditions and influences migration date and age	Zeug et al., 2014
days over minimum discharge	incubation / rearing/ migration	Pulses over minimum discharge during juvenile incubation, rearing and migration cause stranding induced mortality and reduced habitat stability	Bradford et al., 1997; Freeman, et al., 2001; Zimmerman et al., 2015; Bradford et al., 1995
days over minimum discharge	adult spawning period	Pulses of discharge during adult spawning affect influences adult migration conditions and behavior	Smith et al., 1994; Web et al., 2001; Tetzlaff et al., 2005
cumulative discharge	incubation / rearing	Increased cumulative discharge during incubation and rearing influence foraging opportunities, and scour related mortality	Honea et al., 2016; Goode et al., 2013; Tetzlaff et al., 2005
cumulative discharge	adult spawning period	Cumulative discharge during spawning influences migration conditions and habitat availability for spawners	Tetzlaff et al., 2008; Gibbins et al., 2002
cumulative discharge	migration	Cumulative discharge during migration affects survival of rearing migrating juveniles and migration conditions	Zeug et al., 2014
minimum temperature	spawning and incubating	Minimum water temperature influences maturation rate of embryos, date of emergence, and adult spawner success	Beer and Anderson 2001; Murray and McPhail 1988; Geist et al., 2006; Hodgson and Quinn 2002; Goniea et al., 2006
minimum temperature	rearing/ migration	Minimum temperature during juvenile migration period influences juvenile growth and migration timing	Beakes et al., 2014; Jonsson and Ruud-Hansen 1985; Marine and Cech 2004
cumulative temperature	all	Cumulative temperature influences rate of embryo maturation, juvenile growth and adult spawning behavior, and physiological stress of adult salmon	Murray and McPhail 1988; Marine and Cech 2004; Sykes et al., 2009; Wagner et al., 2005
6-hour rate of change in discharge	emergent/ rearing	Rapid changes in discharge can result in high rates of mortality for pre-emergent and newly emergent fry residing in shallow gravel bar habitats.	Bradford et a. 1995; Bradford 1997; Saltviet et al. 2001

Table 2. Start and end dates for freshwater life history periods by species.

Species	Age class	Life history period	Time period
Chinook	Adult	Adult spawning & Migration	Jul 1 to Oct 31
Pink	Adult	Adult spawning & Migration	Aug 1 to Oct 1
Chinook and Pink	YOY (0+)	Incubation and juvenile rearing	Oct 1 to Jan 31
Chinook and Pink	YOY (0+)	Juvenile outmigration	Feb 1 to May 1
Coho	Yearling (1+)	Adult spawning	Nov 1 to Jan 1
Coho	Yearling (1+)	Juvenile rearing	Feb 1 (smolt year-1) to Feb 1
Coho	Yearling (1+)	Juvenile outmigration	Feb 1 to Jun 30

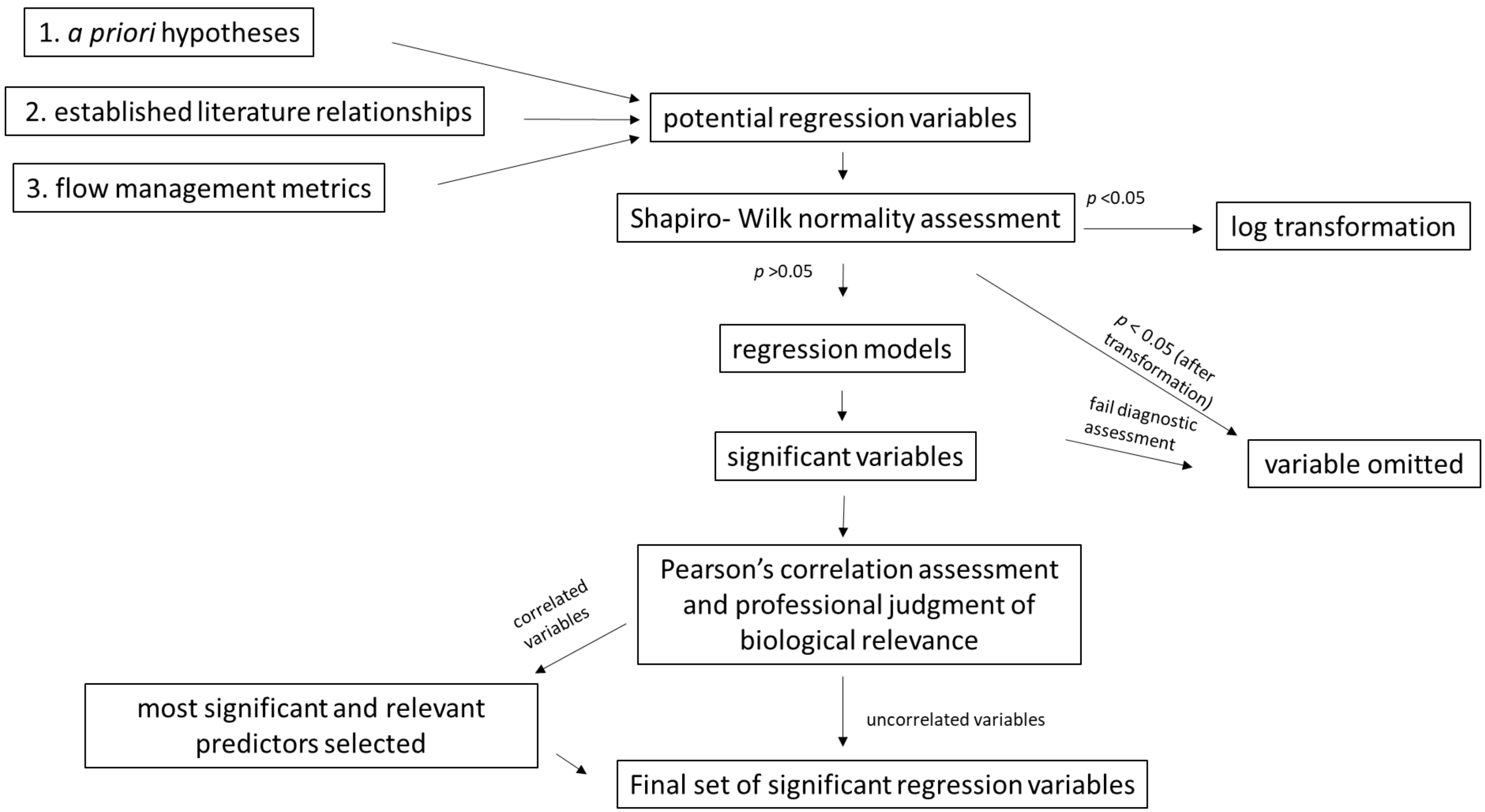

Figure 3 Schematic of analysis methods used for CMSMON1a data synthesis.

2.4.2 Variable Assessment via Matrices of Correlation Coefficients

Multivariate linear regression is typically used to assess the effects of temperature and discharge on salmon productivity (e.g., Arthaud et al., 2010, Zeug et al., 2014). However, limitations of our dataset precluded our ability to do so (e.g., abundance data instead of productivity data, large numbers candidate variables, multicollinearity). The presented statistical analyses used systematic and informed methods to coerce a complex multivariate dataset into a series of univariate linear regressions. To visually assess the multivariate nature of the dataset and support regression results, matrices of correlation coefficients among variables of interest were created for each species. Also known as heatmaps, these matrices show the strength and direction of correlation between all combinations of variables included within, which are ordered according to their similarity to one another (shown by a dendrogram). Inclusion of such a multivariate visual assessment in this synthesis report both informs interpretation of results from analyses and provides a finer scale assessment of relationships with productivity, and among relevant environmental variables. This provides a broad visual summary of relationships between environmental variables and abundance and further context for interpreting regression results.

The variable set used to make correlation coefficient matrices was specific to each species and included all those generated using a priori hypotheses that passed the assumptions of normality (before or after log-transformation). Matrices therefore included variables that significantly predicted abundance in linear regressions and well as non-significant variables. Only variables that had a correlation coefficient of >|0.3| with abundance were included in the heatmaps. Variables were clustered using a complete agglomeration method and Euclidean distance was used to create a dendrogram that provides a measure of similarity among variables (Warnes et al., 2016) . The colour of each cell represents the strength of the correlation between two variables, with the shade and vibrancy of the colour representing the direction and strength of the correlation, respectively.

2.5 IFA-WUP Comparisons of Mean Juvenile Salmon Abundance and Discharge Statistics

We used t-tests to test for a significant difference in mean juvenile salmon abundance during the IFA (2000 to 2006, N=6) and WUP (2007 to 2019, N=12) flow treatments for all species but Pink Salmon that are only present in odd years (due to a low sample size; IFA N=3, WUP N=6). The type of t-test (Student’s or Welch’s) selected for each species was dependent on whether the assumptions of equal variance (Levene’s Test) and normality (Shapiro-Wilk) were met (α = 0.05).

We also used t-tests to assess whether discharge variables identified as significant during linear regression modelling were significantly different between IFR and WUP flow regimes to determine if flow management decisions may have affected environmental variables. We could not assess the change between IFR and WUP regimes for water temperature variables because data from the IFR period was insufficient.

The power of t-tests comparing IFA and WUP juvenile salmon abundances was projected to be low (< 0.80) during the development of the monitoring program when pre-hoc tests were done to determine study design (Parnell et al., 2003). Statistical power, or the probability a test will detect a true effect, is dependent on both sample size and the size of the effect to be measured. Detecting smaller effects takes larger sample size (more years) than detecting larger effects (Cohen 1992). Given the small number of abundance estimates under each flow treatment and the high variability among the data it was predicted, at the 5-year review period, a sample size of 10 to 800 years under each flow treatment would be required to detect a 50% change in abundance with a statistical power of 0.8, depending on species (Melville & McCubbing, 2012). No further power analyses have been conducted.

3.0 RESULTS

3.1 Pink Salmon

3.1.1 Pink Salmon Juvenile Abundance Estimates

Between 2001 and 2018, annual juvenile Pink Salmon abundance ranged from 82,834 to 29,314,436 with a mean abundance of 8,265,871 (SD 11,065,313) (Table 4, Figure 4). Annual abundance estimates had high precision with coefficients of variation (*cv*) ranging from 0.01 to 0.17 (Table 4). Annual catch of Pink Salmon from RSTs ranged from 27,038 to 1,900,820 fish (Table 4). Between 10% and 57% of Pink Salmon were found to migrate out of side-channels (Table 4). Average side-channel production was 22% of total YOY Pink Salmon abundance (Table 4).

Table 3. Annual abundance estimates for YOY Pink Salmon leaving the Cheakamus River between February 15 and May 1 from 2002 to 2018. Capture of YOY Pink Salmon in odd years were near zero in the Cheakamus River.

Year	Mean abundance	SD	97.5% Lower	97.5% Upper	*cv*	Annual catch	Percent counted in side-channels
2002	1,671,625	286,619	1,274,882	2,303,970	0.17	27,038	Not assessed
2004	82,834	13,474	60,785	113,686	0.16	2,742	Not assessed
2006	303,488	9,817	285,605	323,715	0.03	41,336	Not assessed
2008	2,060,948	89,979	1,898,856	2,247,535	0.04	41,873	57%
2010	6,157,377	606,896	5,191,698	7,547,475	0.10	238,730	10%
2012	29,314,436	630,824	28,145,838	30,583,733	0.02	1,447,749	11%
2014	25,387,473	31,4061	24,782,837	26,014,983	0.01	1,900,820	14%
2016	5,491,140	260,514	5,032,642	6,046,211	0.05	258,353	19%

2018	3,921,349	126,521	3,700,595	4,183,279	0.03	261,693	29%

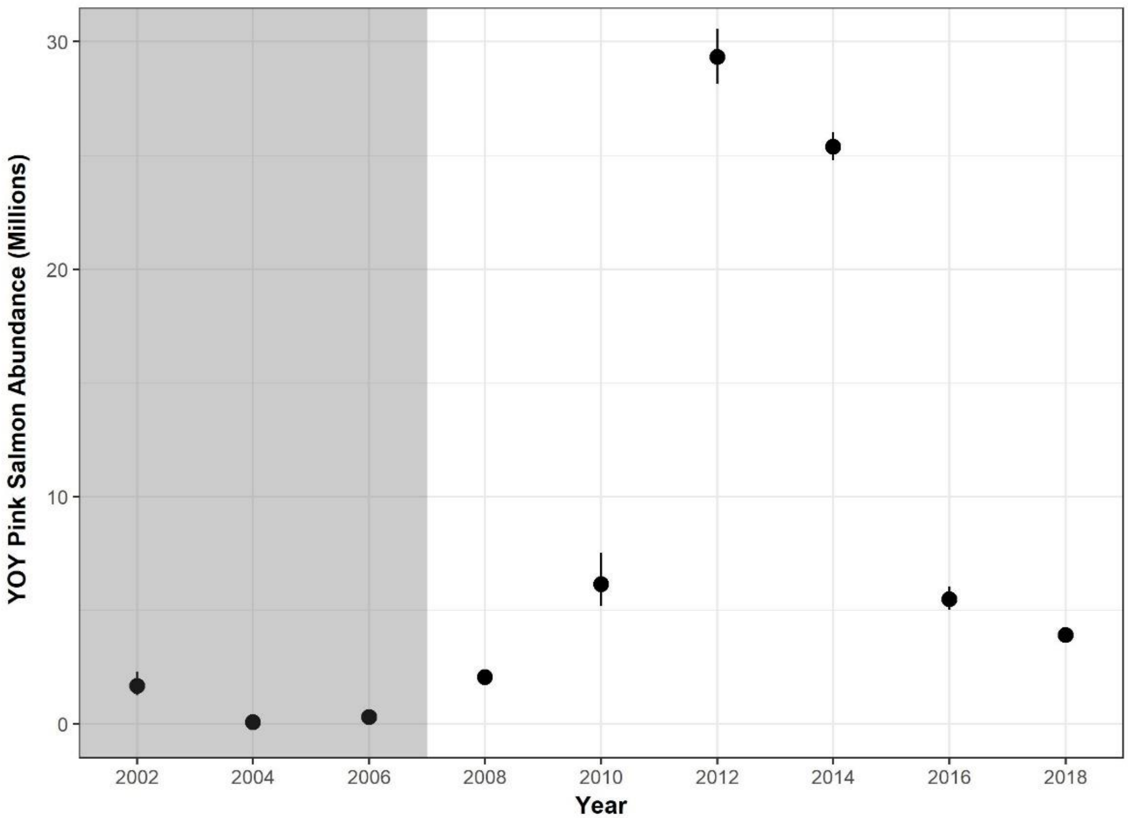

Figure 4. Annual abundance estimates of YOY Pink Salmon in the Cheakamus River. Error bars represent 97.5% confidence intervals. Grey shaded area represents abundance estimates under IFA flow conditions. Non-shaded area represents abundance estimates under WUP flow conditions.

YOY Pink Salmon abundance increased between IFA and WUP time periods, but no statistical testing was conducted for to low sample sizes (*N*=3 IFA, *N*=6 WUP). Mean YOY Pink Salmon abundance during IFA (2002-2006) and WUP (2008-2018) time periods was 686,706 (SD 861,934) and 12,055,454 (SD 11,996,220), respectively. Annual abundance estimates during IFA and WUP time periods did not overlap (ranged from 82,680 to 1,671,625 and from 2,060,948 to 29,314,436, respectively).

3.1.3 Linear Regression Modeling

For Pink Salmon, a priori hypotheses generated a total of 62 candidate environmental variables, 47 of which passed the assumptions of normality and were included in the linear regression modelling (Appendix A). Three of these 47 variables were found to have a significant linear relationship with Pink Salmon abundance (Appendix B):

- Fall discharge: Discharge variance in October and minimum discharge in November both had negative linear relationships with Pink Salmon abundance, suggesting stable baseflows during the incubation and rearing period may be associated with higher Pink Salmon abundance. However, both regressions were influenced by 2004 data and when this high leverage point was removed, statistical significance was lost[2]. The relationship between abundance and minimum discharge in November was the most significant (Table 4, Figure 5).
- Late Winter/ early Spring discharge: Minimum discharge in February showed a positive linear relationship with Pink Salmon abundance, suggesting higher discharges during the onset of the outmigration may be associated with higher Pink Salmon abundance (Table 4, Figure 5).

We used a t-test to determine whether the significant discharge variables in Table 4 were statistically different between the WUP and IFR periods. Only minimum February discharge was significant, being higher in the WUP regime relative to the IFR regime (Table 4).

Table 4. Summary of a subset of significant regression results between log YOY Pink Salmon abundance and Cheakamus River environmental variables. Results of t-test comparisons significant variables between IFA/ WUP are also presented.

Variable	**Life stage**	**Predictors of abundance**				**IFA/WUP comparisons**
		df	***p*-value**	***R*²**	**Direction**	**t (p-value)**
Log_{10} minimum November discharge	Incubation/ emergence	7	0.010	0.61	negative	6.00 (<0.001)[a]
Minimum February discharge	Juvenile outmigration/ rearing	7	0.003	0.72	positive	0.23 (0.820)[b]

a=student's t-test, b=Welch's t-test

[2] With the 2004 data point removed, the R^2 value for minimum November discharge decreased from 0.61 to 0.26 and the model was no longer significant (p-value changed 0.01 to 0.19). Similarly, the R^2 value for discharge variance in October declined from 0.47 to 0.03 and the model was no longer significant (p-value 0.04 to 0.07).

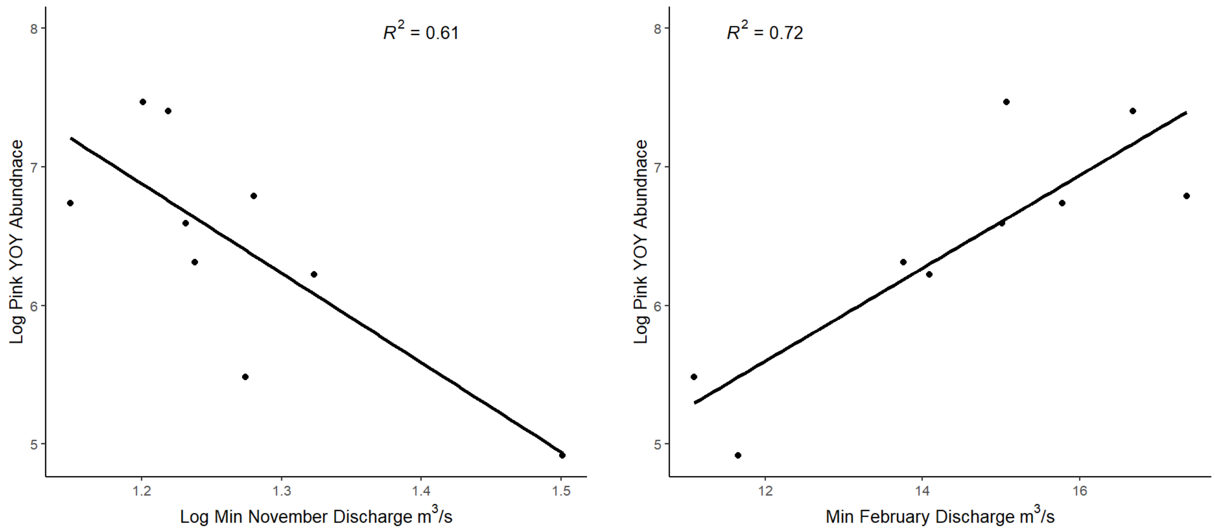

Figure 5. Plots of a subset of significant linear relationships between YOY Pink Salmon abundance and Cheakamus River environmental variables.

3.1.4 Visual assessment of Correlation Coefficients

Of the 47 environmental variables meeting the assumptions of normality, 21 had Pearson's correlation coefficients with abundance of >|0.3| and were included in heatmaps. The four variables that clustered closest with abundance (all positively correlated) were discharge variance in the late summer (August and September), and minimum discharge in the late winter (February and March). The close clustering of minimum discharge in February and March agrees with and supports linear modelling results indicating that increased minimum discharge in late winter and early spring may be associated with higher YOY Pink Salmon abundance. Taken together, the weight of evidence suggests that the most influential variable affecting Pink Salmon abundance may be minimum discharge in the late winter. However, the clustering of discharge variance in the August and September does not agree with the regression modeling. Discharge variance in October and minimum discharge in November did significantly predict abundance in the regression modeling, but in addition to the difference in timing, the direction of the relationship differed between the statistical modeling (negative) and heatmaps (positive).

As with linear modeling, the outlier data of 2004 is still a concern as an influential point.

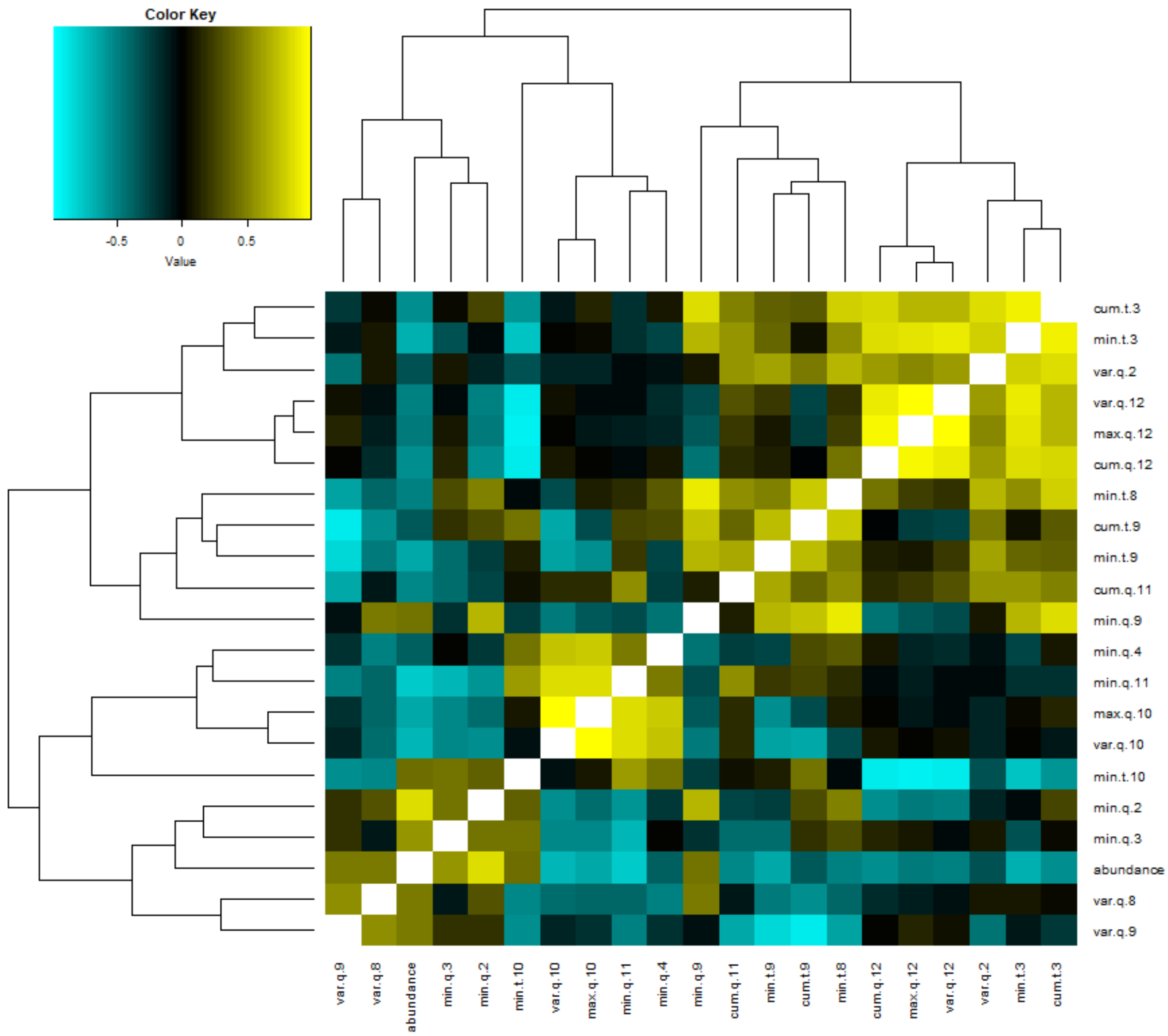

Figure 6. Dendrogram clustering and matrix of Pearson's correlation coefficients with YOY Pink Salmon abundance. An r ≥ |0.3| was used as a threshold for variables to be included in the dendrogram. Black lines form the dendrogram and indicate clusters of related variables based on Euclidian distance. The row and column labels show abbreviated environmental variable names using the naming convention: 'category.variable.month' whereby cum = cumulative, var = variance, min = minimum, max = maximum, q = discharge, t = temperature; the month is designated numerically.

3.2 Coho Salmon

3.2.1 Coho Salmon Juvenile Abundance Estimates

Between 2001 and 2017, annual Coho Salmon abundance ranged from 28,712 to 119,815 (mean 76,908 SD 25,202; Table 6, Figure 7). The coefficient of variation for Coho smolt abundance estimates ranged from 0.04 to 0.27. Annual catch of Coho Salmon from RSTs ranged from 1,048 to 15,060 fish. The percentage of smolts migrating out of side-channels was between 9% and 36% of the total annual abundance.

Table 5. Annual estimates of Coho Salmon smolt abundance generated by the BTSPAS model.

Year	Mean abundance	SD	97.5% Lower	97.5% Upper	*cv*	Annual catch	Percent counted in side-channels
2001	74,537	12,713	68,534	94,444	0.17	3,696	36%
2002	100,653	26,972	74,291	160,517	0.27	2,549	Not assessed
2003	118,161	9,833	104,299	141,550	0.08	5,823	Not assessed
2004	71,481	15,437	53,504	108,386	0.22	1,048	Not assessed
2005	61,472	8,316	48,448	80,513	0.14	1,609	Not assessed
2006	35,444	3,744	29,416	44,350	0.11	1,165	Not assessed
2007	97,832	5,882	87,798	110,736	0.06	7,237	Not assessed
2008	81,624	11,367	63,999	108,508	0.14	3,036	Not assessed
2009	60,686	8,238	50,802	80,920	0.14	6,614	22%
2010	101,271	3,687	95,281	109,805	0.04	10,681	24%
2011	62,593	4,359	55,276	72,393	0.07	5,238	14%
2012	66,944	5,599	58,222	79,329	0.08	6,194	19%
2013	83,707	3,321	77,765	90,817	0.04	7,244	18%
2014	119,815	15,425	99,185	157,584	0.13	15,060	19%
2015	28,712	1,541	26,014	32,108	0.05	2,748	17%
2016	69,120	8,539	57,206	90,552	0.12	6,250	Not assessed
2017	73,390	14,148	61,775	98,141	0.19	13,431	9%

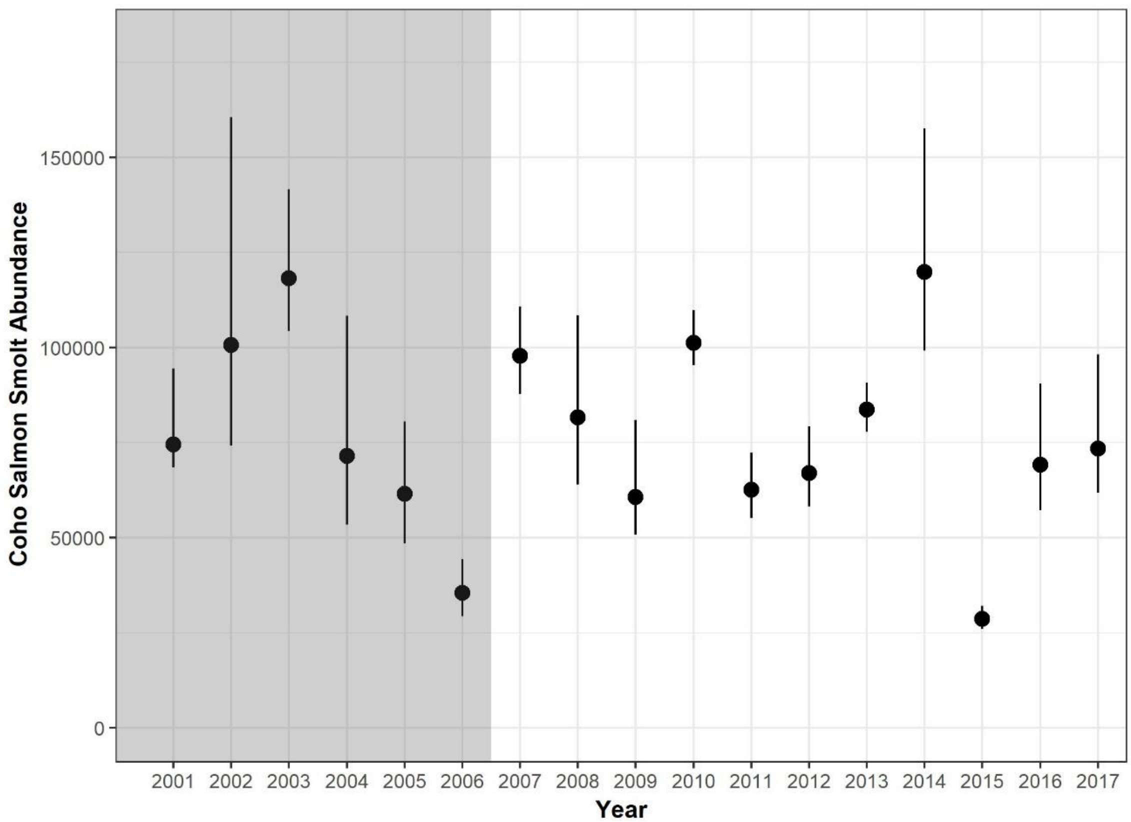

Figure 7. Annual abundance estimates of Coho Salmon yearling smolts leaving the Cheakamus River. Error bars represent 97.5% confidence intervals. Grey shaded area represents abundance estimates under IFA flow conditions. Non-shaded area represents abundance estimates under WUP flow conditions.

Mean Coho Salmon smolt abundances during IFA and WUP time periods were 76,958 (SD 29,183) and 76,881 (SD 24,298), respectively. Annual smolt abundance overlapped during the IFA and WUP time periods (ranged from 35,444 to 118,161 and from 28,712 to 119,815, respectively). There was no significant difference in mean smolt abundance between the IFA and WUP flow treatments (Student's t-test: $t(15)=0.16$, p-value=0.99).

3.2.3 Linear Regression Modeling

For Coho Salmon, a priori hypotheses generated a total of 77 candidate environmental variables, 59 of which passed the assumptions of normality and were included in the linear regression modelling (Appendix A). Six of these 59 variables were found to have a significant linear relationship with Coho Salmon abundance (Appendix C); however, there was considerable collinearity among predictors. We identified three general themes, within which variables with strong linear relationships and/or clear biological significance were selected to show as examples.

- Late Summer temperature: Minimum September temperature showed a negative linear relationship with Coho smolt abundance (Table 6, Figure 8); however, this relationship was highly influenced by the low abundance in 2015. With this point removed the relationship was no longer significant. The winter of 2014/ 2015 was extremely wet with several discharge events over 300 m^3s^{-1}.
- Winter discharge: Cumulative December discharge, maximum December discharge, discharge variance in December, and discharge variance in February all showed negative linear relationships with Coho smolt abundance. These results suggest that consistent baseflows and low discharge variation during the winter portion of the Coho Salmon parr rearing period may be associated with higher Coho smolt abundances. The R^2 values were low for these regressions; therefore, results should be considered cautiously. Variance in December discharge was the most significant predictor of abundance (Table 6, Figure 8).
- Fall discharge: Variance in discharge in October showed a negative linear relationship with Coho smolt abundance (Table 6, Figure 8), suggesting lower flows during the fall parr rearing period may be associated with higher Coho smolt abundance; however, the low R^2 value suggests this relationship is weak.

We used a t-test to determine whether the significant discharge variables in Table 6 were statistically different between the WUP and IFR flow regime periods, but none yielded significant results.

Table 6. Summary of a subset of significant regression results between Coho Salmon smolt abundance and selected variables. Results of t-test comparisons significant variables between IFA/ WUP are also presented.

Variable	Life stage	Predictors of abundance				IFA/WUP comparisons
		df	*p*-value	R^2	Direction	t (p-value)
September minimum temperature	rearing	8	0.01	0.58	negative	NA
Log_{10} December discharge variance	rearing	15	0.02	0.29	negative	0.63 (0.53)[a]
Log_{10} October discharge variance	rearing	14	0.05	0.24	negative	0.98 (0.34)[a]

a=student's t-test, b=Welch's t-test

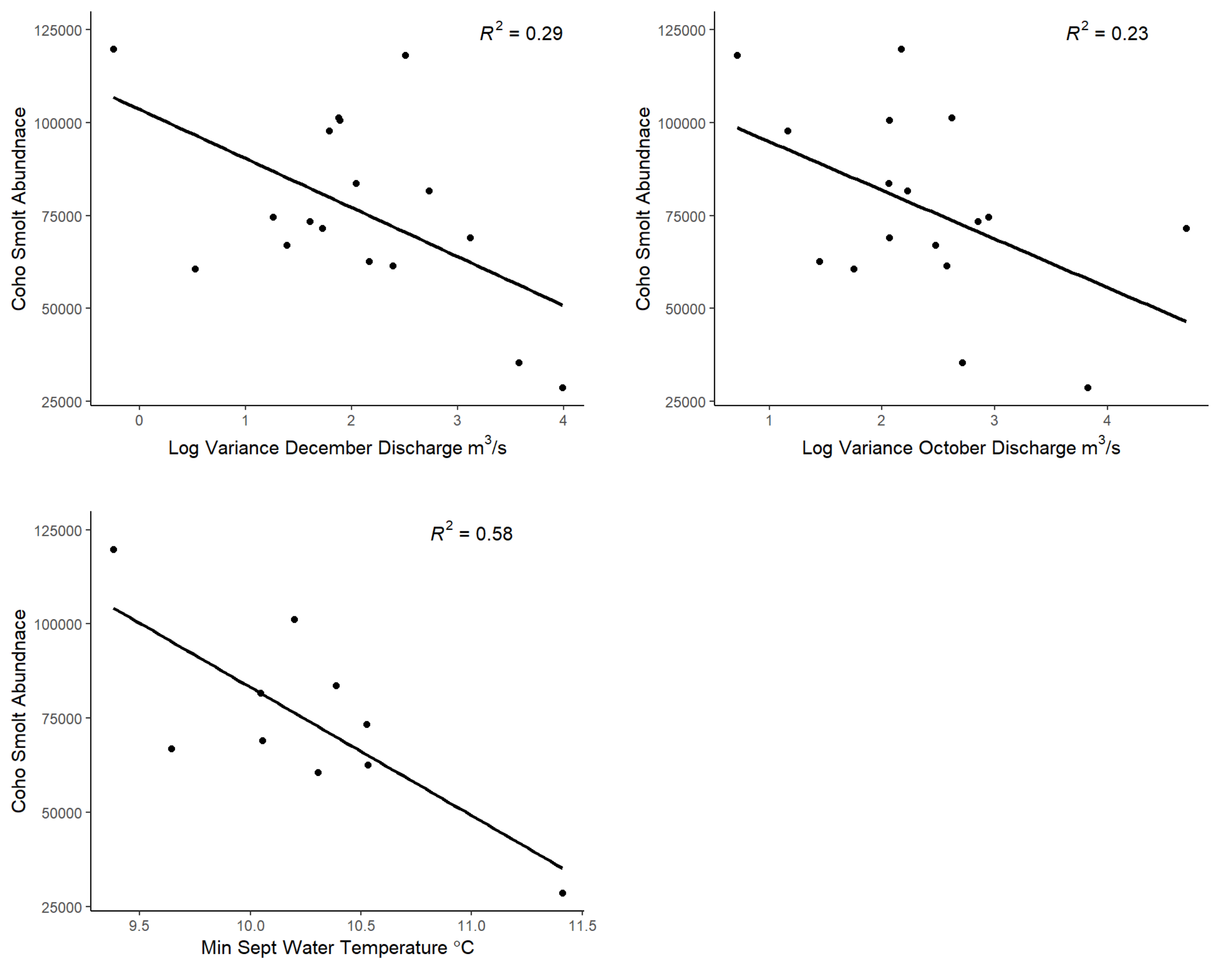

Figure 8. Plots of a subset of significant linear relationships between Coho Salmon smolt abundance and selected variables.

3.2.4 Visual assessment of Correlation Coefficients

A total of 59 environmental variables met the assumptions of normality and were considered, 22 of which had correlation coefficients >|0.3| with abundance. Heatmap results were inconsistent with linear modelling, suggesting associations between assessed environmental data and Coho Salmon smolt abundance were weak. Variables clustering with abundance (cumulative and maximum discharges in March and ramping rates from March through July) differed from those identified as predictors of abundance in linear modelling). The lack of agreement between the heat map clustering and linear models indicates that the variability in abundance is likely explained equally or more by other variables than by the predictor identified in the linear modeling. These inconsistencies combined with only weakly significant (high *p*-values and low R^2 values) linear models suggests there may be too much variability in the data to identify strong trends or associations, and that discharge variance in the fall and winter may not be an important indicator of Coho Salmon smolt abundance.

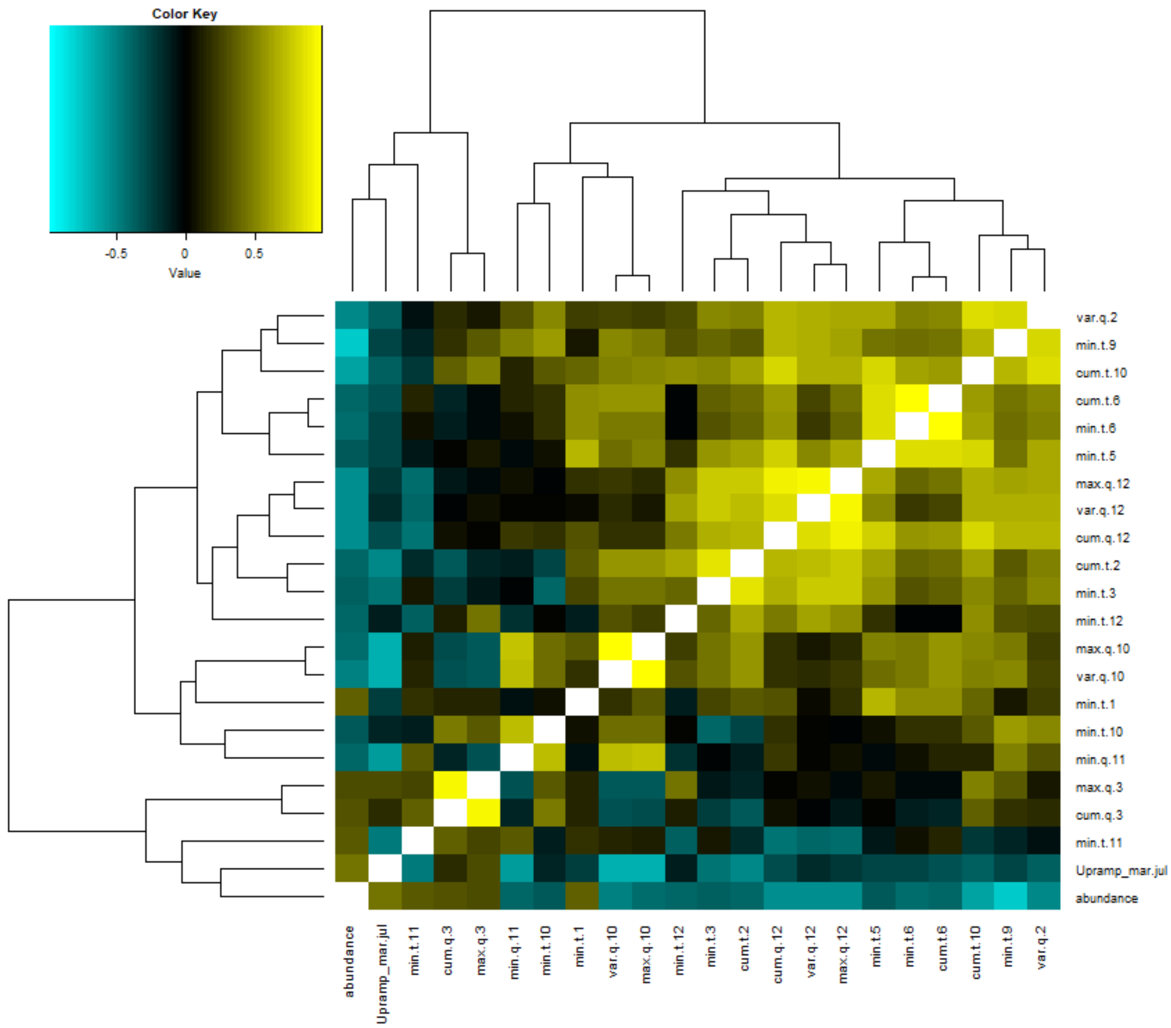

Figure 9. Dendrogram clustering and heatmap of Pearson's correlation coefficients with Coho Salmon abundance. An r ≥ |0.3| was used as a threshold for variables to be included in the dendrogram. Black lines indicate clusters of related variables based on Euclidian distance. The row and column labels show abbreviated environmental variable names using the naming convention: 'category.variable.month' whereby cum = cumulative, var = variance, min = minimum, max = maximum, upramp = the maximum six hour up ramping rate, q = discharge, t = temperature, and the month is designated numerically.

3.3 Chinook Salmon

3.3.1 Chinook Salmon Juvenile Abundance

Between 2001 and 2019, YOY Chinook Salmon abundance ranged from 16,484 to 874,946 (mean 239,657, SD 213,794; Table 8, Figure 8). An abundance estimate was not generated in 2006 (the year following the 2005 caustic soda spill) due to insufficient catch of Chinook Salmon (*N*=499).

Precision of annual YOY Chinook Salmon abundance estimates was the lowest of the three-species reported in CMSMON1a (*cv* range: 0.04 to 0.67, Table 8). Annual catch of YOY Chinook Salmon from RSTs was also lower than for Pink or Coho salmon. In 16 of the 18 years for which YOY Chinook Salmon abundance estimates were generated, annual catch was less than 10,000 fish (Table 8). Very few (<10 per year) YOY Chinook Salmon were captured in side-channel traps; abundance estimates were not generated for side-channels.

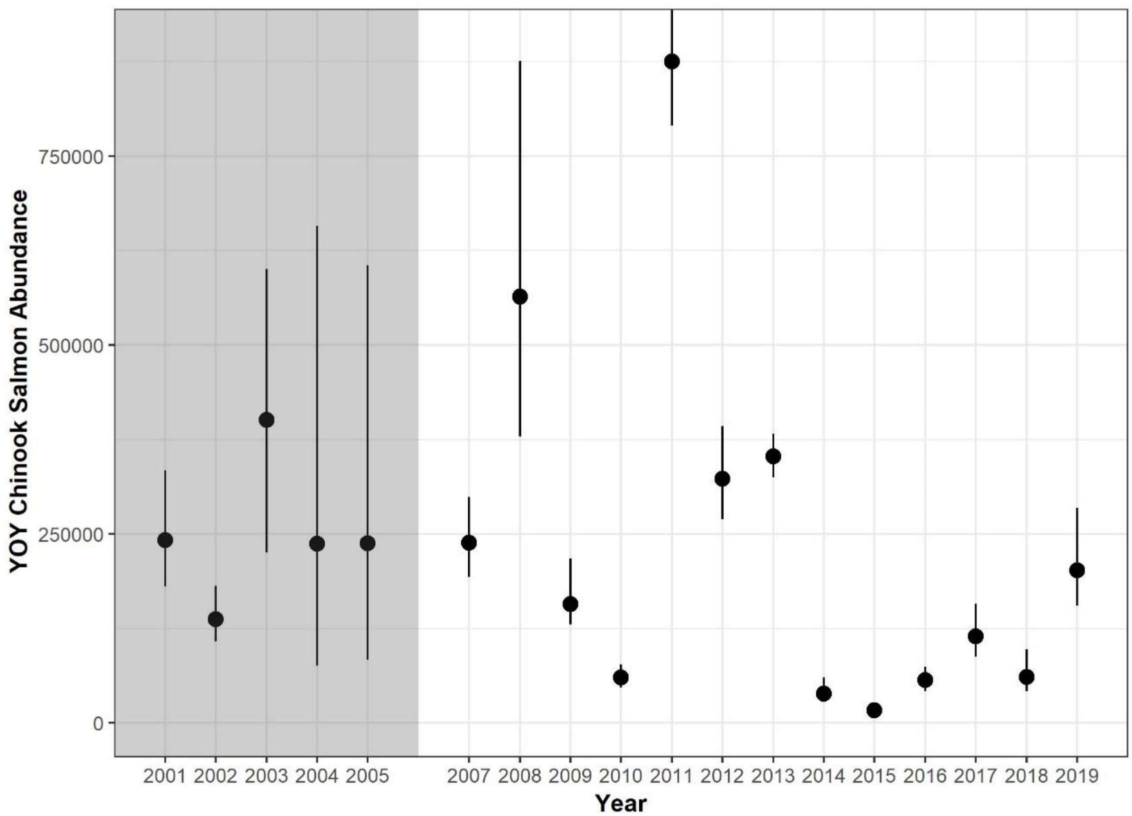

Figure 10. Annual abundance estimates for YOY Chinook Salmon in the Cheakamus. Error bars represent 97.5% confidence intervals. Grey shaded area represents abundance estimates under IFA flow conditions. Non-shaded area represents abundance estimates under WUP flow conditions.

Table 7. Annual estimates of YOY Chinook Salmon abundance generated by the BTSPAS model. No abundance estimate was generated in 2006 due to insufficient catch.

Year	Mean abundance	SD	97.5% Lower	97.5% Upper	*cv*	Annual catch
2001	167,946	39,688	180,674	333,839	0.16	8,578
2002	131,623	18,966	107,404	181,068	0.14	7,567
2003	385,534	98,652	225,488	600,794	0.25	5,859
2004	204,896	159,17	76,061	657,876	0.67	1,232
2005	211,909	154,69	83,365	605,230	0.65	1,107
2006	NA	NA	NA	NA	NA	499
2007	198,588	27,475	193,121	299,055	0.12	8,737
2008	564,313	132,30	378,680	876,185	0.23	5,127
2009	157,151	21,335	130,562	217,512	0.14	8,039
2010	60,040	7,799	47,132	77,166	0.13	3,649
2011	874,946	46,220	790,305	970,473	0.05	31,933
2012	323,375	32,315	269,226	392,903	0.10	8,787
2013	352,356	14,881	325,128	382,873	0.04	22,248
2014	39,001	9,413	27,941	59,812	0.24	3,154
2015	16,484	3,100	12,062	24,014	0.19	1,111
2016	56,470	8,474	41,910	74,511	0.15	1,922
2017	114,146	20,781	87,365	157,560	0.18	6,477
2018	60,931	15,408	42,317	97,189	0.25	3,659
2019	202,127	34,350	155,042	284,848	0.16	7,786

Mean abundance of YOY Chinook Salmon during the IFA and WUP time periods was 250,860 (SD 94,732) and 235,348 (SD 248,372), respectively. Annual YOY Chinook Salmon abundance estimates were normally distributed after $\log_{10}$ transformations and the assumption of equal variance was met. Mean Chinook Salmon abundance was not statistically different between the IFA and WUP time periods (Student's t-test: $t(17)=0.63$, p-value=0.55).

3.3.3 Linear Regression Modeling

For Chinook Salmon, a priori hypotheses generated a total of 66 candidate environmental variables, 50 of which passed the assumptions of normality and were included in the linear regression modelling (Appendix A). Thirteen of these 50 variables were found to significantly predict abundance (Appendix D); however, there was considerable collinearity among predictors. We identified four general themes, within which variables with strong linear relationships and/or clear biological significance were selected to show as examples.

- Winter discharge: Minimum discharge in January, minimum discharge in February, and up-ramping rate over the winter were highly correlated with each other and all had a weakly significant negative linear relationship with abundance (R^2-values ~ 0.3). These results suggest that consistent and low baseflow in the winter incubation and rearing period may be associated with higher Chinook Salmon abundance. Of all variables within this theme, minimum discharge in January was the strongest predictor of abundance (Table 8, Figure 11).
- Summer discharge: Maximum discharge in July, discharge variance in July, and minimum discharge in August all showed significant positive linear relationships with Chinook Salmon abundance. This suggests that higher and more variable discharge in the summer during adult spawning may be associated with higher Chinook Salmon abundance. The strongest linear relationship of this variable grouping was minimum August discharge (Table 8, Figure 11).
- Summer temperature: Minimum temperature in July and cumulative temperatures in July, August, and September were correlated with each other and all had a significant negative linear relationship with Chinook Salmon abundance. This suggests that higher water temperatures in the summer during adult spawning is associated with lower Chinook Salmon abundance. The strongest linear relationship was for cumulative August temperature (Table 8, Figure 11).
- Spring temperature: Minimum and cumulative temperatures in April were correlated and had significant negative linear relationships with Chinook Salmon abundance. This suggests higher water temperatures in the spring during the juvenile outmigration may be associated with lower Chinook Salmon abundance. Cumulative April temperature was the strongest predictor of abundance (Table 8, Figure 11).

We used a t-test to determine whether the significant discharge variables in Table 8 were statistically different between the WUP and IFR periods. Neither January or August minimum discharge were significantly different between WUP and IFA flow regimes (Table 8).

Table 8. Summary of a subset of significant regression results between YOY Chinook Salmon abundance and Cheakamus River environmental variables. Results of t-test comparisons significant variables between IFA/ WUP are also presented.

Variable	Life stage	Predictors of abundance				IFA/WUP comparisons
		df	*p*-value	R^2	Direction	t (p-value)
minimum July temperature	Adult migration	11	0.002	0.63	Negative	Na
cumulative April temperature	Juvenile migration	11	0.003	0.55	Negative	Na
Log_{10} August minimum discharge	Adult spawning	16	0.001	0.50	Positive	0.57 (0.95)[a]
minimum January discharge	Juvenile rearing	16	0.020	0.29	Negative	-1.42 (0.20)[a]

a=Welch's ttest, b=Student's ttest.

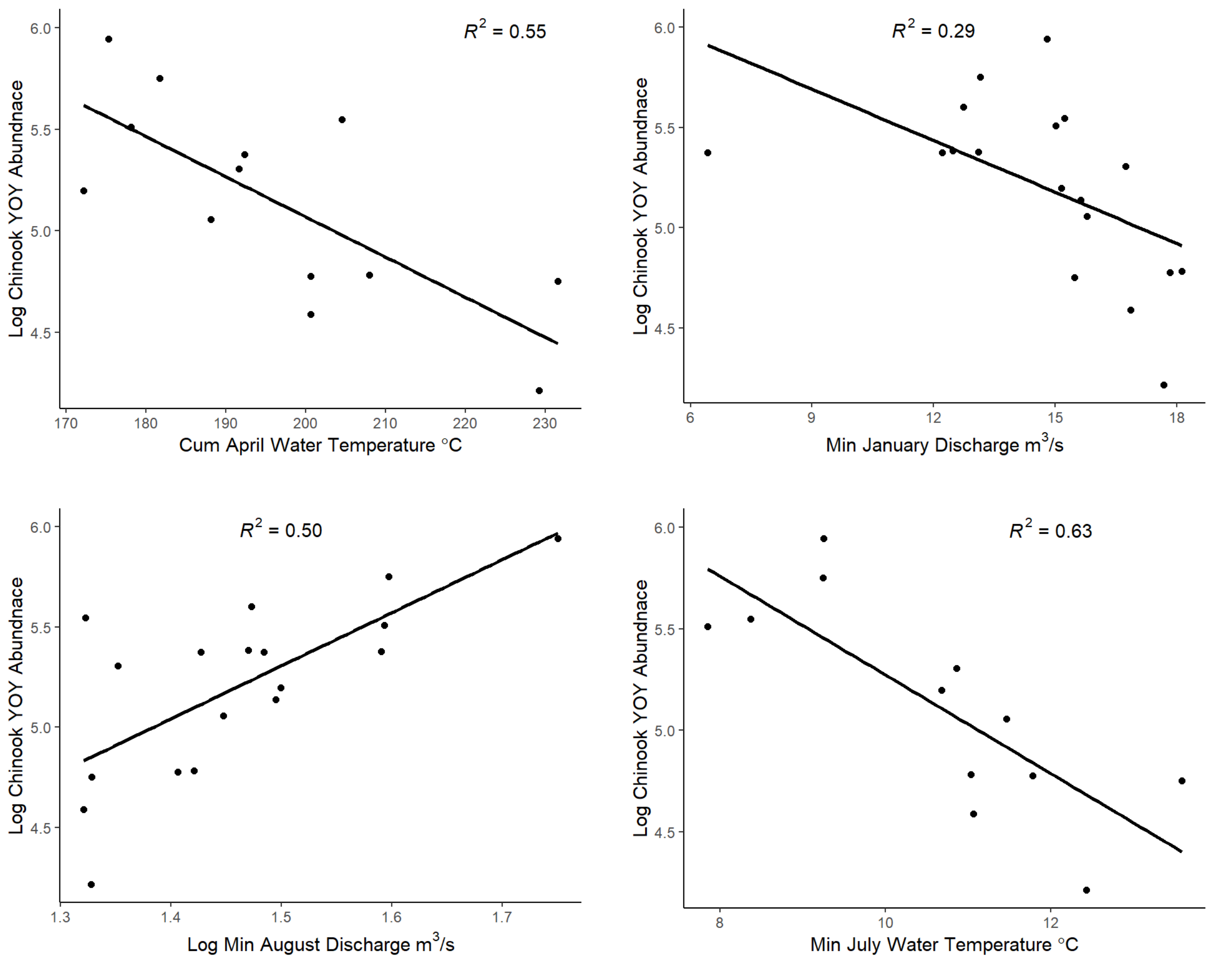

Figure 11. Plots of a subset of significant linear and log-linear relationships between YOY Chinook Salmon abundance and Cheakamus River environmental variables.

3.3.4 Visual assessment of Correlation Coefficients

A total of 50 environmental variables met the assumptions of normality, but after restricting the assessment to variables with a Pearson's correlation coefficients with abundance of >|0.3|, 22 remained. Minimum discharge in August was most closely associated with Chinook Salmon abundance, which agrees with linear modelling results. The heatmap also showed that maximum discharge and discharge variance in July were positively associated with abundance. Taken together, results suggest summer discharge may be an influential environmental variable associated with Chinook Salmon abundance.

Interestingly, several temperature variables from the Spring and Summer significantly predicted abundance but did not cluster with abundance.

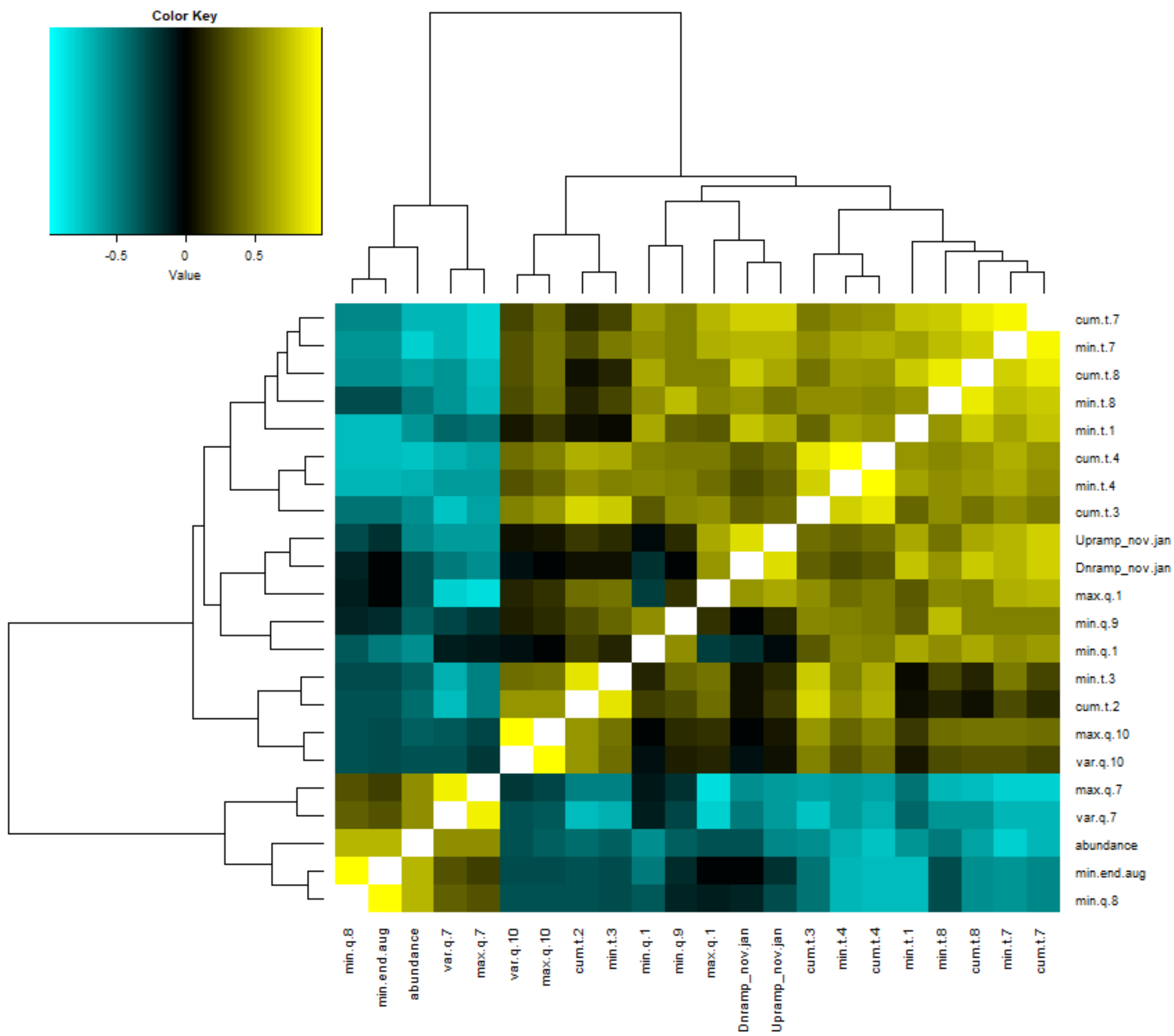

Figure 12. Dendrogram clustering and heatmap of Pearson's correlation coefficients with YOY Chinook Salmon abundance. An r ≥ |0.3| was used as a threshold for variables to be included in the dendrogram. Black lines indicate clusters of related variables based on Euclidian distance. The row and column labels show abbreviated environmental variable names using the naming convention: 'category.variable.month' whereby cum = cumulative, var = variance, min = minimum, max = maximum, upramp = the maximum six hour up ramping rate, dnramp= the maximum six hour down ramping rate, min.end.aug = the discharge from August 15 to August 30, q = discharge, t = temperature, and the month is designated numerically.

4.0 DISCUSSION

Fluctuations in river discharge can have substantial effects on Pacific salmon, the magnitude of which depends on the timing, magnitude, frequency and rate of change in flows (Harnish et al., 2013). Discharge in un-regulated rivers in the south coast of British Columbia generally follow a predictable pattern with large discharge increases occurring in the spring (snow melt) and fall (rainy-season onset), and small discharge fluctuations throughout the year due to rain events. Discharge patterns downstream of hydroelectric facilities may not follow those in un-regulated rivers and may affect spawning, incubation and juvenile rearing in Pacific salmon (Malcolm et al., 2012; Young et al., 2011; Zeug et al., 2014).

The Cheakamus River WUP process identified key uncertainties regarding how anadromous salmonids respond to changes in discharge resulting from operations at Daisy Dam (BC Hydro 2007). In CMSMON1a, juvenile abundance data were collected for Pink, Chinook, and Coho Salmon to determine how discharge in the Cheakamus River affects juvenile fish production, and whether juvenile productivity changed between the IFR and WUP flow regimes. This report synthesizes data from 2000 to 2019 to answer two specific management questions:

> MQ1. What is the relation between discharge and juvenile salmonid production, productivity, and habitat capacity of the mainstem and major side-channels of the Cheakamus River?
>
> MQ2. Does juvenile salmonid production, productivity, or habitat capacity change following implementation of the WUP flow regime?

4.1 *MQ1: What is the relationship between discharge and juvenile salmonid production, productivity, and habitat capacity of the mainstem and major side-channels of the Cheakamus River?*

We used linear regressions to explore potential relationships between environmental variables (discharge and temperature) and juvenile fish abundance in the Cheakamus River. Accounting for all potential environmental variables was beyond the scope of CMSMON1a, and the significant relationships identified between juvenile abundance and discharge and temperature metrics may be confounded by additional unaccounted for variables or multicollinearity. Nonetheless, the significant relationships identified in analyses are supported by published literature regarding environmental predictors of juvenile abundance and salmonid productivity in the Pacific Northwest and/or are biologically related to mechanisms known to affect salmon at different life stages, as discussed below.

Pink salmon

Pink Salmon abundance was not strongly associated with recorded environmental variables in the Cheakamus River, and results did not change with the addition of 2018 data. Late winter discharge (February and March) may be an influential variable, and higher minimum discharge during outmigration may be positively associated with YOY Pink Salmon abundance. Increased winter discharge has been shown to affect the migration timing and survival of juvenile Chinook Salmon (Aurthod et al. 2010; Zeug et al., 2014) and other species of Pacific salmon in the Columbia River Basin (Čada et al., 1997).

Increases in discharge during the late winter/early spring are common in the Cheakamus River due to snow melt and spring storm events. These events may offer beneficial migration conditions for juvenile salmonids. However, without corroborative adult abundance data it was not possible to determine if this relationship is in fact real, or if higher WUP winter flows happened to coincide with higher adult abundance in WUP years.

Linear modelling results also indicated that high minimum discharge in November during fry emergence was negatively associated with abundance. Although high discharge and ramping events have been shown to negatively effect newly emerged salmon fry (Puffer et al., 2017; DeVries, 1997), our results were heavily influenced by a single high data point from 2004 (the year following a 100-year flooding event) and should be considered cautiously. Pink Salmon begin emerging in late November in the Cheakamus River, during which time they inhabit gravel bars and river margins and are therefore sensitive to stranding in areas that dewater during sudden changes in discharge (Puffer et al., 2017). Incubating eggs and juvenile salmonids are also sensitive to discharges that are high enough to mobilize small substrate particles which can result in scouring of redds (DeVries, 1997). The frequency of extreme storm events that result in high discharges in November in the Pacific Northwest is predicted to increase with climate change (Tohver et al., 2014), possibly restricting the scope of flow management during this period. The relationship between fall stranding and emerging Pink Salmon should be further explored in future monitoring programs to determine whether fall flow management, such as ramping rate, could benefit abundance.

Coho Salmon

No strong relationships between environmental variables and Coho Salmon smolt abundance were identified using linear models. A weak finding from variable clustering was a positive correlation between higher discharge in March and smolt abundance. Like Pink Salmon, increased discharge during the migration period may improve survival of Coho Salmon juveniles (Čada et al. 1997; Zeug et al. 2014). High winter discharge events, which are generally representative of natural hydrological events in the Cheakamus River, have been associated with an earlier smolt migration to the marine environment in other systems (Rebenack et al., 2015). It is unlikely that Daisy Dam operations could be changed to mitigate the effects of high winter discharge on the spring Coho migration abundance or migration timing due to the small storage capacity of the reservoir. Additionally, it may be beneficial for some smolts to leave in the fall or winter to provide better foraging opportunities for juveniles remaining in the river over winter (Chapman, 1962). The relationships between survival and migration timing in Coho Salmon are not well understood at present.

Although several potential variables were indicated to influence Coho salmon abundance, the associations and regressions were not strong and should be considered cautiously. Identifying relationships between environmental variables and Coho Salmon smolt abundance is particularly challenging because juveniles spend approximately 18 months rearing in the Cheakamus River. Juveniles are exposed to environmental conditions throughout the year, making it difficult to identify influential variables. Signals from individual seasons may be diminished by summarizing data over a long-time period. Similarly, because there are multiple life history stages for Coho Salmon in freshwater (incubating eggs, YOY, parr, smolt), relationships to environmental variables in one life stage may be confounded or masked by interactions between multiple factors over other life history stages.

Recent research suggests Coho Salmon juvenile life history is not fixed to a stream-type with marine migrations beginning one to two years after emergence. Spring and fall emigrant YOY and yearling smolts are increasingly being found to contribute to adult returns across the extent of their North American range (Koski, 2009; Bennett et al., 2015). Given the improved understanding of variable juvenile life histories in Coho Salmon, further research would be required to examine freshwater survival in the Cheakamus River to fully understand the relationship between winter discharge and Coho Salmon abundance and productivity.

Chinook Salmon

Our results indicate that higher and more variable discharges in July and August may be associated with increased juvenile Chinook Salmon abundance. Summer discharge and water temperature were also identified to potentially influence Chinook Salmon abundance in the 2017 analysis (Lingard et al. 2018). August is the peak spawning period for summer Chinook Salmon in the Squamish River watershed, including the Cheakamus River. The WUP allows for a discretionary decrease from 38 to 20 $m^3 s^{-1}$ on August 15, unless directed by the Water Comptroller to maintain 38 $m^3 s^{-1}$ for recreational purposes. For large-bodied fish like Chinook Salmon, higher discharges (38 $m^3 s^{-1}$) during the adult spawning period may provide better migration conditions and opportunities for spawning in habitat that is too shallow at 20 $m^3 s^{-1}$. Moderate peaks in discharge during the spawning period have been documented to influence the timing of pre-spawning river entry by adult Atlantic Salmon (Tetzlaff et al., 2008) as well as the presence of large adults in shallow water spawning habitats (Malcolm et al., 2012).

We also found that high water temperatures during the summer adult spawning period (July and August) were negatively associated with juvenile Chinook Salmon abundance. Water temperatures experienced by sexually maturing adult salmonids can influence gamete production (Pankhurst et al. 1996), gamete quality (Lahnsteiner and Leitner 2013) and survival (Teffer et al. 2018). Embryo development rates in salmonids are also influenced by water temperature, with faster development occurring at higher temperatures (Fuhrman et al., 2018). In the Cheakamus River, most of the thermal units required for embryo development are likely obtained in August through October before water temperatures drop over the winter. High water temperatures during August may affect emergence and migration timing of YOY Chinook Salmon the following spring. Murray and McPhail (1988) found Chinook Salmon juveniles would emerge 115 days (4 months) after spawning if reared at 8°C but juveniles reared at 14°C would emerge at 63 days (2 months). In November of 2019, emergent Chinook Salmon were observed during a field survey in the Cheakamus River (Instream Fisheries Research, unpublished data). A difference of a few degrees in daily water temperature in August likely influences when juvenile Chinook Salmon emerge and begin their downstream migration.

Water temperature in August was negatively correlated with minimum August discharge. A previous WUP monitor found that water temperatures downstream of Daisy Dam are influenced by dam operations, however the authors concluded that the effects of dam operations on water temperature are mitigated in the anadromous reach by inflows from tributaries (Rubble and Culliton Creeks) (McAdam 2001). However, the temperature study did not explore how dam releases might be used to mitigate the effects of solar radiation and air temperature on water temperature. The negative correlation between August minimum water temperature and August minimum discharge suggests that higher discharges may result in lower water temperatures. Taken together, the relationship between discharge, temperature and emergence timing is a critical area for further research on Cheakamus River Chinook Salmon.

April water temperature was also identified as a significant variable potentially affecting Chinook Salmon abundance. Neither spring nor summer temperature variables that significantly predicted abundance clustered with abundance in the correlation coefficient matrix. However, there is considerable biological support for negative effects of high water temperatures during the summer adult spawning period (July and August) on juvenile Chinook Salmon abundance, as discussed, but little biological evidence to support the significant relationship detected with water temperature in April. One possibility is that higher water temperature in April may affect late emerging juveniles (from the smaller proportion of fish spawning in October), thereby influencing migration timing (Jonsson and Ruud-Hansen 1985; Marine and Cech 2004; Beakes et al. 2014).

4.2 *MQ2 Did juvenile salmonid production, productivity, or habitat capacity change following implementation of the WUP flow regime?*

MQ2 was difficult to address because there were few pre-WUP data points, the time series was short relative to the length of salmon life cycles, and abundances were highly variable during both flow regimes. Pacific salmon have complex life histories spanning freshwater and marine environments with highly variable annual abundances among and within populations. The ability to detect trends in salmon abundance using data from a single life history stage within the time frame of most monitoring programs (< 20 years) is often constrained by high variability in abundance and the multitude of potential environmental factors influencing survival (Korman and Higgins 1997; Ham and Pearsons 2000; Parnell et al., 2003; Wagner et al., 2013). The power of the t-tests used in this analysis to detect changes in salmon abundance within the time frame of this monitoring program were predicted to be below 0.80 in both pre study simulations and at the five-year review; therefore, a possibility of a type 2 error (i.e., failure to detect a difference in means) is beyond the recommended threshold (Parnell et al., 2003; Melville & McCubbing 2012).

Assessing trends in Pink Salmon abundance was particularly challenging due to their alternate year presence in watersheds. Compared to other species, the absence of Pink Salmon in some years extends the time required to detect a change in abundance (Melville & McCubbing 2012). Although it was not possible to test a difference in mean abundance between IFA and WUP flow treatments due to low sample size, mean Pink salmon abundance under the WUP flow treatment was 20-fold greater than the mean abundance under the IFA flow treatment. It is unlikely that this increase was in response to WUP flows as the trend has been observed in multiple odd-year Pink Salmon populations on both sides of the Pacific Ocean (Irvine et al., 2014; see spawner abundance data in Appendix E). The increasing trend of odd-year Pink Salmon observed during the WUP flow treatment likely resulted from favourable environmental conditions in the Pacific Ocean (Irvine et al., 2014).

We did not find a significant difference in mean abundance of Coho Salmon smolts between flow treatments. The Coho dataset was the most robust because the annual study period captured the entire run (Lingard et al., 2016) and abundance estimates were generated for all years between 2001 and 2017. Although it is difficult to tease out the effects of the 2003 flood and 2005 spill from other confounding factors, these two events likely had significant impacts on Coho abundance during the IFA flow period (Melville & McCubbing 2006). It is possible that the difference in abundance between IFA and WUP flow treatments would have been greater had these two events not occurred during the IFA flow treatment.

The range of YOY Chinook Salmon abundances between IFA and WUP flow treatments was similar and mean abundances did not differ significantly. However, there were notable limitations in the Chinook Salmon abundance data. The precision of the estimates was low due to sparse catches in all years. The migration of YOY Chinook consistently started prior to the trapping period, resulting in incomplete estimates of annual abundance (Lingard et al., 2016). Chinook Salmon also display a range of juvenile rearing and emigration strategies not enumerated by the RST trapping program which further limit the ability to detect changes in juvenile Chinook Salmon abundance in relation to the WUP flow treatment.

4.3 Remaining Uncertainties

CMSMON1a collected juvenile abundance data for Pink, Coho and Chinook salmon, but without adult abundance data we could not determine productivity (the relationship between recruits and spawning adults). From incubation to outmigration, juvenile salmon abundance can be affected by discharge (Zeug et al., 2014; Arthaud et al., 2010), water temperature (Fuhrman et al., 2018), predator abundance (Walsworth & Schindler, 2016), watershed productivity (Wipfli et al., 2003), and density dependent survival among juvenile salmonids (Einum and Nislow 2005). The conditions experienced by spawning adults, including marine productivity and fishing pressure, also have a large effect on juvenile abundance. Without knowing productivity, annual changes in juvenile abundance may be incorrectly attributed to freshwater variables such as discharge, when in fact they were driven by adult returns (MacKenzie et al., 2013).

Our ability to answer the management questions was also affected by the relatively short time series of data being considered. The time required to detect population level effects depends on the life cycle and life history of the species (Babcock et al., 2010; Peterman 1990); with longer time periods required for longer lived species with highly variable annual abundances. In the context of salmon population dynamics, the study duration of CMSMON1a was relatively short compared to the length of a salmon life cycle resulting in few generations being monitored over the duration of the project. Effects on abundance will take time to appear given only one life stage was monitored. That is, if the WUP influenced abundance of every cohort from 2007 onwards, the effect to subsequent generations would be attenuated as these fish return to spawn. With few generations monitored under each treatment, the time period monitored may not have been sufficient to detect WUP effects. Pink Salmon life cycles are two years in duration. A three-year life history dominates in south coast Coho Salmon populations (Beamish et al., 2004), while Chinook Salmon return to spawn between ages 3 and 6. Taking this into consideration, one to three generations were monitored under the IFA flow treatment and two to five generations were monitored under the WUP flow treatment. Having few observations resulted in some years having a high leverage effect on regression relationships. In some cases, a single influential point may make a non-significant relationship appear significant, while in other cases they may represent the upper and/or lower limits of the true relationship (and additional data are required to verify the relationship).

A power analysis was completed for the Cheakamus River during the design of the Cheakamus WUP monitoring program. In this analysis Parnell et al. (2003) predicted 12 years prior and 12 years post implementation of the WUP were required to detect a -25% change in Coho smolt abundance with statistical power of 69%. For Coho Salmon, which has the most robust data set with the least amount of variation of all three species, only 6 data points were collected under the IFA and 10 under the WUP. For Pink Salmon, only 3 data points pre and 6 data points post WUP were collected. For Chinook Salmon 5 and 13 data points were collected under pre and post WUP conditions, respectively.

Chinook Salmon in the Cheakamus River

A substantial uncertainty remains regarding the effects of the WUP flow treatment on juvenile Chinook Salmon. Chinook Salmon are documented to exhibit a wide range of juvenile life history strategies (Miller et al., 2010). In the last five years, additional focus has been paid to evaluating juvenile life histories of Chinook Salmon in the Pacific Northwest (Volk et al. 2015; Bourrete et al. 2016). Prior to new discoveries, juvenile Chinook Salmon have been described as having one of three life-history types that emigrate as newly emergent YOY (30-50 mm), in the fall as larger (< 60 mm) YOY, or the following spring as yearling smolts (> 80 mm). However, increasingly it appears Chinook Salmon may leave on a continuous basis from watersheds with juveniles rearing in other tributaries of the same river or in estuaries for variable periods of time (Bourrete et al. 2016). Discharge has been shown to affect the age at migration in juvenile Chinook Salmon in other rivers in North America (Zeug et al., 2014).

Determining whether the WUP or other flow treatments affect the productivity of Chinook Salmon juveniles in the Cheakamus River would require a study of multiple life history strategies and survival rates between life history stages, as well as further understanding habitat usage and the contribution of various life history strategies to the over all population. Such an approach has been employed for Chum Salmon and Steelhead trout in CMSMON1b and CMSMON3, respectively. A study of this nature would require a long monitoring period (multiple generations per flow treatments to be compared) to establish a productivity dataset under varying environmental conditions with adequate replication. The amount of time required for adequate statistical power will depend on the variability in annual survival as well as the magnitude of change in abundance stakeholders would like to detect.

Given the potentially constant emigration of Chinook Salmon juveniles, novel approaches that focus on the specific effects of discharge on individual life stages using detailed telemetry to assess movement and muscle activity (Taylor et al. 2012), physiologic correlates of stress (Wagner et al. 2004, Crossin et al. 2008, Cook et al. 2011), or measures of fitness such as fat content (Crossin and Hinch 2005, Cooke et al. 2015) may be necessary to elucidate the effects of operations of the survival and fitness of this species. More targeted studies of fish behaviour and movement may provide answers regarding the effects of operations on salmon in shorter time periods (5-10 years) than monitoring programs focused on abundance and productivity at the population level.

Low abundance of Chinook Salmon from 2014 to 2018 was a trend of concern and was likely related to regional trends of poor ocean survival in the Strait of Georgia (Preikshot et al., 2013). However, Chinook Salmon populations have considerable heterogeneity in abundance and marine survival trends among Strait of Georgia populations and life history types (Ruff et al., 2017). The lack of adult abundance data and variability among populations within the south coast region confound our ability to determine whether Cheakamus River conditions are contributing to the recent low abundance of Chinook Salmon, or whether these effects are due to ocean conditions (Appendix E). Consequently, there is a need to determine if freshwater survival is contributing to the low abundance of Chinook Salmon in the Cheakamus River.

Climate Change

A notable uncertainty is how climate change will affect all species in the watershed and whether the WUP can be modified to help mitigate some of these potential effects. Storm events in the fall and winter months are projected to increase in frequency and magnitude with climate change (Tohver et al., 2017). Additionally, stream water temperatures are forecasted to increase with climate change (Van Vliet et al., 2013) and salmon populations are at risk of extirpation from some watersheds (Crozier 2015).

The relationships presented in this report indicate a sensitivity of juvenile salmon to high winter discharges and high-water temperatures during the summer and fall, which may be exacerbated by climate change. For example, increasing water temperature during the summer and early fall spawning and incubation period is projected to be a significant limiting factor for Chinook Salmon populations as climate change progresses (Honea et al., 2016). Although the Cheakamus River has over 3 kilometers of side-channel habitat to protect from high discharge events in the mainstem, the majority of Pink, Chinook, and Coho salmon juveniles were found to originate out of mainstem habitats in the Cheakamus River. The reliance of Pink, Chinook and Coho salmon on mainstem habitat for rearing and spawning in the Cheakamus River indicates a need to further understanding of habitat use and freshwater survival in these species to support the development of water management solutions to buffer these populations from the potential cumulative effects of climate change and Daisy Dam operations.

6.0 CONCLUSION

We monitored juvenile salmon abundance of Pink, Chinook and Coho salmon in the Cheakamus River for nineteen years between 2001 and 2019. Six years of data were collected prior to implementation of the

Cheakamus WUP flow regime. The monitoring program was implemented to reduce uncertainties surrounding the relationship between juvenile salmon abundance and discharge in the Cheakamus River below Daisy Dam, as outlined in the Cheakamus Water Use Plan 2007.

We were successful in developing relationships between juvenile salmon abundance and discharge in the Cheakamus River to answer MQ1. The relationships developed in this analysis will be informative for how future BC Hydro operations of the Cheakamus River will be determined. For Pink Salmon and Coho Salmon, we found that higher minimum discharges in late winter (February and March) may be associated with higher juvenile abundances. For Chinook Salmon, higher minimum discharges in August may be associated with higher juvenile abundance. Although these findings are significant for management of the Cheakamus, we caution managers to consider the short time duration, and limited scope of the monitoring program to a single life stage.

We could not conclusively determine whether juvenile abundance was different between IFA and WUP flow regimes (MQ2) due to the short monitoring duration and high variability in abundance data. Answering MQ2 was especially problematic for Pink and Chinook Salmon which had highly variable and/or sparse data. For Coho Salmon, annual abundances were relatively consistent among treatments and the 5-year review power analysis indicated statistical power of the t-test would approach 0.70 to detect a 50% change smolt abundance by the end of the WUP monitoring period (Melville and McCubbing 2012). In Coho Salmon we can be reasonably certain there was not a 50% or larger change in abundance since implementation of the WUP. For Pink and Chinook Salmon, the statistical power of t-test is too low to make conclusion regarding the effects of the WUP flow treatment on abundance.

Additional monitoring is required to understand the relationship between discharge and juvenile productivity in the Cheakamus River, and determine how salmon populations may respond to BC Hydro operations. We do not recommend carrying on CMSMON1a according to its current experimental design to address MQ2, as additional years would be unlikely to produce sufficient power to detect differences between IFA and WUP. As previous power analyses indicated, decades of data per flow treatment would be required to detect large (50-75%) changes in abundance. However, if the monitor were continued and the scope was widened to include adult abundance and spawning success, other modeling approaches such as stock-recruitment, mixed effect or multiple linear regression modelling could be used to determine the effects of flow co-variates on salmon productivity. A widening of scope would still require long term (> 10 years) monitoring to develop a productivity dataset robust enough for modeling.

An alternative approach to determining flow effects on salmonids would be to undertake targeted hypothesis testing of specific operations on fish behaviour or survival using telemetry or more novel methods to assess, for example, fish condition, activity, and/or stress in individual life stages in the Cheakamus River. The more targeted approach may obtain results in shorter time periods than monitoring population level effects (productivity), but the time required, and strength of analyses will ultimately depend on the variability within and among treatments. With any new monitoring program to assess fish response to dam operations, a thorough evaluation of field methodologies, replicates required for statistical power, and proposed statistical analyses should be completed by prior to implementation. Salmon life histories in the Cheakamus River are complex and monitoring their response to environmental alteration is difficult; however, understanding how this group of species respond to flow management is important for managing aquatic ecosystems now and in the future.

7.0 REFERENCES

Arthaud, D. L., Greene, C. M., Guilbault, K., & Morrow, J. V. (2010). Contrasting life-cycle impacts of stream flow on two Chinook salmon populations. Hydrobiologia, 655(1), 171–188. https://doi.org/10.1007/s10750-010-0419-0

Babcock, R. C., Shears, N. T., Alcala, A. C., Barrett, N. S., Edgar, G. J., Lafferty, K. D., … Russ, G. R. (2010). Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. Proceedings of the National Academy of Sciences, 107(43), 18256–18261. https://doi.org/10.1073/pnas.0908012107

BC Hydro. (2005). Cheakamus Project Water Use Plan. 38p.

BC Hydro. (2007). Cheakamus Project Water Use Plan Monitoring Program Terms of Reference: Cheakamus River juvenile Salmon Outmigration Enumeration Monitoring. 19p.

Beakes, M. P., Sharron, S., Charish, R., Moore, J. W., Satterthwaite, W. H., Sturm, E., … Mangel, M. (2014). Using scale characteristics and water temperature to reconstruct growth rates of juvenile steelhead Oncorhynchus mykiss. Journal of Fish Biology, 84(1), 58–72. https://doi.org/10.1111/jfb.12254

Beamish, R. J., Mahnken, C., & Neville, C. M. (2004). Evidence That Reduced Early Marine Growth is Associated with Lower Marine Survival of Coho Salmon. Transactions of the American Fisheries Society, 133(1), 26–33. https://doi.org/10.1577/T03-028

Beer, W. N., & Anderson, J. J. (2001). Effect of spawning day and temperature on salmon emergence: interpretations of a growth model for Methow River chinook. Canadian Journal of Fisheries and Aquatic Sciences, 58(5), 943–949. https://doi.org/10.1139/cjfas-58-5-943

Bennett, T. R., Roni, P., Denton, K., McHenry, M., & Moses, R. (2015). Nomads no more: early juvenile coho salmon migrants contribute to the adult return. Ecology of Freshwater Fish, 24(2), 264–275. https://doi.org/10.1111/eff.12144

Bonner, S. J., & Schwarz, C. J. (2011). Smoothing population size estimates for time-stratified mark-recapture experiments using Bayesian P-splines. Biometrics, 67(4), 1498–1507. https://doi.org/10.1111/j.1541-0420.2011.01599.x

Bourret, S. L., Caudill, C. C., & Keefer, M. L. (2016). Diversity of juvenile Chinook salmon life history pathways. Reviews in Fish Biology and Fisheries, 26(3), 375–403. https://doi.org/10.1007/s11160-016-9432-3

Bradford, M. J., Taylor, G. C., Allan, J. A., & Higgins, P. S. (1995). An Experimental Study of the Stranding of Juvenile Coho Salmon and Rainbow Trout during Rapid Flow Decreases under Winter Conditions. North American Journal of Fisheries Management, 15(2), 473–479. https://doi.org/10.1577/1548-8675(1995)015<0473:AESOTS>2.3.CO;2

Bradford, M. J. (1997). An experimental study of stranding of juvenile salmonids on gravel bars and in sidechannels during rapid flow decreases. Regulated Rivers: Research & Management: An International Journal Devoted to River Research and Management, *13*(5), 395-401.

Bradford, M. J., Taylor, G. C., & Allan, J. A. (1997). Empirical Review of Coho Salmon Smolt Abundance and the Prediction of Smolt Production at the Regional Level. Transactions of the American Fisheries Society, 126(1), 49–64. https://doi.org/10.1577/1548-8659(1997)126<0049:EROCSS>2.3.CO;2

Čada, G. F., Deacon, M. D., Mitz, S. V., & Bevelhimer, M. S. (1997). Effects of water velocity on the survival of downstream-migrating juvenile salmon and steelhead: A review with emphasis on the Columbia river basin. Reviews in Fisheries Science, 5(2), 131–183. https://doi.org/10.1080/10641269709388596

Chapman, D. W. (1962). Aggressive Behavior in Juvenile Coho Salmon as a Cause of Emigration. Journal of the Fisheries Research Board of Canada, 19(6), 1047–1080. https://doi.org/10.1139/f62-069

Cook, K. V., McConnachie, S. H., Gilmour, K. M., Hinch, S. G., & Cooke, S. J. (2011). Fitness and behavioral correlates of pre-stress and stress-induced plasma cortisol titers in pink salmon (Oncorhynchus gorbuscha) upon arrival at spawning grounds. *Hormones and Behavior*, *60*(5), 489-497.

Connor, E.J., & Pflug, D.E. (2004). Changes in the Distrubtuion and Density of Pink, Chum, and Chinook Salmon Spawning in the upper Skagit River in Response to Flow Management Measures. North American Journal of Fisheries Management, 24(3), 835-852.

Crossin, G. T., & Hinch, S. G. (2005). A nonlethal, rapid method for assessing the somatic energy content of migrating adult Pacific salmon. Transactions of the American Fisheries Society, 134(1), 184-191.

Crossin, G. T., Hinch, S. G., Cooke, S. J., Welch, D. W., Patterson, D. A., Jones, S. R. M., ... & Van Der Kraak, G. (2008). Exposure to high temperature influences the behaviour, physiology, and survival of sockeye salmon during spawning migration. Canadian Journal of Zoology, 86(2), 127-140.

Crozier, L. G., Zabel, R. W., Hockersmith, E. E., & Achord, S. (2010). Interacting effects of density and temperature on body size in multiple populations of Chinook salmon. Journal of Animal Ecology, 79(2), 342-349.

Crozier, L. (2015). Impacts of Climate Change on Salmon of the Pacific Northwest. A Review of the Scientific Literature Published in 2014. NOAA report. 42p.

DeVries, P. (1997). Riverine salmonid egg burial depths: review of published data and implications for scour studies. Canadian Journal of Fisheries and Aquatic Sciences, 54(8), 1685–1698. https://doi.org/10.1139/f97-090

Einum, S., & Nislow, K. H. (2005). Local-scale density-dependent survival of mobile organisms in continuous habitats: an experimental test using Atlantic salmon. Oecologia, 143(2), 203–210. https://doi.org/10.1007/s00442-004-1793-y

Fell, C., Middleton, C., Meville, C. C., & Burnett, N. (2018 in progress). Cheakamus River Chum Salmon Escapement Monitoring and Mainstem Spawning Groundwater Survey Final Report for BC Hydro CMSMON 1b 2018.

Field, A., & Hole, G. (2002). *How to design and report experiments*. Sage.

Freeman, Mary C., Bowen, Zachary H., Bovee, Ken D., Irwin, E. R. (2001). Flow and Habitat Effects on Juvenile Fish Abundance in Natural and Altered Flow Regimes. Ecological Applications, 11(1), 179–190.

Fraser River Panel. (2016). Report of the Fraser River Panel to the Pacific Salmon Commission on the 2015 Fraser Sockeye and Pink Salmon Fishing Season. DFO report. 77p.

Fuhrman, A. E., Larsen, D. A., Steel, E. A., Young, G., & Beckman, B. R. (2018). Chinook salmon emergence phenotypes: Describing the relationships between temperature, emergence timing and condition factor in a reaction norm framework. Ecology of Freshwater Fish, 27(1), 350–362. https://doi.org/10.1111/eff.12351

Geist, D. R., Abernethy, C. S., Hand, K. D., Cullinan, V. I., Chandler, J. A., & Groves, P. A. (2006). Survival, Development, and Growth of Fall Chinook Salmon Embryos, Alevins, and Fry Exposed to Variable Thermal and Dissolved Oxygen Regimes. Transactions of the American Fisheries Society, 135(6), 1462–1477. https://doi.org/10.1577/T05-294.1

Gibbins, C. N., Moir, H. J., Webb, J. H., & Soulsby, C. (2002). Assessing discharge use by spawning Atlantic salmon: A comparison of discharge electivity indices and PHABSIM simulations. River Research and Applications, 18(4), 383–395. https://doi.org/10.1002/rra.685

Goniea, T. M., Keefer, M. L., Bjornn, T. C., Peery, C. A., Bennett, D. H., & Stuehrenberg, L. C. (2006). Behavioral Thermoregulation and Slowed Migration by Adult Fall Chinook Salmon in Response to High Columbia River Water Temperatures. Transactions of the American Fisheries Society, 135(2), 408–419. https://doi.org/10.1577/T04-113.1

Goode, J. R., Buffington, J. M., Tonina, D., Isaak, D. J., Thurow, R. F., Wenger, S., … Soulsby, C. (2013). Potential effects of climate change on streambed scour and risks to salmonid survival in snow-dominated mountain basins. Hydrological Processes, 27(5), 750–765. https://doi.org/10.1002/hyp.9728

Ham, K. D., & Pearsons, T. N. (2000). Can reduced salmonid population abundance be detected in time to limit management impacts? Canadian Journal of Fisheries and Aquatic Sciences, 57(1), 17–24. https://doi.org/10.1139/cjfas-57-1-17

Hard, J. J., Gross, M. R., Heino, M., Hilborn, R., Kope, R. G., Law, R., & Reynolds, J. D. (2008). SYNTHESIS: Evolutionary consequences of fishing and their implications for salmon. Evolutionary Applications, 1(2), 388–408. https://doi.org/10.1111/j.1752-4571.2008.00020.x

Harper, D., & Wilson, G. (2007). Pilot reach bank secured large wood restoration project 2007 –Final Report. https://doi.org/HR-Cheak_PCR-CN-2007.

Hodgson, S., & Quinn, T. P. (2002). The timing of adult sockeye salmon migration into fresh water: adaptations by populations to prevailing thermal regimes. Canadian Journal of Zoology, 80(3), 542–555. https://doi.org/10.1139/z02-030

Honea, J. M., McClure, M. M., Jorgensen, J. C., & Scheuerell, M. D. (2016). Assessing freshwater life-stage vulnerability of an endangered Chinook salmon population to climate change influences on stream habitat. Climate Research, 71(2), 127–137. https://doi.org/10.3354/cr01434

Irvine, J. R. (1986). Effects of varying discharge on the downstream movement of salmon fry, Oncorhynchus tshawytscha Walbaum. Journal of Fish Biology, 28(1), 17–28. https://doi.org/10.1111/j.1095-8649.1986.tb05137.x

Irvine, J. R., Michielsens, C. J. G., O'Brien, M., White, B. A., & Folkes, M. (2014). Increasing Dominance of Odd-Year Returning Pink Salmon. Transactions of the American Fisheries Society, 143(4), 939–956. https://doi.org/10.1080/00028487.2014.889747

Jonsson, B., & Ruud-Hansen, J. (1985). Water Temperature as the Primary Influence on Timing of Seaward Migrations of Atlantic Salmon (Salmo salar) Smolts. Canadian Journal of Fisheries and Aquatic Sciences, 42(3), 593–595. https://doi.org/10.1139/f85-076

Korman, J., & Higgins, P. S. (1997). Utility of escapement time series data for monitoring the response of salmon populations to habitat alteration. Canadian Journal of Fisheries and Aquatic Sciences, 54(9), 2058–2067. https://doi.org/10.1139/cjfas-54-9-2058

Korman, J., Kaplinski, M., & Melis, T. S. (2011). Effects of Fluctuating Flows and a Controlled Flood on Incubation Success and Early Survival Rates and Growth of Age-0 Rainbow Trout in a Large Regulated River. Transactions of the American Fisheries Society, 140(2), 487–505. https://doi.org/10.1080/00028487.2011.572015

Korman, J., & Schick, J. (inprogress). CMSMON-3 Cheakamus River Steelhead Adult Abundance, Fry Emergence-Timing, and Juvenile Habitat Use Abundance Monitoring Final Report

Koski, K. V. (2009). The Fate of Coho Salmon Nomads: The Story of an Estuarine-Rearing Strategy Promoting Resilience. Ecology and Society, 14(1), art4. https://doi.org/10.5751/ES-02625-140104

Lahnsteiner, F., & Leitner, S. (2013). Effect of temperature on gametogenesis and gamete quality in brown trout, Salmo trutta. *Journal of Experimental Zoology Part A: Ecological Genetics and Physiology*, *319*(3), 138-148.

Lingard, S. (2015). Squamish River Estuary Juvenile Chinook Monitoring Program, annual data report 2015. Report Prepared for Squamish River Watershed Society. 35p

Lingard, S., Melville, C. ., & McCubbing, D. J. F. (2016). Cheakamus River Juvenile Salmonid Outmigration Assessment, Annual Data Report 2016. Report prepared for BC Hydro. 81p.

MacKenzie, B. R., Myers, R. A., & Bowen, K. G. (2003). Spawner-recruit relationships and fish stock carrying capacity in aquatic ecosystems. Marine Ecology Progress Series. Inter-Research Science Center. https://doi.org/10.2307/24866486

Malcolm, I. A., Gibbins, C. N., Soulsby, C., Tetzlaff, D., & Moir, H. J. (2012). The influence of hydrology and hydraulics on salmonids between spawning and emergence: implications for the management of flows in regulated rivers. Fisheries Management and Ecology, 19(6), 464–474. https://doi.org/10.1111/j.1365-2400.2011.00836.x

Marine, K. R., & Cech, J. J. (2004). Effects of High Water Temperature on Growth, Smoltification , and Predator Avoidance in Juvenile Sacramento River Chinook Salmon. North American Journal of Fisheries Management, 5947(24), 198–210. https://doi.org/10.1577/M02-142

Mattison, J., Nowlan, L., Lebel, M., Orr, C. (2014). Water for Power, Water for Nature: The Story of BC Hydro's Water Use Planning Program. Vancouver: WWF Canada. 56p.

McAdam, S. (2001). Water Temperature Measurements on the Cheakamus River—Data Report: June 1999 to December 2000. Report prepared for B.C. Hydro.

McCubbing, D. J. F., Melville, C. C., Foy, M., & Wilson, G. (2006). Assessment of the CN sodium hydroxide spill August 5th, 2005 on the fish populations of the Cheakamus River. Report prepared for Ministry of Environment and Cheakamus Ecological Recovery Technical Committee. 131p.

Melville, C. C., & McCubbing, D. J. F. (2001). Cheakamus River Project Water Use Plan Juvenile Salmonid Outmigration Enumeration, Spring 2001. Report prepared for BC Hydro. 94p.

Melville, C. C., & McCubbing, D. J. F. (2012). Cheakamus River Project Water Use Plan Juvenile Salmonid Outmigration Enumeration, Summary Report 2001-2012. Report prepared for BC Hydro. 94p.

Miller, J. A., Gray, A., & Merz, J. (2010). Quantifying the contribution of juvenile migratory phenotypes in a population of Chinook salmon Oncorhynchus tshawytscha. Marine Ecology Progress Series, 408, 227–240. https://doi.org/10.3354/meps08613

Muñoz, N. J., Farrell, A. P., Heath, J. W., & Neff, B. D. (2014). Adaptive potential of a Pacific salmon challenged by climate change. Nature Climate Change, 5(2), 163–166. https://doi.org/10.1038/nclimate2473

Murchie, K. J., Hair, K. P. E., Pullen, C. E., Redpath, T. D., Stephens, H. R., & Cooke, S. J. (2008). Fish response to modified flow regimes in regulated rivers: research methods, effects and opportunities. River Research and Applications, 24(2), 197–217. https://doi.org/10.1002/rra.1058

Murray, C. B., & McPhail, J. D. (1988). Effect of incubation temperature on the development of five species of Pacific salmon (Oncorhynchus) embryos and alevins. Canadian Journal of Zoology, 66(1), 266–273. https://doi.org/10.1139/z88-038

Pacific Salmon Foundation Joint Chinook Technial Comittee, P. S. C. J. C. T. C. (2016). Annual Report of Catch and Escapement for 2015. 225p

Parnell, I.J., Marmorek, D.R., Lister, B., Korman, J. (2003). Cheakamus Water Use Plan : Quantitative evaluation of the statistical and cost performance of alternative. 93p.

Peterman, R. M. (1990). Statistical Power Analysis can Improve Fisheries Research and Management. Canadian Journal of Fisheries and Aquatic Sciences, 47(1), 2–15. https://doi.org/10.1139/f90-001

Preikshot, D., Beamish, R. J., & Neville, C. M. (2013). A dynamic model describing ecosystem-level changes in the Strait of Georgia from 1960 to 2010. Progress in Oceanography, 115, 28–40.

https://doi.org/10.1016/J.POCEAN.2013.05.020

Puffer, M., Berg, O. K., Einum, S., Saltveit, S. J., & Forseth, T. (2017). Energetic Consequences of Stranding of Juvenile Atlantic Salmon (Salmo salar L.). Journal of Water Resource and Protection, 9, 163–182. https://doi.org/10.4236/jwarp.2017.92012

Razali, N. M., & Wah, Y. B. (2011). Power comparisons of Shapiro-Wilk, Kolmogorov-Smirnov, Lilliefors and Anderson-Darling tests. Journal of Statistical Modeling and Analytics, 2(1), 21–33. Retrieved from http://www.de.ufpb.br/~ulisses/disciplinas/normality_tests_comparison.pdf

Rebenack, J. J., Ricker, S., Anderson, C., Wallace, M., & Ward, D. M. (2015). Early Emigration of Juvenile Coho Salmon: Implications for Population Monitoring. Transactions of the American Fisheries Society, 144(1), 163–172. https://doi.org/10.1080/00028487.2014.982258

Rieman, B. E., & Beamesderfer, R. C. (1990). Dynamics of a Northern Squawfish Population and the Potential to Reduce Predation on Juvenile Salmonids in a Columbia River Reservoir. North American Journal of Fisheries Management, 10(2), 228–241. https://doi.org/10.1577/1548-8675(1990)010<0228:DOANSP>2.3.CO;2

Ruff, C. P., Anderson, J. H., Kemp, I. M., Kendall, N. W., Mchugh, P. A., Velez-Espino, A., … Rawson, K. (2017). Salish Sea Chinook salmon exhibit weaker coherence in early marine survival trends than coastal populations. Fisheries Oceanography, 26(6), 625–637. https://doi.org/10.1111/fog.12222

Sandercock, F. K. (1991). Life History of Coho Salmon (Oncorhynchus kisutch). In C. Groof & L. Margolis (Eds.), Pacific Salmon Life Histories (pp. 397–445).

Smith, G. W., Smith, I. P., & Armstrong, S. M. (1994). The relationship between river flow and entry to the Aberdeenshire Dee by returning adult Atlantic salmon. Journal of Fish Biology, 45(6), 953–960. https://doi.org/10.1111/j.1095-8649.1994.tb01065.x

Sykes, G. E., Johnson, C. J., & Shrimpton, J. M. (2009). Temperature and Flow Effects on Migration Timing of Chinook Salmon Smolts. Transactions of the American Fisheries Society, 138(6), 1252–1265. https://doi.org/10.1577/T08-180.1

Tetzlaff, D., Gibbins, C., Bacon, P. J., Youngson, A. F., & Soulsby, C. (2008). Influence of hydrological regimes on the pre-spawning entry of Atlantic salmon (Salmo salar L.) into an upland river. River Research and Applications, 24(5), 528–542. https://doi.org/10.1002/rra.1144

Tetzlaff, D., Soulsby, C., Youngson, A. F., Gibbins, C., Bacon, P. J., Malcolm, I. A., & Langan, S. (2005). Variability in stream discharge and temperature: a preliminary assessment of the implications for juvenile and spawning Atlantic salmon. Hydrology and Earth System Sciences, 9(3), 193–208. https://doi.org/10.5194/hess-9-193-2005

Tohver, I. M., Hamlet, A. F., & Lee, S.-Y. (2014). Impacts of 21st-Century Climate Change on Hydrologic Extremes in the Pacific Northwest Region of North America. JAWRA Journal of the American Water Resources Association, 50(6), 1461–1476. https://doi.org/10.1111/jawr.12199

Van Vliet, M. T. H., Franssen, W. H. P., Yearsley, J. R., Ludwig, F., Haddeland, I., Lettenmaier, D. P., & Kabat, P. (2013). Global river discharge and water temperature under climate change. Global Environmental Change, 23(2), 450–464. https://doi.org/10.1016/J.GLOENVCHA.2012.11.002

Volk, E. C., Bottom, D. L., Jones, K. K., & Simenstad, C. A. (2010). Reconstructing Juvenile Chinook Salmon Life History in the Salmon River Estuary, Oregon, Using Otolith Microchemistry and Microstructure. Transactions of the American Fisheries Society, 139(2), 535–549. https://doi.org/10.1577/T08-163.1

Wagner, T., & Congleton, J. L. (2004). Blood chemistry correlates of nutritional condition, tissue damage, and stress in migrating juvenile chinook salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences, *61*(7), 1066-1074.

Wagner, G. N., Hinch, S. G., Kuchel, L. J., Lotto, A., Jones, S. R., Patterson, D. A., … Farrell, A. P. (2005). Metabolic rates and swimming performance of adult Fraser River sockeye salmon (Oncorhynchus nerka) after a controlled infection with Parvicapsula minibicornis. Canadian Journal of Fisheries and Aquatic Sciences, 62(9), 2124–2133. https://doi.org/10.1139/f05-126

Wagner, T., Irwin, B. J., Bence, J. R., & Hayes, D. B. (2013). Detecting Temporal Trends in Freshwater Fisheries Surveys: Statistical Power and the Important Linkages between Management Questions and Monitoring Objectives. Fisheries, 38(7), 309–319. https://doi.org/10.1080/03632415.2013.799466

Walsworth, T. E., & Schindler, D. E. (2016). Long time horizon for adaptive management to reveal predation effects in a salmon fishery. Ecological Applications, 26(8), 2695–2707. https://doi.org/10.1002/eap.1417

Webb, J. H., Gibbins, C. N., Moir, H., & Soulsby, C. (2001). Flow Requirements of Spawning Atlantic Salmon in an Upland Stream: Implications for Water-Resource Management. Water and Environment Journal, 15(1), 1–8. https://doi.org/10.1111/j.1747-6593.2001.tb00296.x

Wipfli, M. S., Hudson, J. P., Caouette, J. P., & Chaloner, D. T. (2003). Marine Subsidies in Freshwater Ecosystems: Salmon Carcasses Increase the Growth Rates of Stream-Resident Salmonids. Transactions of the American Fisheries Society, 132(2), 371–381. https://doi.org/10.1577/1548-

8659(2003)132<0371:MSIFES>2.0.CO;2

Young, P. S., Cech, J. J., & Thompson, L. C. (2011). Hydropower-related pulsed-flow impacts on stream fishes: a brief review, conceptual model, knowledge gaps, and research needs. Reviews in Fish Biology and Fisheries, 21(4), 713–731. https://doi.org/10.1007/s11160-011-9211-0

Zeug, S. C., Sellheim, K., Watry, C., Wikert, J. D., & Merz, J. (2014). Response of juvenile Chinook salmon to managed flow: Lessons learned from a population at the southern extent of their range in North America. Fisheries Management and Ecology, 21(2), 155–168. https://doi.org/10.1111/fme.12063

Zimmerman, M. S., Irvine, J. R., O’Neill, M., Anderson, J. H., Greene, C. M., Weinheimer, J., … Rawson, K. (2015). Spatial and Temporal Patterns in Smolt Survival of Wild and Hatchery Coho Salmon in the Salish Sea. Marine and Coastal Fisheries, 7(1), 116–134. https://doi.org/10.1080/19425120.2015.1012246

8.0 Appendices

Appendix A: Candidate environmental variables

Variables selected for regression modeling with Cheakamus River juvenile Pink, Coho and Chinook salmon abundances for BC Hydro CMSMON1a. Some variables identified during hypothesis development could not be used during regression modelling because they violated assumptions necessary for linear regression modelling.

Transformation	Variable Name	Species
log10	April Discharge Variance	Chinook, Coho, Pink
log10	August Discharge Variance	Chinook, Coho, Pink
no	Cumulative April Temperature	Chinook, Coho, Pink
no	Cumulative August Temperature	Chinook, Coho, Pink
log10	Cumulative December Discharge	Chinook, Coho, Pink
no	Cumulative February Temperature	Chinook, Coho, Pink
log10	Cumulative January Discharge	Chinook, Coho, Pink
no	Cumulative January Temperature	Chinook, Coho, Pink
no	Cumulative July Temperature	Chinook, Coho
no	Cumulative June Discharge	Coho
no	Cumulative June Temperature	Coho
no	Cumulative March Discharge	Chinook, Coho, Pink
no	Cumulative March Temperature	Chinook, Coho, Pink
no	Cumulative May Discharge	Coho
log10	Cumulative November Discharge	Chinook, Coho, Pink
no	Cumulative November Temperature	Chinook, Coho, Pink
no	Cumulative October Temperature	Chinook, Coho, Pink
no	Cumulative September Temperature	Chinook, Coho, Pink
log10	December Discharge Variance	Chinook, Coho, Pink
log10	February Discharge Variance	Chinook, Coho, Pink
log10	January Discharge Variance	Chinook, Coho, Pink
log10	July Discharge Variance	Chinook, Coho
log10	June Discharge Variance	Coho
log10	March Discharge Variance	Chinook, Coho, Pink
log10	Maximum 6 hr Down Ramp March to July	Coho
no	Maximum 6 hr Down Ramp November to January	Chinook, Pink

log10	Maximum 6 hr Up Ramp March to July	Coho
log10	Maximum 6 hr Up Ramp November to January	Chinook, Pink
log10	Maximum April Discharge	Chinook, Coho, Pink
log10	Maximum December Discharge	Chinook, Coho, Pink
log10	Maximum January Discharge	Chinook, Coho, Pink
log10	Maximum July Discharge	Chinook, Coho
log10	Maximum June Discharge	Coho
log10	Maximum March Discharge	Chinook, Coho, Pink
log10	Maximum May Discharge	Coho
log10	Maximum November Discharge	Chinook, Coho, Pink
log10	Maximum October Discharge	Chinook, Coho, Pink
no	Minimum April Discharge	Chinook, Coho, Pink
no	Minimum April Temperature	Chinook, Coho, Pink
	Minimum August Discharge	Chinook, Coho, Pink
no	Minimum August Temperature	Chinook, Coho, Pink
no	Minimum December Temperature	Chinook, Coho, Pink
log10	Minimum Discharge Aug 15-31	Chinook, Coho, Pink
no	Minimum February Discharge	Chinook, Coho, Pink
no	Minimum February Temperature	Chinook, Coho, Pink
no	Minimum January Discharge	Chinook, Coho, Pink
no	Minimum January Temperature	Chinook, Coho, Pink
no	Minimum July Temperature	Chinook, Coho
no	Minimum July Temperature	Coho
no	Minimum June Discharge	Coho
no	Minimum June Discharge	Coho
no	Minimum March Discharge	Chinook, Coho, Pink
no	Minimum March Temperature	Chinook, Coho, Pink
log10	Minimum November Discharge	Chinook, Coho, Pink
no	Minimum November Temperature	Chinook, Coho, Pink
log10	Minimum October Discharge	Chinook, Coho, Pink
no	Minimum October Temperature	Chinook, Coho, Pink
no	Minimum September Temperature	Chinook, Coho, Pink
log10	November Discharge Variance	Chinook, Coho, Pink

log10	October Discharge Variance	Chinook, Coho, Pink
log10	September Discharge Variance	Chinook, Coho, Pink

Appendix B: Significant linear regressions for Pink Salmon

Significant results for linear models with Cheakamus River YOY Pink Salmon abundance.

Variable Name	Transformation	R^2	*p*-value	Slope Direction
October Discharge Variance	Log_{10}	0.47	0.04	Negative
Minimum November Discharge	None	0.61	0.01	Negative
Minimum February Discharge	Log_{10}	0.72	<0.001	Positive

Appendix C: Significant linear regressions for Coho Salmon

Significant results for linear or log-linear models with Cheakamus River Coho Salmon smolt abundance.

Variable Name	Transformation	R^2	*p*-value	Slope Direction
Cumulative December Discharge	Log_{10}	0.29	0.03	Negative
October Discharge Variance	Log_{10}	0.24	0.05	Negative
December Discharge Variance	Log_{10}	0.29	0.02	Negative
Maximum December Discharge	Log_{10}	0.29	0.02	Negative
February Discharge Variance	Log_{10}	0.27	0.03	Negative
Minimum September Temperature	None	0.58	0.01	Negative

Appendix D: Significant linear regressions for Chinook Salmon

Significant results for linear or log-linear models with Cheakamus River YOY Chinook Salmon abundance.

Variable Name	Transformation	R^2	*p*-value	Slope Direction
Minimum Discharge August 15-31	Log_{10}	0.49	0.00	Positive
Maximum July Discharge	None	0.29	0.02	Positive
July Discharge Variance	Log_{10}	0.30	0.02	Positive
Minimum January Discharge	None	0.29	0.02	Negative
Maximum 6hr Upramp (m^3hr^{-1}) November to January	Log_{10}	0.26	0.03	Negative
Minimum August Discharge	Log_{10}	0.50	0.00	Positive
February Discharge Variance	Log_{10}	0.27	0.03	Negative
Cumulative April Temperature	None	0.56	0.00	Negative
Minimum April Temperature	None	0.44	0.01	Negative
Cumulative July Temperature	None	0.50	0.01	Negative
Cumulative August Temperature	None	0.39	0.03	Negative
Minimum July Temperature	None	0.63	0.00	Negative
Cumulative September Temperature	None	0.34	0.05	Negative

Appendix E: Spawner Abundance Data

To compensate for the lack of adult abundance data and inform whether trends in juvenile abundance may have been related to regional (marine) conditions and/or conditions within the Cheakamus River, spawner abundance and marine survival data were compiled for adjacent populations in the south coast of British Columbia (Table 1). From 2001 to 2015, spawner abundance and marine survival were highly variable among salmon populations in the south coast of British Columbia (Figures 1, 2, and 3).

Table 1. Marine survival and adult escapement data collected from south coast British Columbia streams for comparison to Cheakamus River juvenile salmon abundance.

Species	Variable	Brood Years Available	Data Source
Coho Salmon	St. of Georgia Marine Survival	1999 to 2012	DFO unpublished data
Coho Salmon	Squamish Nation Cheakamus River Adult Counts	2000 to 2016	Squamish Nation unpublished data
Chinook Salmon	St. of Georgia Marine Survival	2000-2012	DFO unpublished data
Chinook Salmon	Squamish Nation Cheakamus River Adult Counts	2000-2016	Squamish Nation unpublished data
Chinook Salmon	Fraser Ocean Type Age 3 Adult Chinook Escapement	2000 -2015	Pacific Salmon Commission Joint Chinook Technical Committee (2016)
Chinook Salmon	Lower St. of Georgia Ocean Type Age 3 Adult Chinook Escapement	2000-2015	Pacific Salmon Commission Joint Chinook Technical Committee (2016)
Pink Salmon	Squamish Nation Cheakamus River Adult Counts	2000-2016	Squamish Nation unpublished data
Pink Salmon	Fraser River Adult Pink Production	2001-2015	Fraser River Panel (2016)

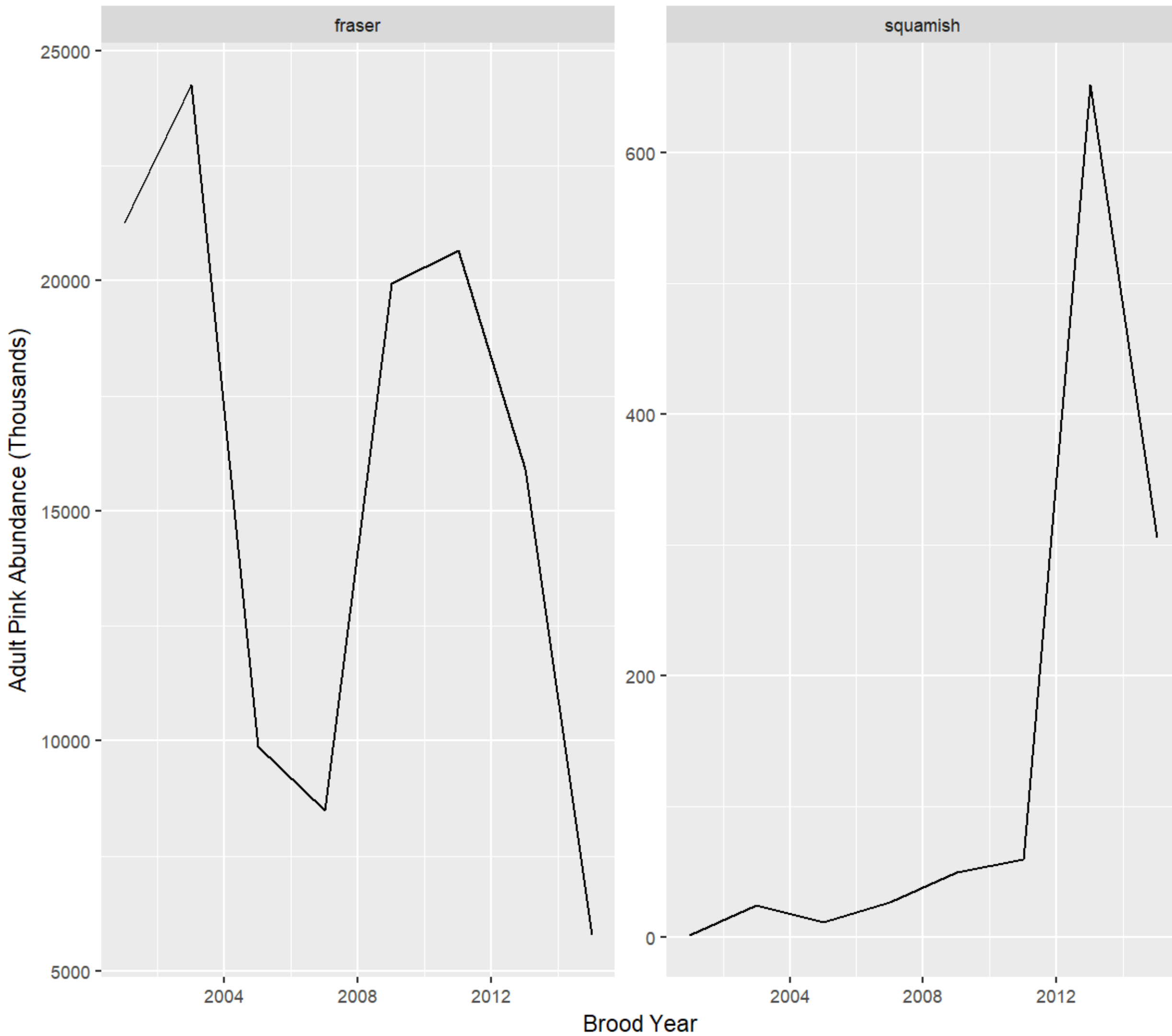

Figure 1 Plot of Fraser River and Squamish River Adult Pink Salmon abundance estimates 2001-2015.

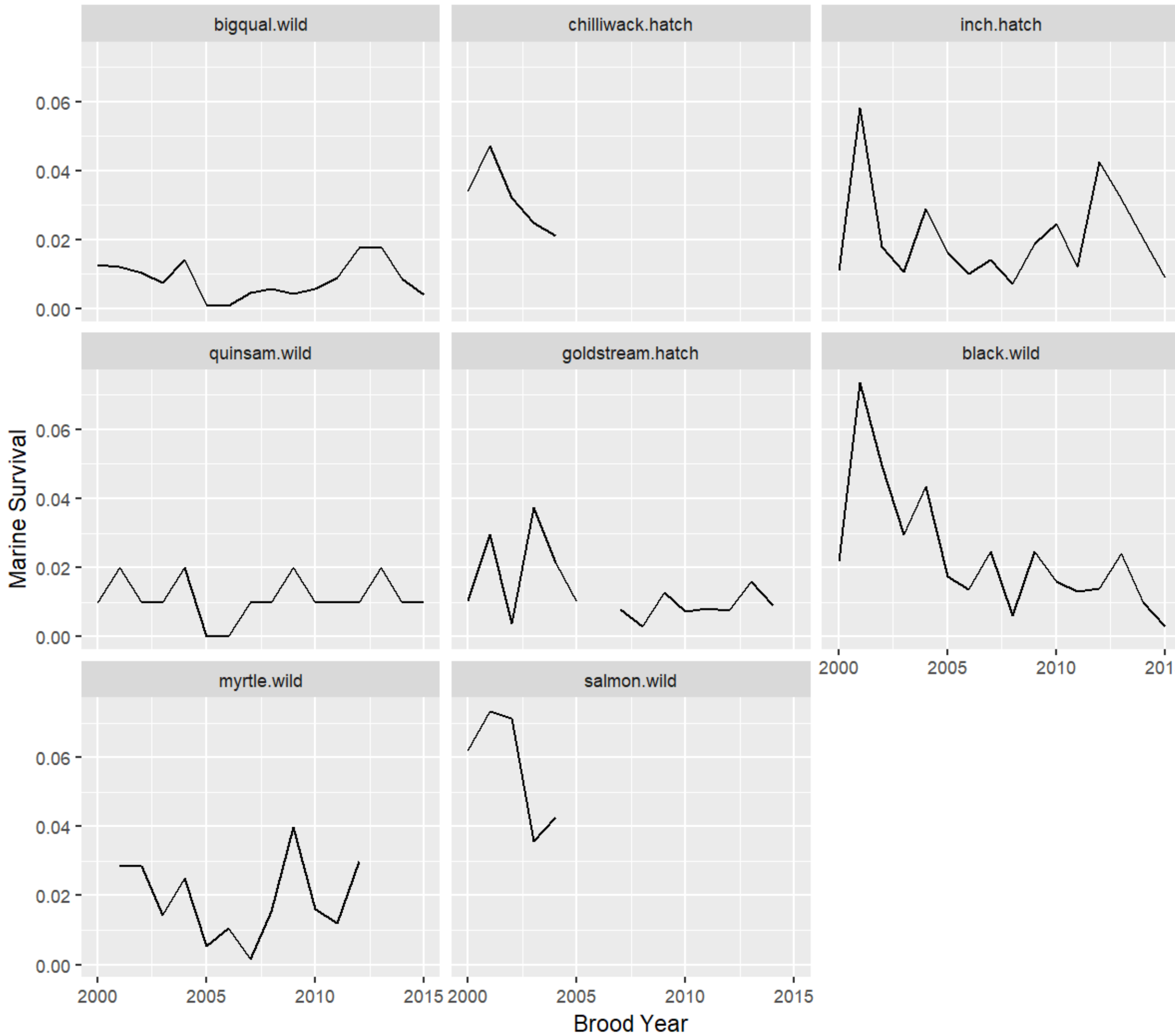

Figure 1 Marine Survival for Coho Salmon for multiple populations in the Straight of Georgia (DFO unpublished data).

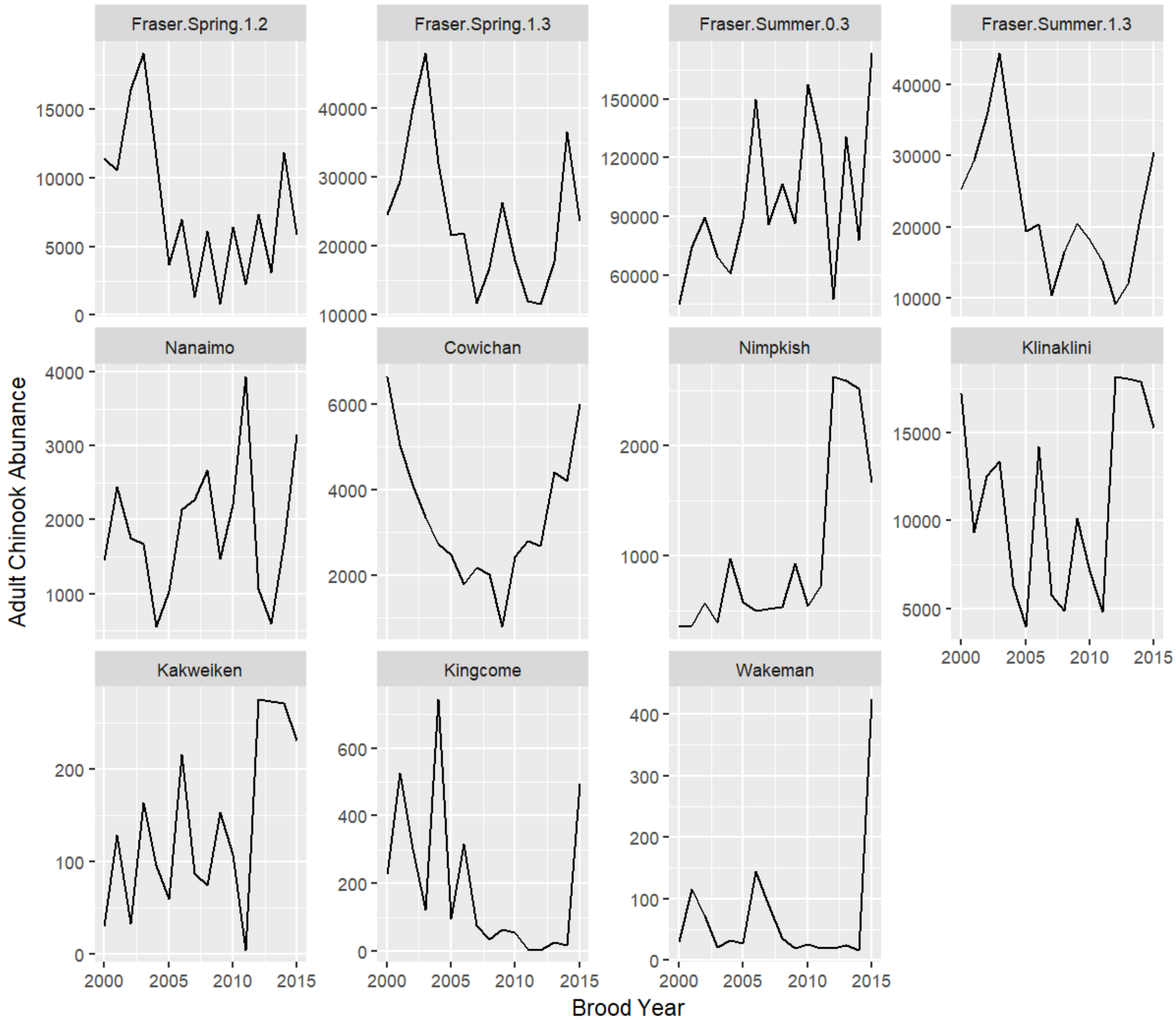

Figure 2 Chinook Salmon adult abundance data from other south coast British Columbia watersheds. Data sourced from Pacific Salmon Commission Join Chinook Technical Committee (2016).