



Cheakamus Project Water Use Plan

**Project: Synthesis of Adult and Juvenile Steelhead
Data to Evaluate Effects of the WUP Flow Regime in
the Cheakamus River**

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Executive Summary

This report provides a synthesis of information to address three questions about the effects of flow and operations of Daisy Lake Dam on Steelhead Trout in the Cheakamus River identified during the original Water Use Planning (WUP) process: 1) Do high flows in July and August negatively affect Steelhead fry that have recently emerged? 2) Does flow effect Steelhead juvenile production, as indexed by the number of fry, parr, smolts, and returning adults? and 3) Has the current WUP flow regime led to changes in Steelhead production, as indexed by adult returns, juvenile abundance, and smolt production?

Monitoring programs on the Cheakamus River provide estimates of annual Steelhead escapement (adult returns or spawners), juvenile abundance and survival rates, and smolt production. Escapement and smolt enumeration programs were initiated well before WUP-funded studies which began in 2007, and they provide the only data to compare production under the WUP flow regime relative to earlier flow regimes (question 3 above). Escapement estimates are available from 1996 to 2020 (24 years) and are based on repeat swim count data combined with radio telemetry information. Smolt run size is determined based on catches from a Rotary Screw Trap, and the dataset for Steelhead is 13 years long, with only 3 years available prior to WUP flows. The WUP-funded study for Steelhead has provided juvenile abundance estimates for various life stages (fall fry, age 0+, 1+, and 2+ parr in spring) from fall 2008 to fall 2020 (no data collected in spring of 2019) based on electrofishing and snorkeling surveys. Survival rates between juvenile life stages are computed from the ratio of abundance estimates, and can be related to flow conditions.

Steelhead escapement was low (average 169) prior to the Interim Flow Agreement (the pre-IFA period as characterized by returns from 1996-2001). In comparison, the average escapement produced under IFA flows (383, escapement from 2002-2007) was twice as high as during the pre-IFA period and this difference was statistically significant ($p=0.002$). The average escapement produced under WUP flows (since 2010), was 1.5-fold higher (578) than during the IFA period, and the difference was also statistically significant ($p=0.008$).

Variation in freshwater survival rate depends on more than just the flow regime, and non-flow factors must therefore also be accounted for when trying to determine the differences in steelhead production across flow regimes. As escapement trends are affected by changes in both freshwater and marine survival rates, differences in escapement between pre-IFA, IFA, and WUP periods must be corrected to account for differences in marine survival rates between these periods. Based on information from other Steelhead rivers in southern BC and Washington, marine survival was on average 1.3-fold higher in years effecting adult returns during the WUP regime relative to the IFA regime. In addition, Steelhead juvenile data from the Cheakamus River showed that annual survival rates of parr were more than 4-fold higher in years with large returns of adult pink salmon. Adult pink salmon return to the Cheakamus in only odd years, and the returning abundance is reliably indexed based on estimates of fry abundance the following spring from rotary screw traps. Only 3 estimates of pink fry abundance were available for the IFA period (averaging 0.7 million fry, highest value of 1.7 million) while there were six estimates of pink fry abundance during the WUP period (averaging 12.1 million fry, lowest value of 2.1 million). The high adult pink returns only occurred during the WUP period and were caused by broad-scale changes in marine survival given similar trends in other rivers. After accounting for effects of differences in marine survival of Steelhead and pink salmon returns between WUP and IFA periods, we estimate that freshwater production of Steelhead in the Cheakamus River has declined by 20% under the WUP flow regime.

The average run size of Steelhead smolts in the Cheakamus River from rotary screw trap data was 17% lower during the WUP period than during the IFA period, but this difference was not statistically significant. This difference increases to -44% if enhanced survival rates due to higher pink abundance during the WUP period are accounted for. The escapement- and smolt-based estimates indicate that Steelhead production under the WUP flow regime has declined by a modest amount, but this conclusion should be considered uncertain. This uncertainty arises from the estimation of correction factors for effects of marine survival and pink salmon abundance on subsequent Steelhead escapement, as well as the limited number of years (2) available for estimating smolt abundance during the IFA period.

Stock-recruit models predicted negative effects of rapid decreases in flow during summer on Steelhead egg-fall fry survival rates in the Cheakamus River (WUP questions 1 and 2). Models that included effects of density (number of eggs deposited) and rapid decreases in discharge (downramp rate) during July-September explained 65% of the variation in the log of egg-fall fry survival rates. This model provided a substantial increase in predictive ability relative to other flow-based models, and predicted egg-fry survival rates decrease with increases in downramp rates. The effect was substantive as the model predicted that production in the year with the lowest ramping rate was 3-fold higher compared to the year with the highest ramping rate. However, the estimated effect of downramp rate was exclusively driven by conditions in a single year (2010) when flows and downramp rates were very high (160 cms flow drop over one day). Predictions were not consistent with patterns seen in other years with more modest downramp rates (9-50 cms flow drop over one day). The second-best egg-fry survival model predicted that survival rates increased with the proportion of time when flows during summer were less than 40 cms ($r^2=0.52$). Thus, there is modest support for the hypothesis that high flows during the emergence period can have negative effects on fry survival rates.

Survival between fry in the fall and age-0+ parr in spring, which quantifies their overwinter survival rate, averaged 24% across years. The downramp rate over the winter explained the more variation in the log of overwinter survival rates (70%) compared to the other flow covariates that were evaluated. Density-dependence in survival rates between these life stages was negligible and lower than the density effect on egg-fry survival rates.

Annual survival rates of parr were on average four-fold higher in odd years when pink salmon return to spawn. This difference likely occurred because Steelhead parr consume pink salmon eggs which are a rich lipid source, which in turn increases their survival rate over winter. This effect was not observed for Steelhead fry which are too small to consume pink salmon eggs in the late summer and fall. A model predicting the log of annual survival rates of Steelhead parr as a function of pink salmon fry abundance (indexing pink salmon adult returns the previous fall) but no density (age 0+ Steelhead parr in spring) effects had substantively better predictive ability compared to a model with no pink or density effects. Flow-based models generally did not improve model fit

or did not make sensible predictions (e.g., negative density dependence). These findings are consistent with the literature on effects of flow on juvenile salmonids, which document that early life stages (eggs and fry) are more sensitive to flow effects than older life stages (parr).

Multiple lines of weak evidence indicate potential modest impacts of the WUP flow regime on the Steelhead population in the Cheakamus River. These include: 1) reduced freshwater production relative to the IFA regime based on pink salmon- and marine survival-adjusted escapement; 2) reduced freshwater production relative to the IFA regime based on differences in smolt production; and 3) reductions in predicted egg-fall fry and fall fry – spring age 0+ parr survival rates at elevated ramping rates observed in one year (2010) during the WUP period. However, when we applied the flow covariate models to historical flow data during the IFA and WUP periods, the median survival rate during the WUP period was higher than during the IFA period, a result not consistent with estimated production changes based on escapement and smolt abundance. Thus, because of considerable uncertainty in the magnitude of estimated WUP vs IFA production changes, and uncertainty about potential causes of these effects, differences in regimes are not well determined, but appear to be modest.

Flow covariate models presented here can be used to estimate population-level impacts of alternate flow regimes during the WUP Order Review. However, given the uncertainties described above, the predicted benefits to Steelhead production from these models under different ramping rates or other conditions, should be considered preliminary, and monitoring under a new flow regime will be needed if there is a desire to verify model predictions and evaluate whether the intended benefits from a new flow regime are being achieved.

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1.0 General Introduction

The Cheakamus River is a productive tributary of the Squamish River that supports populations of Steelhead Trout (*Oncorhynchus mykiss*), Chinook (*Oncorhynchus tshawytscha*), Coho (*Oncorhynchus kisutch*), Pink (*Oncorhynchus gorbuscha*), and Chum (*Oncorhynchus keta*) salmon, as well as resident populations of Rainbow Trout (*Oncorhynchus mykiss*), Bull Trout (*Salvelinus confluentus*), and other species. Daisy Lake Dam impounded the Cheakamus River in 1957 and a proportion of the water entering Daisy Lake Reservoir is diverted to the Squamish River for power generation. The Cheakamus River downstream of Daisy Lake Reservoir extends 26 km to its confluence with the Squamish River and the lower 17.5 kilometers of this river are accessible to anadromous salmon and Steelhead (Fig. 1.1). As a result of the diversion, the Cheakamus River downstream of the dam receives less than 50% of its natural discharge annually (BC Hydro 2005, see Fig. 2.2), and there is much interest in understanding how this altered flow regime effects its fish populations.

The Cheakamus River supports a wild winter-run Steelhead population and a well-known Steelhead fishery. Although adult Steelhead returns are likely much smaller today relative to returns in the 1980s and earlier, the run still attracts considerable angling effort and is one of the more productive wild Steelhead populations in southern BC (Van Dischoeck 2000). Steelhead juveniles rear for two to four years in the Cheakamus River before migrating to sea as smolts. Steelhead juveniles are potentially more sensitive than juveniles from other anadromous species in the Cheakamus River to changes in flow because they have a longer period of freshwater residency. All these factors contribute to a strong interest among resource users and fisheries managers in determining whether changes in the flow regime below Daisy Lake Dam are affecting Steelhead production in the Cheakamus River.

There was considerable debate during the Cheakamus River Water Use Planning (WUP) process on the effects of flow regime on juvenile salmon and Steelhead production (Marmorek and Parnell 2002, BC Hydro 2005). Proponents of the Interim Flow Agreement (IFA) (the regime in place prior to changing to the WUP regime in 2006), which partially mimicked natural inflow patterns, argued that both seasonal and daily elements of the hydrograph could be important to juvenile salmonid production and

that flows above minimum WUP levels would provide benefits in off-channel rearing areas that were not accounted for in the fish habitat modeling conducted during the initial WUP review. Proponents of the WUP flow regime had more confidence in the fish habitat modeling results, which suggested that dam operations do not affect the quantity or quality of mainstem and side channel rearing areas except at very low flows (Fig. 1.2).

Near the end of the initial WUP process, monitoring and research activities to resolve such uncertainties were identified. The key uncertainties for Steelhead identified during the Cheakamus WUP addressed by this project are:

1. Do high flows in July and August negatively affect Steelhead fry that have recently emerged?
2. Does flow effect juvenile production, as indexed by the number of fry, parr, smolts, and returning adults?
3. Has the current WUP flow regime led to changes in Steelhead production, as indexed by adult returns, juvenile abundance, and smolt production?

The first question is based on the concern that maintaining higher flows during and shortly after the Steelhead fry emergence period (July and August), to provide benefits for recreational boaters, could displace fry from preferred shallow edge habitats and reduce the availability of this habitat, ultimately leading to a reduction in egg-fry survival rates, which would in turn lead to reduced smolt production and adult returns. The second question is more general and can be evaluated by comparing various statistics of the flow regime (minimum winter flows, average flow or flow fluctuations during summer) to abundance and survival estimates. The third question focuses on whether survival and abundance estimates for various Steelhead life stages have changed due to the current WUP operation. This can be addressed by comparing abundance estimates obtained prior to implementation of the WUP regime with estimates obtained under the regime.

As part of the new water license agreement for the Cheakamus River, BC Hydro supported a number of monitoring programs to assess the effects of the WUP flow regime on fish populations downstream of the dam (BC Hydro 2007). CMSMON#1a enumerated the number of fry and smolts outmigrating past a Rotary Screw Trap (RST) from late

winter through spring, and in most years this program has provided estimates of Steelhead smolt run size. CMSMON#3 provided estimates of the abundance of returning adult Steelhead spawners (escapement), abundance of various juvenile life stages rearing in the river, and survival rates among life stages. The central objective of CMSMON#3 is to address the three critical uncertainties summarized above, and more broadly to determine if the number of adult returns, juvenile abundance, and smolt production are affected by flows and the WUP flow regime. The overall approach to addressing these questions is relatively straightforward: 1) quantify escapement and juvenile abundance in the fall and spring, and smolt production in the spring; 2) use these metrics to determine the survival rate between life stages and define life stage-specific stock-recruitment relationships; and 3) over time, compare abundance, survival rates and stock-recruitment relationships under different flows, and relate changes in these metrics to particular flow regimes or unique flow events (Fig. 1.3).

Steelhead escapement to the Cheakamus River has been assessed since 1996 (Korman and Schick 2017, Korman et al. 2007). The historical time series of escapement in part reflects the rivers capacity to produce Steelhead under at least three different flow regimes (pre-IFA, IFA, and WUP). The simplest way to determine whether changes in flow have affected Steelhead production is to compare escapement over these regimes (e.g., Fig. 1.3a). However, as escapement is also determined by parental abundance and marine survival, inferences regarding changes in freshwater habitat due to dam operations from this comparison may be weak unless flow effects are very strong relative to these other factors. To address this limitation, estimates of Steelhead fry in the fall, and parr and smolt abundance in the spring can be used to index freshwater productivity (e.g., Fig. 1.3b). Each annual estimate contributes a single data point for freshwater stock-recruitment relationships between the parental escapement and the resulting juvenile abundance. These relationships control for the effect of egg deposition on subsequent juvenile production, and remove any remaining effects associated with changes in marine survival (e.g., Fig. 1.3c). As data points accumulate (Fig. 1.4), it is possible to relate outliers from the stock-recruitment relationships, which indicate substantially higher or lower juvenile Steelhead production per unit escapement, to particular aspects of the flow regime, such as the frequency and magnitude of high flow events during the summer, or

the duration of minimum flow periods during the winter. If the flow regime changes in the future, stock-recruitment relationships developed under the current WUP flow regime can be compared to a relationship estimated under the new regime (e.g., Fig. 1.3c).

Escapement-to-parr or -smolt stock-recruitment relationships are necessary for evaluating population-level effects of flow, but provide little insight into what life stages are most affected or which elements of the flow regime have the biggest effect on juvenile Steelhead survival. For example, higher flows during summer or sudden reductions in flow over this period could increase mortality of recently emerged Steelhead fry, but this mortality may not affect subsequent parr abundance and overall freshwater production if lower densities lead to higher survival the following winter. To account for such dynamics, it is necessary to quantify survival rates and stock-recruitment relationship for multiple juvenile life stages. We therefore develop relationships between escapement and age-0+ Steelhead in the fall (fry), between age-0+ fish in the fall and the following spring (0+ parr), and between age-0+ and age-1+ fish in the spring. The first relationship quantifies incubation success and survival from emergence (summer) into the fall. The second quantifies age-0+ overwintering survival. The third quantifies the annual survival rates for parr.

This report provides a synthesis of Steelhead information from the Cheakamus River. We relate patterns in abundance and survival to planned changes in the flow regimes and examine how unplanned aspects of the flow regime potentially effect production. We evaluate the utility of this information for addressing uncertainties. The objectives of this report are to clarify relationships between flow and Steelhead production in the Cheakamus River based on available data and to determine if the data are sufficient to address critical uncertainties regarding steelhead-flow relationships identified during the WUP. The report is organized into seven chapters. Chapter two summarizes discharge data from the Cheakamus River with the primary intent of showing how historical operations of Daisy Lake Dam affect flow. We focus on describing differences in flow under the pre-Interim Flow Agreement (pre-IFA), IFA, and WUP regimes. Chapter three provides a brief summary of the methods used to estimate Steelhead abundance and survival rates, and analytical methods used in this report. Chapter four summarizes the life history of Cheakamus River Steelhead which is needed

to interpret effects of flow on abundance and survival. Chapter five provides results from the analysis of escapement data. This and a limited amount of smolt production data, is the only information for Steelhead that spans both IFA and WUP flow regimes. Chapter six summarizes key findings based on adult and juvenile data with respect to effects of flow and other factors. It also provides a summary of a study that evaluated stranding of Steelhead fry and parr due to a rampdown event in August 2018 (an event that is now common under the WUP flow regime). Chapter 7 integrates information from previous chapters and provides final conclusions.

2.0 Summary of Effects of Daisy Lake Dam on Discharge in the Cheakamus River

This chapter describes how discharge in the Cheakamus River has changed under the pre-Interim Flow Agreement (pre-IFA), IFA, and Water Use Planning (WUP) regimes. We analyze the long-term record of discharge from the Cheakamus River at the Brackendale gauge (WSC gauge 08GA043, Fig. 1.1). We also analyze flow records provided by BC Hydro on discharge from Daisy Lake Dam, turbine flows into the Squamish River, and inflow to Daisy Lake.

Patterns of natural inflows to the Cheakamus River are driven by snowmelt floods during the spring freshet, moderate and declining flows through summer and early fall, and a long low-flow period during late fall and winter punctuated by occasional floods driven by rainfall events (Fig. 2.1). The timing and volume of diversion rates from the Cheakamus River to the Squamish River, which affects flow downstream of the Daisy Lake Dam, have varied considerably since impoundment (Fig. 2.2). From 1958-1994, diversions rates were used to maximize by power generation within the constraints of the original water license. Historical operations did not always follow the original water license which specified that a minimum of 45% of the annual inflow to Daisy Lake Reservoir be released into the Cheakamus River from Daisy Lake Dam, with the remaining 55% potentially diverted to turbines and released in the Squamish River. These violations ultimately led the Department of Fisheries and Oceans to issue an Interim Flow Order (IFO) to BC Hydro in 1997 (BC Hydro 2005). This order was subsequently modified and called the Interim Flow Agreement (IFA), and the BC water comptroller ordered BC Hydro to implement the agreement in April 1999. The IFA specified that the greater of $5 \text{ m}^3 \cdot \text{s}^{-1}$ or 45% of the previous seven days average inflow to Daisy Lake Reservoir be released into the Cheakamus River downstream of the dam (within a daily range of 37-52%). These changes led to more water in the Cheakamus River downstream of the dam (Fig. 2.2) and a more naturally-shaped hydrograph. They also resulted in a 25% reduction in hydroelectric generation from 790 GWh/yr to 590 GWh/hr (Marmorek and Parnell 2002). In February 2006, the operating constraints were modified based on a recommended flow regime that came from a Water Use Planning (WUP) process

conducted between 1999 and 2002 (BC Hydro 2005). The WUP flow regime was based on meeting minimum flows at the dam and further downstream at the Brackendale gauge (Fig. 1.1), and operating rules no longer depended on releasing a fixed fraction of inflows to the reservoir (BC Hydro 2005, Table 2.1). The WUP flow regime also specified maximum rates of discharge change, which varied with the magnitude of discharge released from Daisy Lake Dam. These rules resulted in prolonged periods of minimum flows, and occasional rapid and large flow increases and decreases (Fig. 2.1). The new flow regime led to an increase in economic returns from power generation of about \$7 million/year relative to generation under the IFA regime (Marmorek and Parnell 2002).

The effect of operations on flow in the Cheakamus River are greatest during winter when inflows are lowest, because the diversion takes a greater proportion of the inflow to Daisy Lake. There has been a noticeable change in minimum flows during winter under different operating regimes (Fig. 2.3). Minimum flows in winter have been slightly higher under the WUP flow regime relative to the IFA regime, and minimum flows were much lower during the pre-IFA period.

Operations during late spring and summer are dominated by inflows to Daisy Lake, which often exceed the storage capabilities of the reservoir and the capacity of the tunnels ($\sim 65 \text{ m}^3 \cdot \text{s}^{-1}$) which divert water to the Squamish River. Occasional maintenance on Daisy Lake Dam and at the Cheakamus Powerhouse temporarily reduces reservoir storage and diversion capacity, which affects releases from the dam (Fig. 2.4). Flows into the Cheakamus River downstream of the dam have been greater in years when maintenance has occurred at the powerhouse (e.g., 2010 and 2011). Other operations during this period have occasionally led to sudden reductions in flow (e.g. drops in early and mid-August 2010 to help Chinook broodstock collection). The patterns in flow changes in August are clearly less abrupt under the IFA regime (Fig. 2.4a) compared to the WUP regime (Fig. 2.4b). This occurred because the IFA regime limited the extent of rapid flow changes because releases from Daisy Lake Dam were determined by the previous weeks' inflow.

Prescribed ramping rates controlling the rate of change in discharge from Daisy Lake Dam specified in the WUP (Table 2.1) substantially exceed guidelines from Fisheries and Oceans Canada (FOC 2012) and have the potential to impact survival rates

of juvenile Steelhead. We provide two examples to highlight these potential effects (Table 2.2). In the first case discharge at the Brackendale gauge is reduced from $100 \text{ m}^3 \cdot \text{s}^{-1}$ to $60 \text{ m}^3 \cdot \text{s}^{-1}$. This is a relatively modest change compared to some that have occurred (2010 and 2012 in Figure 2.4). The downramp rate at Daisy Lake Dam would be a minimum of $13 \text{ m}^3 \cdot \text{s}^{-1} \cdot \text{hr}^{-1}$ (Table 2.1), thus this flow change would require about three hours to implement. This is equivalent to a stage change of 9.2 and $9.9 \text{ cm} \cdot \text{hr}^{-1}$ at the Brackendale gauge site and the pedestrian bridge, respectively. These ramp rates are about 4-fold greater than the FOC $2.5 \text{ cm} \cdot \text{hr}^{-1}$ guideline. Actual flow changes at the Brackendale gauge would occur more slowly owing to wave attenuation, but the recorded stage changes are still very rapid. For example, on August 17, 2010, discharge at the Brackendale gauge decreased from $94 \text{ m}^3 \cdot \text{s}^{-1}$ to $72 \text{ m}^3 \cdot \text{s}^{-1}$ in one hour and to $56 \text{ m}^3 \cdot \text{s}^{-1}$ in two hours. This is equivalent to a stage change of about $15 \text{ cm} \cdot \text{hr}^{-1}$ (6-fold higher than the FOC guideline). Another common flow reduction occurs when flows at the Brackendale gauge site in mid-August are reduced from minimum rafting flow levels of $38 \text{ m}^3 \cdot \text{s}^{-1}$ to the seasonal minimum flow requirement of $20 \text{ m}^3 \cdot \text{s}^{-1}$ (Table 2.1). This flow change occurs in about 1.5 hrs. given a downramp rate of $13 \text{ m}^3 \cdot \text{s}^{-1} \cdot \text{hr}^{-1}$ (Table 2.1), which is the most likely scenario given that releases from Daisy Lake Dam are greater than $10 \text{ m}^3 \cdot \text{s}^{-1}$. This rate of flow change translates to a stage change rate of approximately 17 and $19 \text{ cm} \cdot \text{hr}^{-1}$, about 7-fold higher than the FOC ramping guideline. Wave attenuation and purposeful reductions in the downramp rate by BC Hydro operators attempting to mitigate stranding impacts (C. Rombough, BC Hydro, pers. comm.) have resulted in less drastic rates of change, but are still well above the guideline until very recently (summer of 2019).

3.0 Methods

In this chapter we summarize the methods used for estimating steelhead escapement, juvenile abundance, and smolt production for Steelhead in the Cheakamus River. We also describe analytical approaches.

3.1 Escapement

Steelhead escapement to the Cheakamus River is estimated by a model which integrates data on raw counts from swim surveys, run-timing determined from radio telemetry, and mark-recapture to determine observer efficiency during swims (Korman et al. 2007, Korman and Schick 2017, Fig. 3.1). The area surveyed for returning Steelhead is limited to the upper 14.5 km of the anadromous portion of the river, and begins approximately 500 m below a natural barrier, extending to the confluence with the Cheekye River (Fig. 1.1). On each survey, a team of three divers floats the entire survey area and record the number of Steelhead, resident rainbow trout, and bull trout that are observed. These surveys have been conducted over 24 years between 1996 and 2020 (no surveys were conducted in 1997 due to disputes around the IFA). In early years (1996-2000) an average of 6 swims were conducted per year. Effort increased beginning in 2000 and has averaged 11 swims/year between 2000 and 2020. Since 2000, swims are typically conducted on a weekly or bi-weekly basis from early March through early May. High flow conditions during late April and May limit survey opportunities which makes it difficult to quantify the abundance of the late-timed component of the run.

To convert counts of Steelhead to estimates of the number of Steelhead present on each survey, the observer efficiency of the swim crew on each survey needs to be estimated. In a subset of years (2001, 2003-2005, 2009-2011, 2016-2017), adult steelhead were captured by angling and given an external tag that would be visible to divers, as well as a radio tag to determine the number of tagged fish in the survey area on each swim survey. In essence, the ratio of the number of tags observed to the tags present on a swim is used as the observer efficiency estimate for that swim, and the expanded count on the swim (abundance) is determined by the ratio of the observed count to the observer efficiency (i.e. $\text{abundance} = \text{count}/\text{efficiency}$). A relationship between observer efficiency and river conditions (water clarity and discharge) is used to estimate observer efficiency

on swims and in years when radio telemetry data are not available, and to improve the precision of estimates when tagging information is available.

To convert estimates of the number of fish present on each swim to an annual escapement estimate, information on run-timing is required. Steelhead have a prolonged period of migration and spawning. Fish enter the Cheakamus River from December through May, and exit the system from early-April through late June. Radio telemetry information provides information on survey life (how long adult steelhead spend in the survey area) and departure timing (Fig. 3.1). Information on run-timing determined through survey life, departure timing, and repeat counts in each year is used to estimate the fraction of the annual run present on each survey. Data on counts, observer efficiency, and run-timing is integrated in a model to estimate the annual escapement (Korman et al. 2007, Korman and Schick 2017). A creel and angler logbook program was initially conducted in years when telemetry was conducted (2000, 2001, 2003-2005, 2009-2011, and 2016-2017). Beginning in 2012, scale collection was also conducted in years when telemetry was not conducted. This program provides information on the ratio of hatchery and wild fish in years when hatchery fish returned to the Cheakamus River (2009-2011), and information on the size and age structure of returning adults (based on the collection of scales) and the proportion of females. Some of this information is used to translate estimates of escapement in estimates of egg deposition.

We used swim counts combined with radio telemetry data collected in 2016 and 2017 to estimate abundance of resident rainbow trout in the Cheakamus River. Radio-tagged resident trout were given different colored external tags in 2016 and 2017, and only tags placed in the same year that the fish were counted in were used in the analysis. In 2017, tags placed in 2016 could have fallen off or would be more difficult to see due to the accumulation of algae. As for Steelhead, the radio tags allowed us to determine how many tagged resident trout were in the swim area during each survey. The abundance estimation model assumes no resident rainbow trout leave the swim area during the survey period. This was confirmed through the examination of radio tagging data, which showed that none of the 51 effectively tagged resident trout left the survey area prior to the last swim dates in 2016 and 2017.

We estimated resident trout abundance for each year swims were conducted. Detection probability for resident trout was very high and not sensitive to river conditions. As a result, we estimated detection probability from all swims when tags were present, and expanded the number of residents counted on each swim to determine the number present. These values were then used to estimate abundance, which was effectively a weighted-average of swims-specific abundance estimates. As for the Steelhead model, detection probability on each swim is assumed to be drawn from a hyper-distribution whose mean and variance is estimated using data from all swims when tags were present. The expansion of counts for each swim depends on that swim's detection probability if tags were present, or a random draw from the detection probability hyper-distribution if none were present.

3.2 Juvenile Abundance

The abundance of juvenile steelhead in the fall and spring is estimated by a combination of electrofishing and snorkel surveys (Korman et al. 2012). We used a multi-gear two-phase sampling design to estimate the abundance of age 0+-, 1+-, and 2+ juvenile steelhead in the Cheakamus and Brohm Rivers. Data from Brohm River provides a reference or control system to compare with results from the Cheakamus, which is influenced by flow regulation. We first conducted habitat surveys in both systems to quantify the length of shoreline that was potentially useable by juvenile Steelhead. In the Cheakamus River, we classified useable shoreline habitat into riffle, shallow, and deep water habitat types and used different gears to sample these habitats depending on season (fall or spring) and fish age. We have shown that electrofishing provides the most unbiased and precise estimates of age-0+ abundance in habitat types where the gear can be effectively applied (riffle and shallow water habitat), while snorkeling provides the most unbiased and precise estimates of abundance for age-1+ and older juvenile steelhead in shallow- and deep-water habitats (Korman et al. 2010b). Fall estimates of abundance are based exclusively on electrofishing as water clarity is too turbid for snorkeling, while spring abundance estimates are based on data from both electrofishing and snorkel surveys. With this design we can effectively quantify age 0+, 1+, and 2+ parr abundance in the spring, but only 0+ abundance in the fall.

Abundance is estimated using a two-phase sampling design. We sample a large number of index sites using a single pass of effort. At a sub-sample of sites, we conduct two-day mark-recapture experiments to quantify detection probability. We define detection probability as the proportion of individuals at a site that are either captured by electrofishing or seen by a diver based on a single pass of effort. Abundance at index sites is estimated by expanding the observed number of fish by the estimates of detection probability determined from mark-recapture experiments. The abundance of fish in the shorelines that are not sampled is estimated based on average fish densities and variation in density across sampled sites. The total estimate of abundance for the river is the sum of estimates from sampled and unsampled shorelines. We use a Hierarchical Bayesian Model (HBM) to implement this approach to estimate posterior distributions of abundance, from which expected values (means), medians, and 95% credible intervals are calculated.

3.3 Smolt Production

The abundance of Steelhead, Chinook, and coho smolts, as well as the abundance of Chinook, Pink, and Chum fry, are estimated using data from two Rotary Screw Traps (RST) located at river KM 5.5 adjacent to the North Vancouver Outdoor School (Melville et al. 2012, Fig. 1.1). Unmarked fish captured at the RST location are marked, transported upstream, and released. The marked fish are assumed to mix with the unmarked fish as they move downstream, and some (along with unmarked fish) are re-captured at the RSTs. The recapture of the marked fish provides information on the capture efficiency of the RSTs, which is then used to expand the number of unmarked fish captured, to estimate the population of fish passing the RST location. A fraction of Steelhead smolts that are captured have a sample of scales removed to determine their age.

Mark-recapture models are used to convert the catch of marked and unmarked fish into an estimate of the total population that migrates past the RST from mid-February to mid-June when the traps are operated. A variety of models have been used through time, including the unstratified Peterson estimator, the stratified Darroch (1961) estimator, and more recently, a hierarchical Bayesian model (BT SPAS, Schwarz and Bonner 2012). Estimates of run size for Steelhead smolts can be unreliable because the total run is relatively small and the fraction of Steelhead captured by the trap is low. Low

trap efficiency occurs because Steelhead smolts are large and can evade the trap, and a large fraction of the run leaves in May when trap efficiency is low due to high discharge. Because trap efficiency changes through time, the unstratified Peterson estimator will underestimate run size and the uncertainty in run size. Application of the stratified estimator is problematic for Steelhead because too few fish are marked and recaptured within each weekly strata. The Bayesian model provides a statistically sound way of computing a stratified estimate given the sparse data.

3.4 Analytical Methods

3.4.1. Analysis of historical escapement record

Our record of Steelhead escapement to the Cheakamus River begins in 1996 and extends 24 years through 2020 (no data from 1998). This time series spans three different flow regimes (pre-IFA, IFA, and WUP) and therefore has the potential to be used in before-after comparisons of flow regime effects (e.g. Fig. 1.3). However, the number of returning spawners (escapement) depends on: 1) the number of eggs deposited in the brood years contributing to each year's return 4-6 years later; 2) survival rates in freshwater as determined by flow- and non-flow factors; and 3) marine survival rates. Thus, using an escapement trend to make inferences about flow effects on freshwater survival rates requires the use of correction factors to remove non-flow freshwater effects and marine survival effects (see Fig. 4.1). Our first step was to average escapements over pre-IFA, IFA (pre-CN spill) and WUP periods. To correct for marine survival effects, we quantified differences in smolt-adult survival rate (marine survival) over these three periods. We used a composite smolt-adult survival time series from winter-run Steelhead stocks in Puget Sound and the winter-run Keogh River population (Kendall et al. 2017). We also computed the average smolt-adult survival rate for these three periods using data from the Keogh River only (Middleton 2017). Marine survival rates for each return year were based on a weighted average of survival rates two and three years earlier. The weights were based on the proportion of ocean age two and three year fish in each return year. Average escapements for each period were then adjusted based on differences in smolt-adult survival rates among periods.

As shown in the synthesis of juvenile survival data for the Cheakamus (see Chapter 6), there was a four-fold increase in annual survival rates of parr in years when pink salmon returns were high. Pink salmon returns were low during the IFA period and higher in some years during the WUP period (in most odd years). To separate non-flow (pink salmon) and flow effects on freshwater survival rate we therefore needed to remove the pink salmon effect. This was done by adjusting the escapement difference between WUP and IFA periods by the average pink salmon effect on parr survival rates during these periods.

The number of adult Steelhead returning to the Cheakamus River will be determined by freshwater and marine survival rates as well as the number of spawners that produced the returns, often termed brood escapement or spawning stock. Insufficient ‘seeding’ or egg deposition in one or a series of brood years would lead to reduced returns in later years even if freshwater and marine survival rates were constant. This could lead to biases in the interpretation of escapement data to evaluate flow regime effects. For example, low survival rates during the pre-IFA flow regime period would have led to low returns during the IFA regime. As a result, escapement from fish that reared under the IFA regime could be low simply because habitat was under-seeded. This would give the false impression that the IFA regime had a negative effect on freshwater production. The effects of under-seeding can be accounted for by analyzing the escapement data in a stock-recruitment framework, which quantifies the relationship between the spawning stock (escapement or egg deposition) and the resulting adult recruitment from that stock. These relationships can be computed for each regime and therefore correct for potential effects of under-seeding on subsequent escapements.

Development of stock-recruitment models for the Cheakamus River begins with the construction of a stock-recruitment data set where the recruitment for each brood year t is determined based on age-structure data. Recruitment by brood year (R_t) is calculated from,

$$3.1) \quad R_t = E_{t+3}P_{t+3,3} + E_{t+4}P_{t+4,4} + E_{t+5}P_{t+5,5} + E_{t+6}P_{t+6,6},$$

where E is the wild-origin escapement in year $t+a$ and P is the proportion of maiden fish returning in year $t+a$ at total age a . Age proportions were specific to years when a sufficient scale sample was available (2000, 2001, 2003-2005, 2009-2011, 2013-2019)

and in other years were held constant at the multi-year average. As no escapement estimate was available for 1998, we averaged escapements from 1997 and 1999 to calculate escapement for this year. This was necessary to compute the spawning stocks for the 2001-2004 return years. Stock-recruit analyses of adult data are traditionally only applied to semelparous species (spawn only once), or to immature stages of iteroparous species (spawn more than once). In the case of Steelhead, which are iteroparous, the number of repeat spawners (as determined from scales) must be removed from the number of recruits or they would be double-counted in the stock-recruit analysis. We used the average repeat spawner rate across all years based on scale reads to compute the number of maiden recruits (maiden recruits = total recruits * (1-repeat spawner fraction)). We then plotted the number of maiden adult recruits as a function of the spawning stock that produced it, and fitted Beverton-Holt and Ricker stock-recruit models to these data.

The form of the Beverton-Holt model we fit was,

$$3.2) \quad R = \frac{\alpha \cdot S}{1 + \frac{\alpha}{\beta} \cdot S} \cdot e^{\gamma \cdot X}$$

where R is the recruitment, S is the stock size that produced that recruitment (as quantified by escapement or egg deposition), α is the maximum recruits per spawner (or egg deposition) which is often termed stock productivity, β is the maximum number of maiden returns (termed carrying capacity), γ is the WUP offset parameter, and X is a dummy variable set to 0 for recruitments not effected by the WUP flow regime (brood years 2005 and earlier) and 1 if they are effected (brood years 2006 and later). The product of γ and X will therefore be 0 for brood years prior to the WUP, and α and β define the pre-WUP stock-recruitment curve. As X is one in for brood years effected by WUP flows, γ shifts recruitment up ($\gamma > 0$) or down ($\gamma < 0$) by the same amount for any stock size in WUP years. The form of the Ricker model we fit was,

$$3.3) \quad R = S \cdot e^{\alpha + \beta \cdot S + \gamma \cdot X}$$

where α is the log of maximum recruits per spawner and β is the density-dependent effect. Parameters for both Beverton-Holt and Ricker models were estimated by

maximum likelihood in Excel using Solver assuming that error in the log of predicted and observed recruitment (Beverton-Holt) or the log of recruits/spawner (Ricker) is normally distributed.

Estimates of spawning stock that determine subsequent recruitment can be improved by accounting for inter annual variation in sex ratios and fecundity of spawners. To evaluate these factors for Cheakamus Steelhead, we computed egg deposition in years when information on sex ratio and female fork length were available from angling surveys. Annual egg deposition was computed as the product of total escapement, the proportion of the escapement made up of females, and fecundity. The latter was computed based on annual average female fork length from the Cheakamus River and a fecundity-female fork length relationship for winter-run Steelhead from the Keogh River (Ward and Slaney 1993). The multi-year average egg-deposition to escapement ratio was used to compute total egg deposition (based on the product of the ratio and escapement) in years when year-specific egg deposition estimates could not be computed due to an absence of data from anglers. Stock-recruitment models were fit using escapement and egg deposition as the measures of stock size (S in eqn. 3.2 or 3.3 = escapement or eggs).

3.4.2. Emergence Timing

Determining the emergence timing of juvenile Steelhead in the Cheakamus River is needed to address the WUP management question on potential negative effects of higher flows in July and August on survival rates of recently emerged fry. This life stage is potentially very sensitive to variation in flow because it is highly dependent on shallow and slow-moving water at the rivers' edge. This habitat is destabilized by rapid fluctuations in flow which can occur during regular operations, especially when ramping rates are high (Nislow and Armstrong 2012).

We estimated emergence timing for Cheakamus River Steelhead by using an integrated analysis that combined estimates of spawn timing from radio telemetry data with water temperature data and incubation time models. Unlike Pacific salmon, female Steelhead spawners return to the ocean shortly after spawning and spend very little time defending redds. Thus, information on departure timing from female Steelhead that were

radio-tagged can be used to define spawn timing. We fit departure timing data available in years when radio telemetry was conducted using a hierarchical Bayesian model that predicted departure timing using normal distributions. The mean and the variance of the distributions for each year are assumed to be random effects drawn from hyper-distributions. These hyper-distributions represent the mean date of departure timing and the extent of variation in the mean date across years, and the variance in departure timing within-years, and the extent of variation in that variance among years. Year-specific and hyper-parameters were fit by assuming that the observed number of fish departing by date (from radio-telemetry data) were random variables drawn from a multinomial distribution, with proportions predicted by the normal departure timing model for each year. Source code for the model is presented in Table 3.1.

Emergence timing was predicted from spawn timing (departure timing) based on water temperature. The required time for incubation between fertilization (spawning) and emergence can be predicted based on the number of Accumulated Thermal Units (ATUs, Jensen et al. 1992). The required ATUs will depend on water temperature which increases over the incubation period. Thus, we predicted the ATU requirement for each potential spawning date (daily from April 1 to June 1) based on the moving average of water temperatures beginning on each date. These requirements were then compared to the actual ATUs (by date) to determine the emergence date for each potential fertilization date. The spawn-timing curve for each year was then adjusted to predict an emergence-timing curve based on the number of days required for emergence for each potential spawning date.

3.4.3 Analysis of juvenile data

Abundance for each life stage and year was estimated by the juvenile HBM. These values were then used to estimate survival rates between adjacent life stages (e.g. fall age-0+ fry to spring age-0+ parr) for each year. Linear relationships for fall age-0+ fry to spring age-0+ parr were fit to describe the average overwinter survival rate, and for abundance of 0+ parr in spring to abundance of 1+ parr the following spring to estimate the annual survival rate of parr. We fit two different linear relationships for the annual

parr survival rate owing to obvious differences in survival in odd and even years related to pink salmon abundance.

Fitting linear relationships between the abundance of one life stage and another implicitly assumes no density-dependence in survival rates. Typically, survival rate decreases with increasing density. For a more rigorous analysis, we fit a Ricker stock-recruitment relationship to predicted abundance of one life stage (e.g., spring age 0+ parr) as a function of the abundance of the previous life stage (e.g., fall fry). This model accounts for both density-dependent effects and potential flow effects on survival rates. The model we used was,

$$3.4) \quad R_t = S_t \cdot e^{\alpha + \beta \cdot S_t + \gamma \cdot X_t}$$

where R_t is the abundance of one life stage (e.g., spring age-0+) in year t , S is the estimated abundance of the previous life stage (e.g., fall fry), α is the log of the maximum survival rate when there are no density or flow effects (productivity), β is a density-dependent effect, and γ is the effect of flow covariate X_t . The product of γ and X_t therefore represents the shift in the stock-recruitment curve in log space in year t due to the value of the flow covariate in that year. As X_t is a standardized annual covariate value ($X_t = \frac{x_t - \mu}{\sigma}$), this formulation results in a base recruitment curve at the mean level of the covariate value, since the standardized value would be 0 in this case (thus $\exp(\gamma \cdot X_t) = 1$). A variety of flow covariates were computed using daily discharge records at the WSC Brackendale gauge. These include measures of rapid discharge change (maximum discharge increase and decrease over 1 day), average discharge, variation in discharge, maximum discharge, and the proportion of days discharge was $\leq 25, 40, 60, 80,$ and $100 \text{ m}^3 \cdot \text{s}^{-1}$.

Stock-recruitment models were fit to the egg deposition – fall fry (E-FF), fall fry – spring age 0+ parr (FF-P0+), and spring age 0+ parr – spring age 1+ parr (P0+-P1+) juvenile abundance data. Most covariates for the E-FF model were based on discharge values over the months of July and which is when Steelhead emerge from redds or are still very small and likely still vulnerable to flow changes. Time intervals for the rapid

discharge change covariates (upramp and downramp rates) extended up to the first date of juvenile surveys in fall (typically mid-September). This was done to capture any rapid discharge changes that occurred prior to the date when fry abundance is estimated. The covariates for the FF-P0+ model were computed based on discharge values over the fall and winter (October-March) which spans the period between juvenile surveys. The covariates for the P0+-P1+ model were calculated based on discharge values over winter (October-March) or summer (April-September) periods. The period of observation in this case is one year, and summer and winter flow covariates allow us to evaluate the relative importance of summer or winter flows on annual survival rates.

The Ricker flow covariate model was extended for the P0+-P1+ model to account for the important effect of pink salmon abundance on annual parr survival rates,

$$3.5) \quad R_t = S_t \cdot e^{\alpha + \beta \cdot S_t + \delta \cdot P_t + \gamma \cdot X_t}$$

where P_t is the abundance of pink fry at the Rotary Screw Traps, which provides an index of abundance of the enhanced food supply for steelhead parr due to Pink salmon returns.

Parameter estimates for α , β and γ (and δ for P0+-P1+ model) were estimated using a Bayesian model that assumed observations of $\log(R/S)$ fry abundance were normally-distributed random variables with means predicted by eqn.'s 3.4 and 3.5. The models accounted for uncertainty in estimates of S_t and P_t to avoid bias due to the errors-in-variable problem (independent variables in regression models are assumed to be measured without error). Observation error in $\log(R/S)$ was computed from the juvenile HBM output and treated as data in the Bayesian model to estimate the extent of process error for the stock-recruitment relationship. This provides an estimate of the true variation around the stock-recruitment curve after accounting for observation error in the dependent variable $\log(R/S)$. This process error is used in the calculation of performance measures for the WUP operating review.

Models were fit using WinBUGS (Spiegelhalter et al. 1999, Table 3.2). Uninformative priors were used for all model parameters. We ran the Markov Chain Monte Carlo (MCMC) for 50,000 iterations, discarded the first 20,000 to remove any "burn-in" effects and stored every 15th iteration to reduce autocorrelation. Three chains

were initialized from different randomly determined starting points. Convergence of the chains were visually assessed by monitoring trace plots of Markov chains for each parameter, as well as by examining the Gelman-Rubin convergence diagnostics (all \hat{R} values < 1.01).

4.0 Steelhead Life History in the Cheakamus River

This chapter provides a summary of life history characteristics of Steelhead in the Cheakamus River. This information provides the context and background to evaluate potential effects of flow regimes on freshwater production (Fig. 4.1). Different life stages will have different sensitivities to changes in flow regime (Nislow and Armstrong 2012), and it is therefore necessary to define the timing of each life stage. We pay particular attention to defining the emergence period, since post-emergent fry are small and particularly sensitive to flow variation. We summarize the freshwater and ocean age structure of Steelhead in the Cheakamus River since it determines how escapement and juvenile data is compared between IFA and WUP regimes. Steelhead is an anadromous form or morph of *Oncorhynchus mykiss*. The non-anadromous morph of *Oncorhynchus mykiss*, commonly referred to as resident rainbow trout, are also found in the Cheakamus River. These two morphs are not independent. Progeny of Steelhead parents can remain in freshwater for their entire life and conversely, progeny from resident rainbow trout can go to sea (Kendall et al. 2014). The following review of Cheakamus mykiss life history provides information on both Steelhead and resident rainbow trout as both are potentially affected by changes in flow regime. We also provide a summary of differences in escapement and juvenile Steelhead abundance in the Cheakamus and Brohm Rivers. This information is needed to determine how independent these populations are, as survival rates in Brohm River are not affected by operational changes at Daisy Lake Dam.

4.1 Arrival Timing of Steelhead

Steelhead in the Cheakamus River are classified as a winter-run stock because they return in winter and early spring, unlike summer-run stocks which return in summer and fall. We estimate arrival timing into the escapement survey area, which is the anadromous section upstream of the Cheekye-Cheakamus confluence. Arrival timing is estimated from the escapement model which incorporates counts from repeat swims as well as data from radio telemetry available in about half of the years of the 24-year escapement record. This model shows that Steelhead begin to enter the survey area as early as late-January (Fig. 4.2). Peak arrival dates typically occur in early- to mid-April. The number of spawners present in the survey area is the difference between the number

that have arrived by date and the number that have departed by date. The escapement estimation model indicates that abundance typically peaks in late-April to early-May.

4.2 Spawn- and Emergence-Timing of Steelhead

As female steelhead spend little time defending their redds after spawning, information on departure timing of female steelhead from radio telemetry provides a good measure of spawn timing. For a given date of spawning, emergence timing can be reliably predicted based on water temperature. Thus, combining information on spawn timing from radio telemetry and water temperature data can be used to estimate the emergence timing distribution. As recently emerged fish are dependent on shallow and slow-water habitat at the rivers' edge, estimates of emergence timing define periods of vulnerability to high flows (WUP hypothesis 1) or flow variation (WUP hypotheses 2 and 3).

The cumulative proportion of steelhead departing is well approximated by a normal distribution (Fig. 4.3). The date at which 50% of the tagged fish depart is the median departure date, and the steepness of the curve depends on the amount of variation in departure date among individuals (a flatter curve indicates greater variation). Hatchery fish that returned to the Cheakamus River in 2009-2011 departed later than wild fish (top-left panel), so our analysis of historical departure-timing data is restricted to wild fish only. Female Steelhead left the Cheakamus River (at the Cheekye-Cheakamus confluence) earlier than male Steelhead (top-right panel or comparison of bottom panels in Fig. 4.3). This occurs because, for a given arrival date, male Steelhead spend more time in the Cheakamus River, likely to maximize spawning opportunities (Fig. 4.4).

A hierarchical Bayesian model (HBM) of departure timing fit the radio telemetry data well, though there was considerable uncertainty in annual estimates (Fig. 4.5). Water temperature rises steadily during the spawning-incubation period and there was considerable variation in water temperature regimes in some years (Fig. 4.6, top panel). Colder years like 2011 would lead to longer incubation times and later emergence for a give spawn date. Steelhead typically begin spawning when water temperatures exceed 6° C. Spawn timing was generally similar among years, except in 2016 where it peaked two-three weeks earlier (Fig. 4.6 middle panel) likely because water temperatures by mid-April were warmer (top panel). The effect of water temperature on emergence timing is

apparent in the emergence timing curves (Fig. 4.6, bottom panel). Note the later emergence timing in years with cooler water temperatures (e.g. 2011) and earlier emergence timing in warmer years (e.g. 2009) in spite of very similar spawn timing. There was also less variation in the emergence timing (within years) relative to variation in spawn timing. This occurred because the incubation period for fish spawning later in the seasons is shorter than fish spawning earlier owing to differences in water temperature. Developmental stages of progeny from later-spawned fish catch-up to earlier-spawned fish because they are exposed to a warmer temperature regime. To some extent the average size of fry during our September surveys is related to emergence timing. When emergence is late (e.g. 2011) mean size is smaller (e.g. 38 mm) than when it is earlier (e.g. 2009, mean size 53 mm, Fig. 4.6).

Our emergence-timing curves (Fig. 4.6 bottom panel) indicated that Steelhead in the Cheakamus River typically begin to emerge in early-July (Table 4.1). Median emergence dates ranged from July 5 to August 5, and last dates of emergence ranged from July 20 to August 26. These ranges indicate that July and August are potentially flow-sensitive months that will affect survival rates of recently emerged fry (Fig. 4.7).

4.3 Freshwater Age Structure

Juvenile steelhead typically spent two and three winters in the Cheakamus River before departing as smolts (Fig. 4.8, top panel). Prior to 2011 (smolt outmigration year 2008-2009), an average of 27% of returning spawners had spent 3 years in freshwater, with the majority spending only two years. Between 2012 and 2020, an average of 53% of returning spawners had spent 3 years in freshwater. This change could be driven by an increase in the time required for juvenile Steelhead to reach a size large enough to smolt, or an increase in the marine survival rate of 3 yr smolts relative to 2 yr smolts (because scales were collected from returning spawners). Age at smoltification has been shown to be related to growth rate, with older smolts ages occurring in systems with lower growth rates caused by colder water temperatures or higher juvenile densities. Changes in marine conditions can also affect the relative survival rate of smaller (age 2 yr.) or larger (age 3 yr.) smolts. Freshwater age as determined by scales collected from spawners since 2011 indicates about ½ of the smolts leave as 3 yr olds. This is slightly more than the 44% estimate from the RST since 2008 (roughly corresponding with return period 2011 to

present). Assuming ageing of scales from smolts and returning adults are both unbiased, this difference indicates slightly higher marine survival rates for age 3 yr smolts, which is feasible given their larger size at outmigration (Table 4.2).

Steelhead fry collected during our fall surveys in September had an average fork length of 47 mm between 2007 and 2020 (Table 4.3). They grew an average of 19 mm to reach a size of 66 mm by the spring (following April session). They grew an average of 34 mm over the summer to reach an average size of 102 mm by the next fall. Winter growth of these 1+ fish was on average 7 mm, leading to an average size of 1+ fish the following spring of 108 mm. These fish grew an average of 34 mm over the summer to reach a size of 143 mm by the fall. They then grew an additional 9 mm over winter to reach an average size of 153 mm by the following spring when they would have spent a total of 3 winters in the Cheakamus River. Growth rates during summer were more than double the rates in winter owing to higher water temperatures. The average size of 2 and 3 yr smolts based on scales collected at the RST and length frequencies at the RST was 159 and 181 mm, respectively (Table 4.2). The catch of smolts at the RST peaks in early May, about one month after we measure them during our juvenile surveys as 1+ and 2+ parr. These data indicate substantive growth between April (juvenile surveys) and May (capture at RST) but also reflect differences in migratory strategy. Smaller 1+ parr in April are less likely to smolt, thus the large difference in mean size of 1+ parr and 2 yr smolts occurs in part because only the larger 1+ parr are likely to smolt. There is less of a discrepancy between 2+ parr mean size (153 mm) and the mean size of 3 yr smolts (181 mm) because most age 2+ parr are large enough to smolt.

4.4 Ocean Age Structure

Steelhead returning to the Cheakamus River have typically spent two or three winters at sea (Fig. 4.8). The ocean age structure has shifted from one being dominated by ocean age 2 yr fish prior to 2011 (61%) to one dominated by age 3 yr fish from 2011 to the present (age 2 = 34%, Fig. 4.8). This shift in ocean age structure occurred in the same year (2011) as the shift to older smolts seen in freshwater ages. On average 11% of returning spawners have already spawned at least once before (repeat spawners). Ocean age 3 yr returning spawners are larger than ocean age 2 yr spawners because of the extra

year spent growing at sea (Fig. 4.9). Thus, an increase in the proportion of returning spawners that are ocean age 3 yrs should lead to an increase in the mean size of fish that were caught, which was not the case (green line in Fig. 4.9). This occurred in part because mean size-at-age has been variable. The size of both ocean age 2 and 3 yr fish has declined slightly since 2011, resulting in very similar mean size across age classes before (762 mm) and after (780 mm) 2011.

The total age of Steelhead returning to the Cheakamus River typically ranges from 4 to 6 years old (Fig. 4.10). Mean total age of returns to the Cheakamus River averaged 4.9 years, with an obvious shift from younger (mean age 4.6 yrs) to older fish (5.1 years) beginning in 2011. These age structure data indicate that a 4- or 5-year lag is required to assign escapement from each year to production from pre-IFA, IFA, and WUP periods.

4.5 Resident Rainbow Trout

Adult resident rainbow trout in the Cheakamus River can generally be distinguished from Steelhead based on their size (Fig. 4.11). Resident trout become vulnerable to capture by angling beginning at about 4 yrs. at a mean size of ~ 40 cm (Fig. 4.12). There is considerable variation in size-at-age for resident trout relative to returning Steelhead. This may reflect variation in growth among individuals as well as increased error in age determination (see appendix A1 of Korman and Schick 2017). Abundance of adult resident trout (> 40 cm) is relatively low compared to the escapement of returning Steelhead spawners (Fig. 4.13). Both resident trout and Steelhead showed a sudden increase in abundance beginning in 2010. This occurred four years after implementation of WUP flows in 2006. Based on ageing, it would take a minimum of four years for juveniles rearing under flows in 2006 to be large enough to be counted during swim surveys and be part of the abundance estimate in 2010. A four-year lag would also be required for Steelhead given a minimum smolt age of 2 yrs. and a minimum ocean of 2 yrs. As discussed below, it is uncertain whether the change in flow regime in 2006 caused the increased abundance which began in 2010. 2006 was also the first year after the CN caustic soda spill. The spill would have resulted in much lower densities of sculpins and resident char and older trout which are predators of juvenile mykiss. Reduced competition of older conspecifics (older mykiss) may also have increased growth and

survival rates. Both of these factors could have led to an increase in the proportion of juvenile mykiss that adopt a resident life history strategy (Kendall et al. 2014).

Some resident rainbow trout that were captured and radio tagged in the Cheakamus River did not spend their entire life in the Cheakamus River. We radio-tagged a total 51 individuals captured in 2016 or 2017. Ten of these tagged trout were detected at receivers in the Squamish River downstream of the Squamish-Cheakamus confluence, and a tagged trout was caught by an angler in the Mamquam River. Thus, a minimum of 20% of resident trout in the Cheakamus River make some use of the Squamish River over their lifetime.

Trends in abundance and survival of juvenile mykiss in the Cheakamus are estimated as part of MON-3 to make inferences about effects of flow and other factors on Steelhead. However, inferences about effects on Steelhead are potentially weakened if a large component of the juvenile mykiss population originates from resident rainbow trout. Fortunately, the data indicate that resident rainbow trout only make a modest contribution to the abundance of juvenile mykiss in the Cheakamus River. The average escapement of Steelhead over the 17 years when scale information has been collected (to unequivocally distinguish Steelhead and resident trout) was 510 fish, compared to 108 resident rainbow trout (Table 4.4). Of 153 resident trout that have been captured by angling and sexed between 2012 and 2020, 59% have been males. There is no indication that the resident population in the Cheakamus River is dominated by males as in more interior systems (e.g. Thompson River). Owing to lower abundance and fecundity of resident rainbow trout compared to spawning Steelhead, their estimated contribution to egg deposition averaged only 3% of the total egg deposition for mykiss and there is no indication of an increasing trend in the contribution of eggs from resident trout (Table 4.4).

Otolith microchemistry of juvenile mykiss indicated that 84% and 96% of juvenile mykiss collected in the Cheakamus and Brohm Rivers in spring 2009 had Steelhead mothers, respectively (Korman et al. 2010a). Within the Cheakamus River, only 45% of 11 juveniles sampled upstream of Culliton Creek had an anadromous female parent, compared to 94% of 4 individuals in the reach between Culliton and Cheekye confluences, and 100% of 16 individuals in the reach downstream of the Cheekye

confluence. Thus, the resident morph was more common upstream of Culliton Creek in the Cheakamus River and very rare in Brohm River. The otolith estimates were sampled from fry and parr in 2009 and therefore reflect the contribution of residents and steelhead across the 2008 and 2009 brood years. The estimated contribution from egg deposition estimates (Table 4.4) suggests a much more limited contribution of residents to the juvenile population (~3%) compared to the otolith analysis. However, samples from the otolith analysis were from a systematic upstream-downstream survey design. The majority of the juvenile steelhead population is located downstream of Culliton Creek, and otolith- and egg deposition-based Steelhead/resident trout ratios are in agreement if otolith data upstream of Culliton Creek are excluded. This justifies excluding resident trout from stock size estimates used in stock-recruitment analysis for Steelhead.

4.6 Contribution of Brohm River

Brohm River is a tributary of the Cheekye River that flows into the Cheakamus River (Fig. 1.1). Some of the Steelhead returns we count in the Cheakamus River will spawn in Brohm River, and some progeny from this spawn will reside in Brohm River for a prolonged period of time and therefore not be influenced by dam operations. Hence a basic understanding of the proportion of the Steelhead using this tributary is relevant to flow decision-making for the Cheakamus River. Based on radio telemetry an average of 6.5% of Steelhead spawners tagged in the Cheakamus River eventually moved into Brohm River to spawn. This estimate may be low since most Steelhead were tagged upstream of Cheekye confluence. Challenges with interpreting data from bi-directional radio telemetry antennas at the confluence may have also led to error in the estimates of the proportion of the Steelhead population that spawn in Brohm River. The population of juvenile mykiss in Brohm River, which is almost exclusively Steelhead (as determined by otolith microstructure), is about 10% of the size of the Cheakamus population. The relative size of Cheakamus and Brohm River juvenile populations was pretty consistent across life stages and reasonably close to our estimate of the percentage of spawners using Brohm River (6-7%), especially because we consider the latter value to be an underestimate. There was no indication that Brohm River substantively contributes to the juvenile Steelhead population in the Cheakamus River via movement. Survival rates from

the age-0+ fry in the fall to age-0+ parr in spring was 24% in the Cheakamus and 16% in Brohm (Table 4.5). The lower apparent survival rate in Brohm River could reflect higher mortality or some outmigration of Steelhead into the Cheakamus River over their first winter. Annual survival from age-0+ parr in the spring to age-1+ the following spring was 32% and 47% in Cheakamus and Brohm Rivers, respectively. Assuming that survival rates in Brohm River are not much higher than those in the Cheakamus River, the higher survival rate in Brohm River indicates that few parr outmigrate from Brohm River. These data suggest that freshwater production from these systems can be treated as largely independent.

5.0 Effects of IFA and WUP Flow Regimes on Freshwater Production as Inferred from Escapement and Smolt Production Data

The historical escapement trend for Steelhead in the Cheakamus River can be used to make inferences about the effects of flow on freshwater production and to compare production under pre-IFA, IFA and WUP flow regimes. This time series has been affected by three different flow regimes (Fig. 5.1, Table 5.1). Adult returns were low (average 170) in years when the juveniles that produced these returns reared in freshwater prior to the Interim Flow Agreement (the pre-IFA period as characterized by returns from 1996-2001). The average escapement was more than twice as high under IFA flows prior to the CN sodium hydroxide spill (385, escapement from 2002-2007) and this increase was statistically significant ($p=0.002$). Wild-origin escapement declined over two consecutive years for returns produced from juveniles that were present in the river during the spill (231, escapement in 2008, 2009), but this decline was not statistically significant ($p=0.063$) owing to low sample size ($n=2$). The average escapement since 2010, which was produced from juveniles which reared in the river under WUP flows, was 1.5-fold higher (578) than during the IFA pre-spill period (383), and this increase was statistically significant ($p=0.008$).

Higher Steelhead production in the Cheakamus River during the WUP period was also seen in the stock-recruitment analysis (Fig. 5.2). On average, brood years that reared in the Cheakamus River under WUP flows had recruitments that were two-fold higher than those rearing in the river under pre-IFA or IFA flows. These WUP stock-recruitment curves were estimated by multiplying predictions from the Beverton-Holt model in pre-WUP years by e^γ (see eqn 3.2). That is, e^γ is the estimated magnitude of the shift in the stock-recruitment curve for broods influenced by WUP flows. A similar adjustment was made using the Ricker model (the γ term is added to the log of productivity, α) which results in stock-recruit curve which has a steeper initial slope (greater productivity) and higher carrying capacity. These γ estimates indicate that the Beverton-Holt stock-recruit curve has increased by about 1.6-fold, and productivity of the Ricker model has increased

by about 2.1-fold during the period influenced by WUP flows. The stock-recruitment analysis corrects for potential stock size effects and indicates that there is no confounding effect of limitations in stock size (or ‘seeding rate’ as indexed by escapement or egg deposition) on lower levels of recruitment seen under the IFA flow regime. The stock-recruitment curves indicate that escapements during that period, while substantively lower than under the WUP regime, were large enough to not be limiting for adult returns.

Changes in escapement and stock-recruitment relationships over time are affected by trends in both freshwater and marine survival (Fig. 4.1). Thus, differences in escapement between pre-IFA, IFA, and WUP periods, or differences in stock-recruitment relationships, are not solely caused by changes in flow. In the absence of juvenile data collected under each of these periods, our goal here is to use differences in Steelhead escapement in the Cheakamus River, as effected by flows under these three regimes, to make inferences of their effects on freshwater survival rates. To do this we need to remove effects of marine survival and non-flow related effects on freshwater survival. We used Steelhead smolt-adult survival rates from the literature for the marine survival correction. Kendall et al. (2017) compiled smolt-adult survival rates for hatchery and wild Steelhead from a large number of systems on the Pacific west coast. They found that survival trends varied considerably among regions with higher correlations among regions that were closer together. We therefore used their Puget Sound-Keogh River winter-run time series, which is closest in geographic proximity to the Cheakamus River, to represent survival rates for the Cheakamus River Steelhead population. Although the year-to-year correlation between the Kendall et al. marine survival index (by smolt outmigration year) and Steelhead escapement to the Cheakamus River two years later was poor ($r^2=2\%$), the timing of the increase in marine survival (higher values associated with return year 2010) aligns well with the sudden increase in escapement (Fig. 5.3). Given this pattern, we used these survival rates to correct escapements produced under pre-IFA, IFA (pre-spill), and WUP periods. The marine survival correction applied to each return year was a weighted average of estimates from smolt outmigration years two and three years earlier, with the weights based on the proportion of ocean age two- and three-year fish in each return year. The adult return year range for the pre-IFA period was 1998-2001, and the annual marine survival corrections are based on outmigration years

1995-1999. For the IFA period, the adult return year range was 2002-2007 (4 years after IFA flows were implemented in 1998), and the annual marine survival corrections are based on outmigration years 1999-2005. The return year range for the WUP period was 2010-2019, corresponding to an outmigration year range of 2007-2017. The last smolt year included in the Kendall et al. time series is 2014, so the average marine survival rate during the WUP period is based on 8 years of data (2007-2014). An alternate approach is to replace the Puget-Keogh time series from Kendall et al. with the Keogh time series, which includes one additional year (smolt year 2015). The correlation (r^2) between the Puget-Keogh and Keogh-only time series over outmigration years 1999-2014 was only 28%. We therefore present results using both indicators of Cheakamus Steelhead marine survival rates.

The average Puget Sound-Keogh smolt-adult survival rates were 1.2% and 1.6% over IFA and WUP outmigration periods, respectively (Table 5.2). If the change in this index accurately reflects the change for Cheakamus Steelhead marine survival, it indicates marine survival increased by 1.3-fold under the WUP regime relative to the IFA regime. As a result, the 1.5-fold increase in escapement under WUP flows relative to IFA flows must be reduced by 1.3-fold, resulting in a WUP/IFA marine survival-adjusted ratio of 1.2. This estimate suggests that freshwater production of Steelhead in the Cheakamus River increased by 20% under the WUP flow regime. We repeated this calculation using smolt-adult survival rates from the Keogh River only which are available through 2015 (smolt year). Results using a marine survival adjustment from the Keogh River were very similar. A similar set of computations comparing pre-IFA and IFA periods resulted in an IFA/pre-IFA marine-survival adjusted escapement ratio of three, which indicates that freshwater production increased by more than 200% under the IFA flow regime relative to the pre-IFA regime. Adjusted escapement differences should be considered uncertain as the actual marine survival adjustment for Cheakamus River Steelhead is not measured.

Marine survival-corrected escapement ratios provide an index of the extent of change in freshwater survival rates in the Cheakamus River. However, this index does not separate flow- and non-flow effects. Annual survival rates of Steelhead parr in the Cheakamus River were four-fold higher in years when pink salmon returned in large

numbers (see Chapter 6). As described below, we used a Ricker covariate model to estimate the effect of pink salmon (as indexed by RST-estimated pink fry abundance, Table 5.3) on annual parr survival rates. We then averaged these rates over IFA and WUP to calculate pink salmon adjustments for WUP and IFA periods (Table 5.2). We were unable to incorporate a pink salmon adjustment for the pre-IFA and IFA comparison as reliable estimates of pink salmon abundance are not available for the pre-IFA period (Fig. 5.4). If pink salmon returned in roughly equal numbers during IFA and WUP periods, a pink salmon adjustment to the WUP/IFA escapement ratio would not be required. However, a number of different data sources indicate that pink salmon returns were much higher during the WUP period. The RST program on the Cheakamus River has provided a very reliable index of the run size of outmigrating pink salmon fry since 2002, which presumably reflects the escapement of pink salmon in the previous calendar year. This index shows a much higher abundance of pink salmon during the WUP period compared to the IFA period (Fig. 5.4, Table 5.3). A very similar pattern is seen in the Coquitlam River, suggesting that higher pink salmon returns are caused by an increase in marine survival that is common to both rivers, rather than the unlikely scenario of simultaneous increases in freshwater production in both rivers. Changes to the flow regime in the Coquitlam River in 2000 and again in 2008 may have resulted in higher pink salmon returns in 2011 and later years (as indexed by higher fry numbers in 2012 and later). However, the increase in pink salmon fry production has not been attributed to the change in flow regime (Schick 2015), as pink salmon returns have increased in many other systems, including the Squamish watershed as a whole (Fig. 5.4). Thus, there is pretty strong support for applying a pink salmon correction to adjust the WUP/IFA escapement ratio for Steelhead in the Cheakamus River. Owing to higher pink returns during the WUP period (Table 5.3), the WUP/IFA pink salmon adjustment was 1.49 (Table 5.2). The marine survival- and pink salmon-corrected WUP/IFA Steelhead escapement ratio in the Cheakamus River was 0.8, which implies there has been a 20% reduction in juvenile Steelhead production under the period effected by WUP flows relative to IFA flows (Table 5.2). This estimate should be considered preliminary owing to uncertainty in both pink salmon and marine survival adjustments.

The Rotary Screw Trap program estimated the abundance of Steelhead smolts migrating past the trap for three years during the IFA period and nine years during the WUP period that were not affected by the CN spill (2009-2017). Estimates in the spring of 2006 and 2007 (first two years of WUP) were not available because steelhead smolt numbers were very low due to the spill and no fish were marked due to conservation concerns (Melville and McCubbing 2012). The estimate of smolt abundance in 2008 was the highest on record, and this spike is thought to have occurred due to low numbers of predators and reduced competition from conspecifics (other juvenile Steelhead) due to the spill. This likely led to very high survival rates for juveniles produced from the spawners returning in the first year after the spill (2006). Mean Smolt abundance during the IFA period was ~7,400 compared to ~6,200 during the 2009-2017 period representing non-spill effected WUP years (Fig. 5.5). This 17% decline during the WUP period was not statistically significant ($p=0.28$). The estimated decline in smolt production under the WUP (-17%) was similar to the decline based on the escapement analysis (-20%). If the WUP-average smolt production is reduced to account for higher pink salmon abundance during the WUP period (by dividing the WUP average by 1.49, see Table 5.2), the decline in freshwater production increases from -37% to -44%. These smolt-based IFA-WUP comparisons should be treated with caution owing to low sample size during the IFA period ($n=2$) and considerable uncertainty in annual estimates due to sampling challenges (low numbers caught combined with low capture probability).

The trend in abundance of resident rainbow trout generally followed the trend seen for Steelhead (Fig. 5.6). Abundance of resident trout produced under the IFA flow regime was 5-fold higher than abundance produced under pre-IFA flows and this difference was statistically significant ($p=0.003$). Abundance of resident trout produced under the WUP flow regime was 2-fold higher than under the IFA regime and this difference was also statistically significant ($p=0.041$). Resident trout abundance increased beginning in 2010. As the minimum age of resident trout that are counted during swim surveys is about 4 years, this increase is perfectly aligned with the switch to the WUP flow regime in 2006. However, 2006 was also the first year of spawning after the CN caustic soda spill. Higher growth rates due to lack of predators and reduced competition may have led to an increase in the proportion of Steelhead progeny that switched to a

resident life history, as seen in other systems (Kendall et al. 2014). Other factors may have also led to higher growth rates promoting a shift to a resident life history. The amount of input of phosphorous from the Whistler sewage treatment plant increased substantially in late 2009 owing to a change in their treatment process (Fig. 5.7). We speculate that this increase may have affected algal and benthic invertebrate production in the Cheakamus River, which may in turn have increased juvenile growth rates and the probability of a resident life history. Thus, it is uncertain whether the increase in resident trout abundance during the WUP period was caused by the change in flow regime owing to the confounding effects of the CN caustic soda spill and increased phosphorous loading.

6.0 Effects of Flow and Other Factors on Survival Rates of Early Life Stages of Steelhead

6.1 Juvenile Abundance and Survival

Reliable abundance estimates for juvenile life stages of Steelhead in the Cheakamus and Brohm Rivers are available beginning in fall of 2008. Although visual examination cannot distinguish whether these juvenile fish originated from Steelhead or resident trout parents, or will adopt anadromous or resident life histories, demographic and microchemistry analyses indicates that the majority of mykiss juveniles in the Cheakamus River are born from Steelhead parents and will become Steelhead (see Chapter 4). Thus, we refer to mykiss juveniles as Steelhead. Abundance of recently emerged Steelhead fry is quantified in September from electrofishing surveys. The average abundance of fry in the fall across study years (2008-2020) was 208,000 (Fig. 6.1a). We quantify the abundance of age-0 parr the following spring, which are the survivors of the fall fry (Fig. 4.1), through a combination of electrofishing and snorkel surveys. The average abundance of this life stage across study years (2009-2020, no sampling in 2019) was 49,500. We are unable to quantify the abundance of 1+ parr in the fall as we cannot conduct snorkel surveys due to high turbidity, and electrofishing results in a substantial underestimate of their abundance because the gear is only effective in a very limited range of habitats. However, we can provide reliable indices of abundance of 1+ and 2+ parr in the spring (2009-2018, 2020), whose abundances averaged 17,600, and 2,700 across study years, respectively. Abundance estimates are also available for Brohm River through 2017 (Fig. 6.1b).

Survival rates between juvenile life stages are computed based on the ratio of abundances across successive stages (Fig. 6.2). For the Cheakamus River we can also compute an egg-fall fry survival rate using annual estimates of Steelhead egg deposition derived from data from the escapement monitoring program. The abundance of age-0+ parr in the spring increased with the abundance of fry the previous fall (Fig. 6.3 top). This highly linear relationship ($r^2=0.87$) indicates that survival rate is relatively consistent across years and that there is a limited effect of density on survival rate. If the latter effect

was strong, the data should show a smaller increase in spring age 0+ parr abundance at higher levels of fall fry abundance. That is, the stock (fall fry) – recruitment (spring age 0+ parr) relationship should bend over at higher fall fry densities. A more rigorous evaluation of density-dependence is provided below, which confirms results from this simpler graphical analysis of the data. The slope of the relationship represents the average survival rate between fall fry and spring age-0+ parr, was 0.23.

There were also linear relationships between the abundance of age-0+ parr in the spring and age-1+ parr one year later (Fig. 6.3 bottom) indicating limited evidence of density-dependence mortality between these life stages (see analysis below for a more rigorous evaluation). For a given age-0+ abundance, age-1+ abundance the following year was on average three-fold higher in odd years than in even years. Annual survival rates (slopes) were 0.15 and 0.49 in even and odd years respectively. Higher survival rates of parr in odd years was likely driven by very high pink salmon returns, which in the Cheakamus and South Coast rivers, occurs in odd years only. The two years with the highest annual survival rates were also the ones with the largest two pink salmon return years, providing additional evidence for a pink salmon-Steelhead survival linkage (Fig 6.2, top-right panel). Age-0+ parr in spring transition to age-1+ parr by the following fall. These fish are large enough by fall (Table 4.3) to consume pink salmon eggs. In odd years with high pink salmon returns, 1+ parr have very high condition and their bellies are often distended from the consumption of large numbers of eggs. It seems likely that the availability and consumption of this lipid-rich food source (Gerig et al. 2017) leads to increases in their survival over the winter. Fall fry are too small (45-65 mm) to consume eggs, and this is apparent in their condition during fall surveys. As a result, we do not see an odd-even year pattern in survival from fall fry-spring age-0+ parr (Fig. 6.3 top).

To clarify the relationship between annual parr survival rates and pink salmon abundance, we used a Ricker covariate model to explain variation in the log of the annual survival rate as a function of both spring 0+ parr abundance (a density-dependent effect) and an index of pink salmon abundance. Estimates of pink fry abundance in the spring at the Cheakamus RST were (Table 5.3) were used to index pink salmon abundance the previous fall (odd year had pink fry abundances of zero). We evaluate four alternate models that included only a density-dependent effect, density and pink salmon effects,

the later model with a density-pink salmon interaction term, and a model with only a density effect and the density-pink salmon interaction term (Table 6.2a). The interaction term allows us to evaluate the hypotheses that high pink salmon abundance may only increase survival rates of Steelhead parr in years when juvenile Steelhead densities are high. The model that included only a density-dependent effect (model $b_0 + b_1 * S$ in Table 6.2) explained 20% of the interannual variation in log of the annual parr survival rate and had the weakest out-of-sample predictive power (highest AIC). In contrast, the model that included a pink salmon effect ($b_0 + b_1 * S + b_2 * \text{Pink}$) explained 70% of the variation in log survival rate and had much better predictive power (Fig. 6.4 top). The best model we evaluated included a density and density-pink interaction term (Fig. 6.4 bottom). It explained 74% of the variation in log survival and had the lowest AIC. Based on AIC scores, all models that include a Pink salmon effect are statistically indistinguishable, but are statistically much better predictive models than the model that does not include a pink effect. The extent of density-dependence in annual survival rates is somewhat uncertain as seen by the wide 95% confidence limits in the predicted 1+ parr abundance at the average pink fry abundance (Fig. 6.5). There is particularly large uncertainty at the upper end of age 0+ parr abundance where the mean relationship is very flat. If evidence for density-dependence was strong, higher mortality at earlier life stages predicted by flow covariate models (shown below) would be partially or largely mitigated by reduced density dependence at the parr stage. The data indicate that high levels of compensation appears unlikely, or are at least highly uncertain. Pink salmon corrections for IFA and WUP periods based on the Steelhead parr survival – density + pink salmon model were 1.02 and 1.52, resulting in a net WUP/IFA adjustment of 1.49 ($1.49 = 1.52 / 1.02$, Table 6.2b).

6.2 Effects of Flow on Juvenile Abundance and Survival

We evaluated effects of flow in the Cheakamus River on juvenile Steelhead survival rates using a Ricker stock-recruitment model that included both density and flow covariate effects (eqn. 3.4). We examined 10 alternative flow covariates which included covariates to capture rapid increases and decreases in flow that would cause displacement

and stranding. These were quantified using the maximum increase (upramp) or decrease (downramp) in discharge over one day ($\text{m}^3 \cdot \text{s}^{-1} \cdot \text{day}^{-1}$). Other annual statistics we examined include the average flow (Avg_Q), the standard deviation in flow (SD_Q), the maximum flow (Max_Q) and the proportion of days with flows $\leq 25, 40, 60, 80,$ and $100 \text{ m}^3 \cdot \text{s}^{-1}$ (Prop_Hrs<x).

For the egg-fall fry model, all 10 flow covariate statistics were computed between July 1 up to the first date of sampling for fall fry in September (early September). We also fit a model without any covariate effect as a baseline to judge potential improvements in predictions by including flow covariates. Finally, we fit a model without flow or density effects to evaluate evidence for density dependence in survival rates. We compared the fit of models and their predictive ability based on differences in the proportion of variance in log of egg-fry survival rate that is explained, and by differences in the Deviance Information Criterion (DIC), respectively.

The downramp model provided the best fit to the egg-fall fry data and explained 65% of the variation in the log of egg-fry survival rates (Table 6.3a, Fig. 6.6). This model had a DIC value that was more than 5 units lower than the next best model (upramp). The γ coefficients for both downramp and upramp models were negative, indicating that the greater the absolute change in discharge per day, the lower the survival rate. There was considerable uncertainty in the magnitude of the flow effect as seen by the relatively wide credible intervals (LCL and UCL in Table 6.3a), however there was a very low probability of $\gamma > 0$. That is, although we are uncertain about the magnitude of the negative effect of high ramping rates, the statistics indicate that we can be confident that the effect is negative. However, note that the estimated downramp effect was almost completely driven by the very high downramp rate in 2010 (Fig. 6.6, top-right panel). If the 2010 data point is removed, the downramp model is not any better than other flow models. The model predicts a substantive negative effect of the 2010 downramp on fall fry abundance (Fig. 6.6, panels in right column), but very modest effects over the range of maximum summer downramps observed in other years between 2008 and 2020.

The second-best egg-fry survival model predicted that survival rates increased with an increase in the proportion of time summer flows were less than 40 cms ($r^2=0.52$, $\Delta\text{DIC}=4.7$, Table 6.2a). Thus, there is modest support for the hypothesis that limiting

high flows in summer to 38 cms (a minimum flow for rafting as specified in current WUP flow order) will improve egg-fry survival rates. Egg-fry stock-recruit models based on other flow covariates explained much less variation in survival rate ($r^2=0.27-0.44$) than the downramp model, and had much weaker predictive ability ($\Delta DIC \geq 5.5$, Table 6.3a). There was weak evidence for density dependence as the model without a density effect (covariate = no flow or density effects) had a DIC value that was 1.1 units higher than the model with a density effect (Covariate =no flow but density effect). Limited density dependence is also supported by the wide 95% credible interval in predicted fall fry abundance at higher egg depositions (Figure 6.6 top-left panel).

There was less evidence of density-dependence in survival rates between fall fry and age 0+ parr in spring (Fig. 6.7, top-left panel) as seen by the relatively straight Ricker model (consistent with the linear model in Fig. 6.3) and tighter credible interval. Here the model with a density effect had a DIC value that was 1.2 units higher than the model without a density effect (Table 6.4b). As for the egg-fall fry model, the downramp rate was the best flow covariate model and explained 70% of the variation in the log of survival rates between these life stages (Table 6.4b). This model had very low probability that $\gamma > 0$, indicating relatively high confidence that rapid downramps have a negative effect on survival rates during winter. Unlike the egg-fall fry downramp model the fall fry- 0+ spring parr model was not dependent on a single year (Fig. 6.7 top-right panel) and correctly predicted the direction of the survival rate response for most years (Fig. 6.7, top-left panel).

In contrast, annual survival rates from 0+ parr in spring to 1+ parr the next spring were generally insensitive to flow covariates calculated during winter or summer periods (Table 6.4c). There was a substantial increase in predictive ability by including a pink salmon effect in the Ricker stock-recruitment relationship, with the r^2 increasing from 0 (no flow, density, or pink effects) to 0.6 (no flow or density effects but pink effect). There was very limited evidence for density dependence as the model which included both pink and density effects had a ΔDIC that was 2.3 units higher than the model with only a pink effect (1.9). This is also consistent with the least-squares based model (Fig. 6.5) which shows considerable uncertainty in predicted abundance at higher levels of age 0+ abundance (Fig. 6.5). Adding flow covariates to models with pink and density effects led

to increases in explained variance of 0.15 or less. Three models, including two flow-based ones, had Δ DIC values <2 , and were therefore statistically indistinguishable. In addition, the flow covariates that were selected generally did not support logical hypotheses. The Prop_Hrs \leq 25 cms_Summer model predicted an increase in parr survival rate with an increase in the proportion of time that discharge was $\leq 25 \text{ m}^3 \cdot \text{sec}^{-1}$ (Table 6.4c, Fig. 6.8). While this pattern may be possible for smaller fry that prefer shallow and low velocity habitat that would be more available at low discharge, it seems unlikely for larger parr that can make use of habitat with deeper and faster water. The best model (Prop_Hrs \leq 60 cms_Summer) predicted negative density-dependence (higher survival rates at higher densities), which is unlikely and indicates that the correlation with this flow covariate is probably spurious.

Our findings that higher downramp rates reduce survival rates of early life stages of Steelhead in the Cheakamus River, and the lack of substantive flow effects on parr survival, is consistent with finding in other systems (as reviewed by Nislow and Armstrong 2012). Fry have more limited swimming abilities than parr and are more vulnerable to predation. As a result, they utilize shallow habitats near the shoreline that are destabilized by rapid changes in flow. The larger size and mobility of parr provides a greater ability to control their energetic status and habitat use. As a result, parr are much less likely to experience direct mortality from extreme flood events, high discharge, or rapid flow changes. This pattern is consistent with observations during the August 2018 ramping study (see next section), where many fry but few parr were stranded. In general, fish population studies have shown that biotic and abiotic factors effect survival of early life stages like fry, and growth and movement in later life stages like parr (Nislow and Armstrong 2012). This may explain why our study showed little interannual variation in survival rates for parr except for the food-mediated effect of pink salmon.

We used the egg-fry and fry-spring 0+ parr flow covariate models to evaluate differences between pre-IFA, IFA, and WUP regimes using historical daily flow data from 1968 to 2018. To do this, we calculated the flow covariate statistics at the Brackendale gauge over summer and fall-winter in each year. We then used these historical covariate values in the juvenile stock-recruit models to calculate average spring 0+ parr abundance in each year assuming an average egg deposition (the same in all

years). Predictions are based on a model-weighted average (ModWgt column in Tables 6.4a and b), but since downramp rate dominates the weighting, results largely depend on differences in maximum daily downramp rates among regimes. The model predicted that there was considerable variation in 0+ parr abundance across years within flow regimes, with the greatest variation seen during the pre-IFA period (Fig. 6.9). The model predicted that the median parr abundance across years was slightly higher under the WUP regime compared to the IFA regime. However, these differences were modest considering the high interannual variability in predictions among years within flow regimes. This result is not consistent with differences in estimates of freshwater production across regimes (from escapement- and smolt-based analyses) which suggest production declined under the WUP regime

6.3 Preliminary Ramping Study Results

This section provides a brief summary of results obtained from an assessment of the effects of the August 20th, 2018 rampdown event on salmonid fry populations in the anadromous section of the Cheakamus River. This work was funded through a new ramping study initiated just prior to this event. Information from this survey was originally presented in a memorandum from Ecometric Research to BC Hydro on August 30, 2018. This chapter summarizes key elements of this memorandum and discusses it in the context of results from the egg-fall fry flow covariate models reviewed in section 6.2. The rampdown event that occurred on August 20th, 2018 was pretty typical of rampdown events under the WUP during August in years when inflows are average or low (Fig. 2.4b). Flows were decreased from ~ 40 to $20 \text{ m}^3 \cdot \text{sec}^{-1}$ over about two hours. At the Brackendale gauge site on the Cheakamus River, the flow change resulted in a rate of stage change of $8.1 \text{ cm} \cdot \text{hr}^{-1}$ which is about 3-fold higher than the DFO guideline rate of $2.5 \text{ cm} \cdot \text{hr}^{-1}$. The rampdown rate that was used is specified in the WUP flow order for Daisy Lake Dam and occurs in most years due to the change from the summer minimum flow objective of $38 \text{ m}^3 \cdot \text{sec}^{-1}$ to a minimum of $20 \text{ m}^3 \cdot \text{sec}^{-1}$ that normally occurs in mid- to late-August.

The percentage of the pre-rampdown steelhead fry stranded and killed at six cobble bar sites due to the rampdown ranged from ~ 2 -33% with an average of 10% (see

Table 1c of Korman et al. 2018). A synoptic survey conducted on the day of the rampdown identified 29 locations where fish were found in isolated pools or recently dewatered pools or braids (see Table 2 of Korman et al. 2018). Visual and unverified estimates of the total number of fish located in these pools ranged from about 3,500 - 35,000. A total of 1,254 juvenile salmonids were captured in isolated pools across the two days of salvage efforts, of which 90% were Steelhead fry and 10% were Coho fry. Electrofishing is very challenging in these settings owing to high fish densities. 7% and 4% of the Coho and Steelhead fry captured in the isolated pools died due to electrofishing injuries. Capture probability at these sites was very low, and therefore that only a small proportion of the total population in each pool could be salvaged.

Evaluating the population-level consequences of stranding events is very challenging, and the sampling effort around the August 20th, 2018 rampdown was modest. Nevertheless, this study provided some useful results. The most surprising perhaps is the large number of fish lost from what is a modest ramp (only a $20 \text{ m}^3 \cdot \text{s}^{-1}$ flow change) relative to other WUP years (Fig. 2.4). We estimated that 10% of the steelhead trout fry population in low angle cobble bars that we sampled died due to the August 20th, 2018 ramping event. However, these sites were not randomly selected and they represented habitats where we would expect the highest rates of bar stranding (very wide and low angle bars). Thus, extrapolating this mortality rate to the entire population in the river would very likely lead to a substantial overestimate of overall loss. There is even greater uncertainty with respect to the estimate of fish loss from the synoptic survey, where the range is ~3,500 to 35,000 fish.

The Ricker flow covariate model analysis provides a means of characterizing the effects of ramping events on egg-fry survival rates. Based on the July-August hydrograph in 2018, the downramp model predicts it was relatively benign compared to other WUP years (top-left panel in Figure 6.6). If a minimum of 10% of the steelhead fry population was lost due to the August 20th, 2018 ramping event, losses in other WUP years are expected to be higher.

At this point it is not at all clear whether slower ramp rates will reduce the stranding impacts like those observed on August 20th, 2018. We suspect a prolonged period of higher and steady flows (of $\sim 40 \text{ m}^3 \cdot \text{hr}^{-1}$), which occurred for a month prior to

the August survey in 2018 (see Fig. 1.3 in Korman et al. 2018), would result in small fry colonizing areas that are then dewatered when the flow is reduced. Fry in these locations are particularly vulnerable to being stranded because the distance to the main channel is often considerable, and the topography between these locations and the main channel is often uneven and complex. These areas may always strand young fish even under slower ramping rates. If this is the case, the flow regime prior to a ramping event may be as important as the characteristics of the ramping event itself, as seen from the long-term stranding study in the lower Columbia River (Golder 2016). The minimum flow may also be an important factor controlling stranding losses regardless of ramping rates, with higher losses occurring at lower minimum flows which expose a higher proportion of these vulnerable habitats. The recently developed Telemac model may be useful in identifying flow thresholds where stranding impacts are more likely.

7.0 Conclusions

This project was designed to address the following questions about effects of the WUP flow regime on the Steelhead population in the Cheakamus River:

1. Do high flows in July and August negatively affect Steelhead fry that have recently emerged?
2. Does flow effect juvenile production, as indexed by the number of fry, parr, smolts, and returning adults?
3. Has the current WUP flow regime led to changes in Steelhead production, as indexed by adult returns, juvenile abundance, and smolt production?

There are multiple lines of weak evidence which suggest that the WUP flow regime has the potential to limit Steelhead production in the Cheakamus River (Figure 7.1). There is considerable uncertainty about whether juvenile production has declined under the WUP flow regime relative to the previous IFA regime. However, there is some evidence that suggests that production can be improved by reducing impacts associated with rapid decreases in flow.

First, we estimated that freshwater production under the WUP regime was ~20% lower than production under the IFA regime based on the escapement record after accounting for differences in marine survival, and pink salmon effects on freshwater survival rates of parr over these periods. Second, the record of smolt abundance estimates from the Rotary Screw Trap indicate that smolt production was on average 17% lower in WUP years relative to IFA years, and 44% lower if one applies the pink salmon correction used for the escapement analysis. Escapement- and smolt-based estimates of changes in freshwater production under the WUP regime (relative to IFA) should be considered uncertain. In the case of the escapement-based estimate, there is uncertainty in the assumption that differences in the Puget Sound – Keogh River Steelhead marine survival rate between WUP and IFA periods accurately reflect the differences for the Cheakamus population. The pink salmon correction for the IFA period should also be considered uncertain because only three years were used in its calculation, thus the

correction may misrepresent the pink effect for the longer period over which that regime was in place when escapement data were available. In the case of the smolt-based estimate of WUP vs. IFA freshwater production change, there are only a limited number of samples during the IFA period (2), and annual estimates, especially during the IFA period, were very imprecise.

Third, the Steelhead egg - fry and fall fry - spring age 0+ parr flow covariate models indicate that rapid decreases in discharge can potentially have a substantive negative effect on the population. Models that included these flow effects provided a statistically significant improvement in fit relative to models that did not include them. However, these models should not be considered very reliable, especially the egg-fall fry model, which is totally dependent on one year which had very rapid downramp rates in summer. Stock-recruitment relationships based on only 13 (egg-fall fry) or 11 (fall fry – 0+ spring parr) years of data are uncertain, especially when the relationship includes an extra parameter to assess flow effects. We simply do not have enough observations, and observations over an informative range of flow levels, to quantify flow and density effects more precisely.

Fourth, a ramping study in August 2018 estimated that 10% of the Steelhead fry using low angle cobble bars that we sampled were killed from a single ramping event, and a higher percentage could have been killed from this event when losses in braids and side channels are considered. The cobble bar loss is moderately uncertain because we are unsure how representative the habitats we sampled were relative to the total area that fry are distributed over. The loss in isolated pools and braids is more uncertain owing to the imprecise way that losses were estimated in the field. While it is clear that thousands of Steelhead fry were killed by this single ramping event, this scale of loss represents a small fraction of the total fry population (averaging 200,000 between 2008 and 2020, and 175,000 in 2018).

Potential negative effects of the WUP flow regime on Steelhead production might be expected based on studies in other systems with high ramping rates, and the fact that WUP ramping rates on the Cheakamus River are ~4- to 7-fold higher than the guidelines from Fisheries and Oceans Canada. Recently-emerged fry are very small, thus suitable territories needed for feeding and avoiding predation are limited to microhabitats with

very shallow depth and low velocity. (Armstrong and Nislow 2006). In moderate-sized rivers like the Cheakamus, these microhabitats are only found in the immediate nearshore areas at river margins, braids, and side channels (Nislow and Armstrong 2012). These habitats are very sensitive to flow changes. Rapid changes in discharge and river stage can lead to stranding of fish as stage drops and lateral/downstream displacement as stage rises (Irvine et al. 2008, Young et al. 2011, Nagrodski et al. 2012, Gibeau et al. 2016, Hayes et al. 2019). High flows can result in microhabitat velocities that exceed the limited swimming capacity for smaller fry and can cause catastrophic displacement (Nislow and Armstrong 2012). Due to these factors a number of studies have shown that early juvenile life stages are timed to coincide with periods that provide suitable flow conditions. For example, emergence is usually timed to occur before or after seasonal flooding, and year class failures of age-0 salmonids due to mistimed floods have been observed in a number of systems (see review in Nislow and Armstrong 2012). These studies indicate that hydrological alteration during the fry stage can have negative effects on survival. These observations from other systems are consistent with flow-covariate modelling for juvenile Steelhead in the Cheakamus River. There is no uncertainty that the WUP ramping rates exceed the FOC guideline by the calculated amount, and that high ramping rates in other systems have led to considerable stranding. There is however high uncertainty about the magnitude of the population-level effect of high ramping rates and stranding events.

Our estimated flow effects on post-emergent Steelhead fry in the Cheakamus River are also informative with respect to concerns raised during the original WUP planning process that flows of $38 \text{ m}^3 \cdot \text{s}^{-1}$ in August, intended to extend the commercial rafting season and improve boating conditions, could have a negative effect on survival rates (BC Hydro 2005). The stock-recruitment flow covariate modelling showed that there was more than a 90% probability that egg-fry survival rates increase with the proportion of time that flows were less $40 \text{ m}^3 \cdot \text{s}^{-1}$ in July and August. In addition, steady rafting flows of $\sim 38 \text{ m}^3 \cdot \text{s}^{-1}$ may promote the use of braids and side channels located between gravel bars and the bank by small fish. The August 2018 ramping study demonstrated that these areas are heavily used by Steelhead and coho fry and are dewatered as flows are reduced from $38 \text{ m}^3 \cdot \text{s}^{-1}$ to $20 \text{ m}^3 \cdot \text{s}^{-1}$. The study demonstrated that

the processes of suddenly shifting from one minimum flow to a lower minimum, as occurs when switching from the rafting flow to the minimum flow which follows, can result in fry mortality.

Results from this study indicate that Steelhead production from the Cheakamus River can potentially be increased by reducing stranding impacts via reduction in ramping rates during summer and winter. Although there is near certainty that rapid changes in flow under the WUP regime can strand thousands of Steelhead fry, the extent of this impact on the population as a whole remains to be determined. Flow covariate models presented here can be used to predict population-level impacts of alternate flow regimes during the WUP Order Review. However, the predicted benefits to Steelhead production from these models should be considered uncertain owing to limitations in the range of flow conditions that occurred during the 2008-2020 WUP-funded juvenile monitoring period. The data are not sufficient for these models to accurately predict effects associated with different combinations of pre-rampdown habituation flows, ramping rates, and the final minimum flow after the rampdown. Any change to the WUP flow regime should therefore be considered an experiment that needs to be evaluated through continued monitoring to determine if it is producing the intended benefits.

8.0 References

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Table 2.1. Minimum flows (a) and ramping rates (b) specified in the Cheakamus River Water Use Plan flow order.

a) Minimum Flows

Period	Daisy Lake Dam Discharge (m³·s⁻¹)
November 1 -December 31	3
January 1 - March 31	5
April 1 - October 31	7

	Brackendale Gauge Discharge (m³·s⁻¹)
November 1 - March 31	15
April 1 - June 30	20
July 1 - August 15	38
August 16 - August 31	20 ¹
September 1 - October 31	20

b) Ramping rates

Discharge from Daisy Lake Dam (m³·s⁻¹)	Maximum Rate of Increase (m³·s⁻¹·hr⁻¹)²	Maximum Rate of Decrease (m³·s⁻¹·hr⁻¹)
<10	52	1
10-62	52	13
>62	78	78

¹Unless directed by the Comptroller to maintain flows at 38 m³·s⁻¹.

²Upramp rates in the WUP order are specified in m³·s⁻¹·15 min⁻¹ but are presented on an hourly timestep in this table so that upramp and downramp rates are directly comparable.

Table 2.2. Example stage change calculations showing the rate of vertical drop in water level as flow is reduced from Flow 1 to Flow 2 levels. Results are provided at two locations in the mainstem Cheakamus River using existing stage-discharge rating (WSC Brackendale gauge and the Pedestrian Bridge curve provided by KWL 2014). Both sets of calculations assume a $13 \text{ m}^3 \cdot \text{s}^{-1} \cdot \text{hr}^{-1}$ downramp rate which is the most likely ramping rate at both Flow 1 levels (Table 2.1). See Fig. 1.1 for a map of locations.

	Flow at Brackendale Gauge ($\text{m}^3 \cdot \text{s}^{-1}$)	Stage (cm)	
		Brackendale Gauge	Pedestrian Bridge
Flow 1	100	162.9	191.5
Flow 2	60	134.5	160.9
Hrs for Change	3.1		
Stage change ($\text{cm} \cdot \text{hr}^{-1}$)		9.2	9.9
Flow 1	40	115.6	140.2
Flow 2	20	89.1	110.8
Hrs for Change	1.5		
Stage change ($\text{cm} \cdot \text{hr}^{-1}$)		17.2	19.1

Table 3.1. WinBugs source code for hierarchical Bayesian model predicting spawn timing of Cheakamus River Steelhead given observed female departure dates from radio telemetry.

```

#2 hyper parameters defining the mean of departure timing
mu_lgmuDep~dnorm(0,1.0E-03)           #mean departure date across years
tau_lgmuDep~dgamma(0.01,0.01)        #precision of departure date across years
sd_lgmuDep<-sqrt(1/tau_lgmuDep)      #convert to standard deviation for output only

for (iyr in 1:Nyrs){ #loop across years

  #Draw mean of normal distribution of departure timing for this year
  lgmuDep[iyr]~dnorm(mu_lgmuDep,tau_lgmuDep) #in log space
  muDep[iyr]<-exp(lgmuDep[iyr])

  #Standard deviation of normal distribution for departure timing for this year. Note annual estimates of SD are independent and
  #not drawn from hyper-distribution
  tauDep[iyr]~dgamma(5,5) #semi-informative prior owing to sparse data
  varDep[iyr]<-1/tauDep[iyr]
  sdDep[iyr]<-sqrt(varDep[iyr])

  #Loop across all observed departure dates for current year and predict proportion leaving from a normal distribution with mean
  #and variance defined from parameters above
  for(i in 1:Nrecs[iyr]){
    p1[iyr,i]<-(1/(2*3.14*varDep[iyr]))*exp(-1*pow(ObsDay[iyr,i]-muDep[iyr],2)/(2*varDep[iyr]))
  }

  sump1[iyr]<-sum(p1[iyr,1:Nrecs[iyr]]) #standardize so values sum to 1
  for(i in 1:Nrecs[iyr]){
    p2[iyr,i]<-p1[iyr,i]/sump1[iyr]
  }
}

```

Table 3.1. Con't.

```
#multinomial likelihood predicting the proportion of fish leaving by day relative to observations FemDep
FemDep[iyr,1:Nrecs[iyr]]~dmulti(p2[iyr,1:Nrecs[iyr]],TotFem[iyr])
lgSDep[iyr]<-log(sdDep[iyr])      #for computation of distribution of sd_Dep across years (in log space)
}

#Calculate the mean and sd of for variance in departure timing among yearsto create a hyper distribution for later plotting.
mu_lgSDep<-sum(lgSDep[])/Nyrs #mean of lg SDs
for(iyr in 1:Nyrs){
  SSQ[iyr]<-pow(lgSDep[iyr]-mu_lgSDep,2)
}
sd_lgSDep<-pow(sum(SSQ[])/(Nyrs-1),0.5)
```

Table 3.2. Source code for WinBUGS model estimating stock-recruitment parameters for Steelhead juvenile abundance Ricker flow covariate models. Bold values denote data or

```

#prior on productivity
alpha~dnorm(0,1.0E-03)

#prior on capacity
beta~dnorm(0,1.0E0-2)

#prior on precision for error around stock-recruitment curve (process error)
tau~dgamma(0.01,0.01)
SDproc<-pow(tau,-0.5) #convert from precision to SD

#prior on covariate effect
gamma~dnorm(0,0.01)

#loop across years
for(i in 1:Nrecs){

    #predict stock size based on estimated mean and estimated precision for each year
    #e.g., stock size=fall fry abundance for fall fry–spring age 0+ stock-recruit model
    S[i]~dnorm(Sobs[i], tauS[i])

    #cut() function stops likelihood from influencing S in Ricker model
    Scut[i]<-cut(S[i])

    #predicted log recruits/spawner (e.g., log(spring age 0+/fall fry))
    Pred_LRS[i]<-alpha + beta*Scut[i] + gamma*X[i]

    #total SD computation (sum of known observation variance for log(R/S) and
    #estimated process variance)
    SDtot[i]<- pow(pow(SDobs[i],2)+pow(SDproc,2),0.5)

    #convert from SD to precision
    tauTot[i]<-pow(SDtot[i],-2)

    #normal likelihood on log predicted and observed
    LRS[i]~dnorm(Pred_LRS[i],tauTot[i])

}

```

Table 4.1. Predicted Steelhead emergence timing in the Cheakamus River in years when radio telemetry and summer water temperature data are available. Statistics show the median and 95% credible interval by year. Also shown is the average across years based on average water temperatures since 2008 and the average spawn-timing across all years when telemetry was conducted, as determined by the hyper-distributions of spawn-timing parameters

Year	2.5%	50.0%	97.5%
2009	Jul-04	Jul-15	Jul-27
2010	Jul-09	Jul-27	Aug-13
2011	Jul-17	Aug-05	Aug-26
2016	Jun-24	Jul-05	Jul-20
2017	Jul-05	Jul-20	Aug-06
Average	Jul-05	Jul-20	Aug-17

Table 4.2. Mean size of age 2 and 3 year smolts as determined by scale ageing and length frequency data collected at the Rotary Screw Trap on the Chekamus River. Also shown is the % of 2 yr. smolts (data from Melville and Mcubbing 2012). Due to truncated trapping season beginning in 2018, estimates of Steelhead length-at-age from the RST are only available through spring of 2017.

Year	Mean Fork Length (mm)		
	Age 2 Yr	Age 3 Yr	% Age 2 Yr
2008	160	183	55%
2009	165	189	76%
2010	159	184	53%
2011	162	186	57%
2012	164	179	35%
2013	159	179	55%
2014			
2015	154	168	59%
2016	155	189	86%
2017	156	170	31%
Average	159	181	56%

Table 4.3. Size and growth rates of juvenile steelhead by year and life stage in the Cheakamus River. 0+ winter growth represents growth between the fall of calendar year $t-1$ to spring of calendar year t . Summer growth is computed in the same calendar year (from spring to fall sampling sessions).

Year	Mean Fork Length (mm)					
	0+ fall	0+ spring	1+ fall	1+ spring	2+fall	2+ spring
2008	47	62	106	119	147	159
2009	53	55	106	112	148	156
2010	49	68	107	96	147	146
2011	38	68	106	113	144	154
2012	42	61	92	109	129	153
2013	47	67	89	110	146	152
2014	46	70	101	104	140	152
2015	51	67	86	106	145	152
2016	47	71	96	104	133	151
2017	46	68	103	108	138	152
2018	45	65	102	110	135	153
2019	50		115		156	
2020	45	73	113	112	149	152
Average	47	66	102	108	143	153

Year	Growth (mm)				
	0+ winter	1+ summer	1+ winter	2+ summer	2+ winter
2008	10	44	11	29	10
2009	7	51	6	36	8
2010	15	39	-10	51	-3
2011	19	38	6	31	7
2012	22	32	2	20	9
2013	25	23	18	36	23
2014	23	31	14	37	5
2015	21	20	5	39	12
2016	19	25	18	29	5
2017	20	35	12	30	19
2018	19	38	7	25	15
2020	23	40	-3	36	-4
Average	19	34	7	34	9

Table 4.4. Calculations used to estimate annual egg deposition for Steelhead in the Cheakamus River in years when information on sex ratio and size is available from angling surveys. Egg deposition is computed as the product of escapement, the proportion females, and fecundity. The latter estimates are computed based on mean female fork length and a fecundity-fork length relationship from the Keogh River (Ward and Slaney 1993). Also shown are estimates of egg deposition for resident rainbow trout (based on average fork length of females and proportion females across all years when samples were available), and estimates of the proportion of eggs contributed by resident rainbow trout relative to the total egg deposition from Steelhead and resident trout.

Year	Steelhead							Resident Trout		
	Fork length & Sex Sample Size	Average Female Fork Length (mm)	Average Fecundity	% Females	Total Escapement (Wild+Hatchery)	Total Eggs (‘000s)	Egg (‘000s) - Escapement Ratio	Abundance	Total Eggs (‘000s)	% Resident Eggs
2000	18	700	3,329	50%	78	130	1.7	17	5	4%
2001	27	756	4,219	41%	322	553	1.7	22	7	1%
2003	33	801	5,016	52%	317	820	2.6	107	35	4%
2004	36	769	4,431	44%	345	679	2.0	75	24	3%
2005	38	776	4,552	50%	336	764	2.3	54	18	2%
2009	27	735	3,864	59%	217	498	2.3	18	6	1%
2010	57	691	3,206	44%	1,050	1,477	1.4	183	60	4%
2011	107	794	4,885	61%	888	2,636	3.0	67	22	1%
2012	9	836	5,733	56%	395	1,257	3.2	127	42	3%
2013	24	794	4,883	58%	943	2,687	2.8	174	57	2%
2014	80	766	4,391	51%	545	1,227	2.3	103	34	3%
2015	88	780	4,640	55%	579	1,465	2.5	124	41	3%
2016	31	748	4,068	65%	512	1,344	2.6	330	108	7%
2017	26	806	5,116	50%	711	1,820	2.6	142	46	2%
2018	16	707	3,432	38%	384	494	1.3	112	37	7%
2019	46	773	4,511	50%	422	952	2.3	56	18	2%
2020	48	761	4,289	56%	632	1,524	2.4	133	44	3%
Avg.	42	764	4,386	52%	510	1,196	2.3	108	35	3%

Table 4.5. Average abundance between 2008 and 2018 by life stage in Cheakamus and Brohm Rivers. Units are in thousands of fish. Also shown is the average survival between fry in the fall (0+ fall) and 0+ parr in the spring, and the annual survival rates between 0+ and 1+ parr between consecutive springs. Note that Cheakamus estimates are based on data through fall of 2018, while Brohm estimates are based on data through spring of 2018 (data series discontinued after this date). Cheakamus River fall abundance estimates after 2018 are not included so Brohm and Cheakamus River means are calculated over the same set of years.

Life Stage	Abundance			Survival Across Stages	
	Cheakamus	Brohm	% Brohm	Cheakamus	Brohm
0+ fall	202.6	20.7	10%		
0+ spring	48.3	3.4	8%	24%	16%
1+ spring	15.5	1.6	11%	32%	47%

Table 5.1. Steelhead escapement estimates to the Cheakamus River, 1996-2019. Mean and CV denote the mean and coefficient of annual escapement estimates. Average values of escapement from juvenile fish which reared under pre-Instream Flow Agreement (pre-IFA), IFA, and Water Use Planning (WUP) periods are shown at the bottom of the table. IFA periods are separated by returns that were not and were affected by the CN caustic soda spill (pre-spill and post-spill periods, respectively).

Year	Wild		Hatchery		Wild+Hatchery	
	Mean	CV	Mean	CV	Mean	CV
1996	173	0.17				
1997	111	0.16				
1999	162	0.17				
2000	78	0.19				
2001	322	0.13				
2002	440	0.12				
2003	317	0.09				
2004	345	0.13				
2005	336	0.10				
2006	320	0.11				
2007	541	0.09				
2008	345	0.10				
2009	115	0.19	102	0.34	217	0.19
2010	629	0.09	421	0.16	1,050	0.09
2011	605	0.10	283	0.26	888	0.11
2012	395	0.14				
2013	943	0.09				
2014	545	0.11				
2015	579	0.09				
2016	512	0.10				
2017	711	0.08				
2018	384	0.13				
2019	422	0.13				
2020	632	0.12				
Pre-IFA ('96-'01)	169	0.17				
IFA Pre-Spill ('02-'07)	383	0.11				
IFA Post-Spill ('08-'09)	230	0.15				
WUP ('10-'20)	578	0.11				

Table 5.2. Estimates of the effect of flow regime-related changes in freshwater production of Steelhead in the Cheakamus River based on adjusted escapement ratios. Escapement ratios are first adjusted based on the ratio of smolt-adult survival rates (SAR) between periods. Separate SARs are provided for the Puget Sound – Keogh River aggregate (Kendall et al. 2017), and from the Keogh River only (Middleton 2017). A further adjustment to the WUP/IFA ratio is required to account for an in freshwater survival rates due to higher pink salmon returns, which occurred during the WUP period only (see Table 6.2).

	SAR Index		
	Escapement	Puget-Keogh	Keogh Only
Avg. pre-IFA ('98-'01)	169	1.6%	5.5%
Avg. IFA (pre-spill '02-'07)	383	1.2%	4.1%
Avg. WUP ('10-'19)	578	1.6%	5.3%
		Adjusted	
IFA/pre-IFA escapement ratio	2.26	2.97	3.01
WUP/IFA escapement ratio	1.49	1.20	1.19
WUP/IFA Pink Salmon adjustment	1.49		
Marine survival- and Pink salmon-adjusted WUP/IFA ratio		0.81	0.80
Reduction in freshwater production under WUP relative to IFA		19%	20%

Table 5.3. Estimates of abundance of Pink Salmon fry (in millions) from Rotary Screw Traps on the Coquitlam and Cheakamus Rivers. Also shown are the averages for the IFA (≤ 2006) and WUP (≥ 2008) periods and the ratio of mean abundances.

Year	Coquitlam	Cheakamus
2002		1.67
2004	0.52	0.08
2006	0.21	0.30
2008	0.29	2.06
2010	0.55	6.16
2012	3.56	29.31
2014	6.03	25.39
2016	1.31	5.49
2018	0.11	3.92
IFA	0.37	0.69
WUP	1.97	12.06
WUP/IFA	5.41	17.58

Table 6.1. Juvenile Steelhead abundance and survival for Cheakamus (a) and Brohm (b) River. Abundance for each age class and sampling period is the median of the posterior distribution of the total abundance estimates from the HBM. Survival between periods is the ratio of abundances across adjacent rows. Survival rates are not calculated in cases where abundance estimates needed for the calculation are unreliable. 0+-1+ survival rates in years effected by pink salmon are highlighted in pink.

Table 6.1. Con't.

a) Cheakamus

River	Brood Year	Age (Yr. from Emergence)	Sampling Period	Abundance ('000s)	Survival between Periods	Survival Fall Age-0 Spring Age-1
Cheakamus	2008	Eggs	Spring-08	789		
		0+	Fall-08	236.5	30%	
		0+	Spring-09	48.6	21%	
		1+	Spring-10	18.3	38%	8%
	2009	Eggs	Spring-09	498		
		0+	Fall-09	97.7	20%	
		0+	Spring-10	22.0	22%	
		1+	Spring-11	3.5	16%	4%
	2010	Eggs	Spring-10	1,050		
		0+	Fall-10	70.0	7%	
		0+	Spring-11	31.9	46%	
		1+	Spring-12	19.6	61%	28%
	2011	Eggs	Spring-11	2,636		
		0+	Fall-11	389.4	15%	
		0+	Spring-12	87.3	22%	
		1+	Spring-13	11.56	13%	3%
	2012	Eggs	Spring-12	1,257		
		0+	Fall-12	150.3	12%	
		0+	Spring-13	48.9	33%	
		1+	Spring-14	45.6	93%	30%
2013	Eggs	Spring-13	2,687			
	0+	Fall-13	246.7	9%		
	0+	Spring-14	52.5	21%		
	1+	Spring-15	7.0	13%	3%	
2014	Eggs	Spring-14	1,227			
	0+	Fall-14	151.1	12%		
	0+	Spring-15	22.9	15%		
	1+	Spring-16	14.20	62%	9%	
2015	Eggs	Spring-15	1,465			
	0+	Fall-15	141.4	10%		
	0+	Spring-16	32.9	23%		
	1+	Spring-17	10.5	32%	7%	
2016	Eggs	Spring-16	1,344			
	0+	Fall-16	237.2	18%		
	0+	Spring-17	56.7	24%		
	1+	Spring-18	9.6	17%	4%	
2017	Eggs	Spring-17	1,820			
	0+	Fall-17	332.8	18%		
	0+	Spring-18	79.4	24%		
	1+	Spring-19	No Data	No Data		
2018	Eggs	Spring-18	494			
	0+	Fall-18	175.2	35%		
	0+	Spring-19	No Data	No Data		
	1+	Spring-20	11.3	No Data	6%	
2019	Eggs	Spring-19	952			
	0+	Fall-19	226.1	24%		
	0+	Spring-20	50.1	22%		
2020	Eggs	Spring-20	1,524			
	0+	Fall-20	188.5	12%		

Table 6.1. Con't.

b) Brohm

River	Brood Year	Age (Yr. from Emergence)	Sampling Period	Abundance ('000s)	Survival between Periods	Survival Spring Age-0 Spring Age-1	Survival Fall Age-0 Spring Age-1
Brohm	2008	0+	Fall-08	19.2			
		0+	Spring-09	NA			
		1+	Fall-09	4.5	NA		
		1+	Spring-10	2.7	59%	NA	14%
	2009	0+	Fall-09	20.3			
		0+	Spring-10	4.1	20%		
		1+	Fall-10	3.4	82%		
		1+	Spring-11	1.1	32%	26%	5%
	2010	0+	Fall-10	18.67			
		0+	Spring-11	3.83	21%		
		1+	Fall-11	3.23	84%		
		1+	Spring-12	2.22	69%	58%	12%
	2011	0+	Fall-11	21.87			
		0+	Spring-12	4.32	20%		
		1+	Fall-12	4.04	94%		
		1+	Spring-13	1.51	37%	35%	7%
	2012	0+	Fall-12	30.69			
		0+	Spring-13	3.59	12%		
		1+	Fall-13	5.1	142%		
		1+	Spring-14	2.3	45%	63%	7%
	2013	0+	Fall-13	15.5			
		0+	Spring-14	3.8	25%		
		1+	Fall-14	5.9	154%		
		1+	Spring-15	0.8	14%	22%	5%
	2014	0+	Fall-14	14.8			
		0+	Spring-15	1.9	13%		
		1+	Fall-15	3.10	161%		
		1+	Spring-16	0.89	29%	46%	6%
2015	0+	Fall-15	24.27				
	0+	Spring-16	3.61	15%			
	1+	Fall-16	4.33	120%			
	1+	Spring-17	1.1	26%	32%	5%	
2016	0+	Fall-16	21.0				
	0+	Spring-17	1.8	9%			
	1+	Spring-18	NA	NA	NA	NA	

Table 6.2. Comparison of models predicting the log of the annual survival rate for parr (spring 1+ abundance / spring 0+ abundance) as a function of 0+ parr abundance (denotes as ‘S’ below) and pink salmon abundance as indexed by annual fry estimates from the Rotary Screw Trap (a).

a) Models predicting annual parr survival rate

Model (log(survival)=...)	r²	AIC	b1	b2	b3
b0 + b1*S	0.2	23.2	-0.016		
b0 + b1*S + b2*Pink	0.7	16.3	-0.011	0.047	
b0 + b1*S + b2*Pink + b3*S*Pink	0.74	17.1	-0.014	-0.001	0.0012
b0 + b1*S + b2*S*Pink	0.74	15.1	-0.014	0.001	

Table 6.2. Con't.

b) Pink fry abundance estimates at the Rotary Screw Trap on the Cheakamus River and predicted effect on annual survival rates of Steelhead Parr. The latter estimates are computed as $\exp(b_2 * \text{PinkFry})$, where b_2 is the coefficient for model $b_0 + b_1 * S + b_2 * \text{pink}$ in shown in a). The ratio of the average survival effects over WUP and IFA periods is used to adjust the change in escapement between IFA and WUP periods to account for higher pink abundance during the later period (see Table 5.2).

Year	Pink Fry (millions)	Pink Effect on Steelhead Survival	Pink Effect for IFA and WUP Periods
2002	1.67	1.08	
2003	0	1.00	
2004	0.08	1.00	1.02
2005	0	1.00	
2006	0.30	1.01	
2007	0	1.00	
2008	2.06	1.10	
2009	0	1.00	
2010	6.16	1.34	
2011	0	1.00	
2012	29.31	3.98	1.52
2013	0	1.00	
2014	25.39	3.31	
2015	0	1.00	
2016	5.49	1.30	
2017	0	1.00	
2018	3.92	1.20	
WUP/IFA pink salmon adjustment Ratio			1.49

Table 6.3. Comparison of alternative Ricker flow covariate models predicting Steelhead juvenile abundance as a function of abundance from the previous life stage. Results are shown for egg-fall fry (a), fall fry – spring age 0+ parr (b), and spring age 0+ parr – spring age1+ parr (c). Flow covariates for a) were based on discharges in July and August. Flow covariates in b) were based on conditions between October and March. Flow covariates in c) were based on conditions between April to September (summer) or between October and March (winter). Mean, LCL, UCL denote the mean value of γ (flow covariate effect) and the lower and upper 95% credible intervals, respectively. Prob>0 is the probability that γ is greater than zero. r^2 is the proportion of observed variance in log fry abundance predicted by the model, and Δ DIC is the difference in the deviance information criteria for each model relative to the model with the lowest value (the best model). ModWgt is the model weight determined based on the ratio of each models $\exp(-0.5*\Delta$ DIC) value to the sum of these values across models. Rows highlighted in grey have Δ DIC values ≤ 2 and are the most reliable models.

a) Egg- Fall Fry (spring 2008 – fall 2020)

Covariate	γ (flow covariate effect)				r^2	Δ DIC	ModWgt
	Mean	LCL	UCL	Prob>0			
No flow or density effects	NA	NA	NA	NA	0.00	8.6	
No flow but density effect	NA	NA	NA	NA	0.27	7.5	
Upramp_1Day_(cms/day)	-0.267	-0.550	0.028	3.5	0.48	5.5	0.049
Downramp_1Day_(cms/day)	-0.339	-0.556	-0.117	0.4	0.65	0.0	0.769
Avg_Q_(cms)	-0.232	-0.542	0.080	6.7	0.42	6.7	0.027
SD_Q_(cms)	-0.103	-0.448	0.262	26	0.30	9.3	0.008
Max_Q_(cms)	-0.247	-0.552	0.068	5.4	0.44	6.3	0.034
Prop_Hrs_<25_cms	0.140	-0.193	0.476	81.7	0.33	8.8	0.009
Prop_Hrs_<40_cms	0.287	-0.003	0.584	97.4	0.52	4.7	0.072
Prop_Hrs_<60_cms	0.195	-0.130	0.510	90	0.38	7.7	0.016
Prop_Hrs_<80_cms	0.097	-0.272	0.453	72.1	0.29	9.4	0.007
Prop_Hrs_<100_cms	0.149	-0.205	0.502	81.8	0.32	8.9	0.009

Table 6.4. Con't.

b) Fall Fry – Spring Age 0+ Parr (fall 2008 – spring 2020)

Covariate	γ (flow covariate effect)				r^2	Δ DIC	ModWgt
	Mean	LCL	UCL	Prob>0			
No flow or density effects	NA	NA	NA	NA	0.00	4.2	
No flow but density effect	NA	NA	NA	NA	0.12	5.4	
Upramp_1Day (cms/day)	-0.199	-0.391	0.007	2.7	0.45	2.7	0.097
Downramp_1Day (cms/day)	-0.221	-0.394	-0.051	0.7	0.70	0.0	0.372
Avg_Q (cms)	-0.169	-0.357	0.020	3.9	0.42	3.5	0.066
SD_Q (cms)	-0.183	-0.361	0.002	2.6	0.48	2.4	0.111
Max_Q (cms)	-0.185	-0.369	0.006	2.8	0.48	2.7	0.098
Prop_Hrs <25 cms	0.082	-0.176	0.336	75.5	0.17	7.5	0.009
Prop_Hrs <40 cms	0.086	-0.147	0.297	80	0.19	6.9	0.012
Prop_Hrs <60 cms	0.157	-0.036	0.349	94.8	0.42	4.1	0.048
Prop_Hrs <80 cms	0.179	-0.011	0.358	96.9	0.50	2.9	0.088
Prop_Hrs <100 cms	0.182	-0.007	0.363	97	0.49	2.7	0.098

c) Spring Age 0+ Parr – Spring Age 1+ Parr (spring 2009 - spring 2018)

Covariate	γ (flow covariate effect)				r^2	Δ DIC	ModWgt
	Mean	LCL	UCL	Prob>0			
No flow, density, or pink effects	NA	NA	NA	NA	0.00	7.5	
No flow or density effects but pink effect	NA	NA	NA	NA	0.60	1.9	
No flow effect but density and pink effects	NA	NA	NA	NA	0.70	2.4	
Upramp (cms/day)_ Summer	0.057	-0.414	0.545	61.6	0.71	5.4	0.021
Downramp (cms/day)_ Summer	-0.220	-0.692	0.280	15.3	0.76	3.5	0.053
Avg_Q (cms)_ Summer	-0.157	-1.868	1.599	39.3	0.70	4.9	0.026
SD_Q (cms)_ Summer	0.095	-0.811	1.050	59.2	0.71	5.6	0.018
Max_Q (cms)_ Summer	0.006	-0.640	0.668	50.8	0.70	5.6	0.018
Prop_Hrs <=25 cms Summer	0.311	-0.059	0.679	96.1	0.85	0.0	0.304
Prop_Hrs <=40 cms Summer	0.175	-0.475	0.758	76.1	0.73	4.8	0.028
Prop_Hrs <=60 cms Summer	0.814	-0.563	2.661	89.5	0.80	1.8	0.123
Prop_Hrs <=80 cms Summer	-0.028	-1.155	1.067	48.3	0.70	5.5	0.019
Prop_Hrs <=100 cms Summer	-0.280	-1.491	0.873	28.3	0.74	5.2	0.022
Upramp (cms/day)_ Winter	0.356	-0.322	1.042	88.3	0.77	2.7	0.080
Downramp (cms/day)_ Winter	0.246	-0.456	0.95	79.9	0.74	4.1	0.040
Avg_Q (cms)_ Winter	0.164	-0.343	0.675	77.8	0.73	4.4	0.033
SD_Q (cms)_ Winter	0.226	-0.291	0.745	84.7	0.76	3.7	0.049
Max_Q (cms)_ Winter	0.216	-0.451	0.892	78.2	0.73	4.6	0.031
Prop_Hrs <=25 cms Winter	-0.015	-0.58	0.528	48	0.70	5.8	0.017
Prop_Hrs <=40 cms Winter	-0.073	-0.548	0.4	35.5	0.71	5.6	0.018
Prop_Hrs <=60 cms Winter	-0.122	-0.622	0.405	27.7	0.72	5.0	0.025
Prop_Hrs <=80 cms Winter	-0.185	-0.661	0.315	18	0.74	4.2	0.038
Prop_Hrs <=100 cms Winter	-0.183	-0.668	0.307	19.4	0.74	4.2	0.037

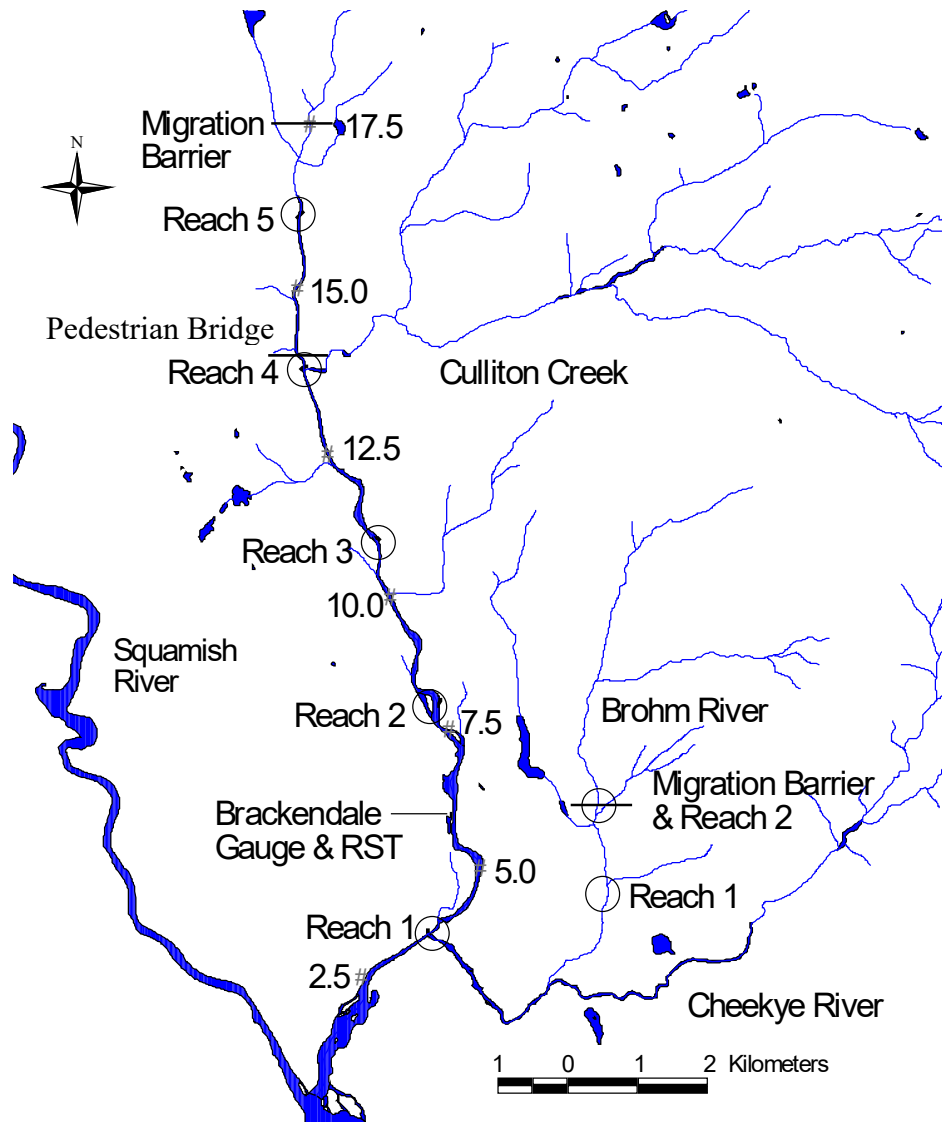


Figure 1.1. Map of the Cheakamus River study area showing the locations of the upstream limit of reach breaks used for habitat and juvenile surveys (open circles), distance (km) from the Squamish River confluence (gray points), migration barriers for anadromous fish in the Cheakamus and Brohm Rivers, and the Water Survey of Canada discharge gauge at Brackendale and the rotary screw trap (RST). Also shown is the location of the Pedestrian Bridge rating curve used in the stage change analysis.

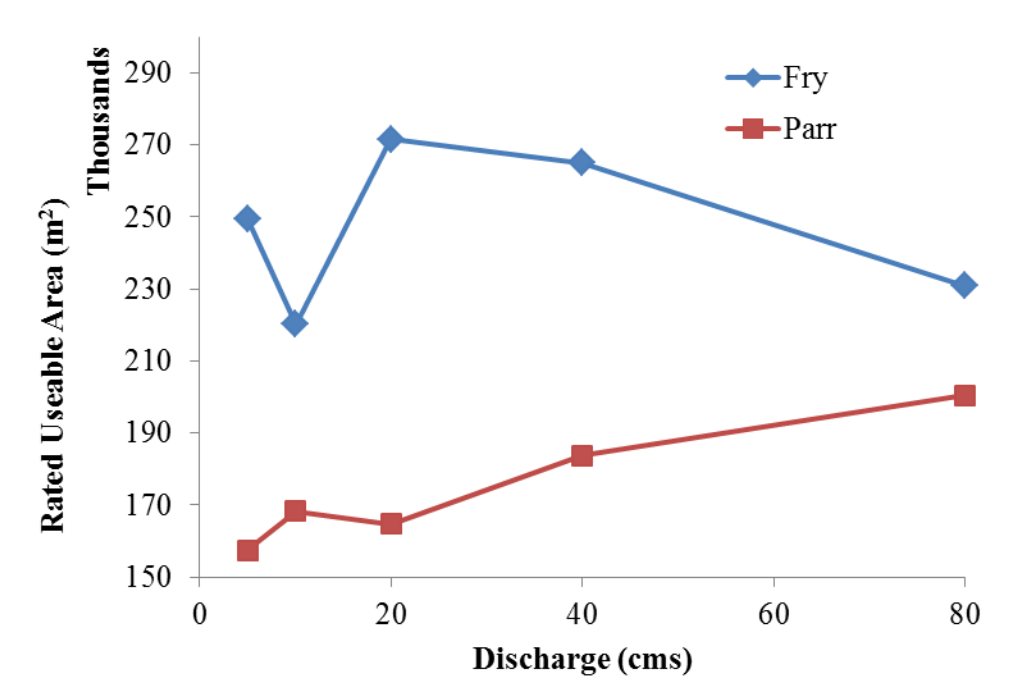


Figure 1.2. Changes in predicted useable juvenile Steelhead habitat in the Cheakamus River (summed across reaches) as a function of discharge. This habitat model was used in the initial WUP assessment (BC Hydro 2007).

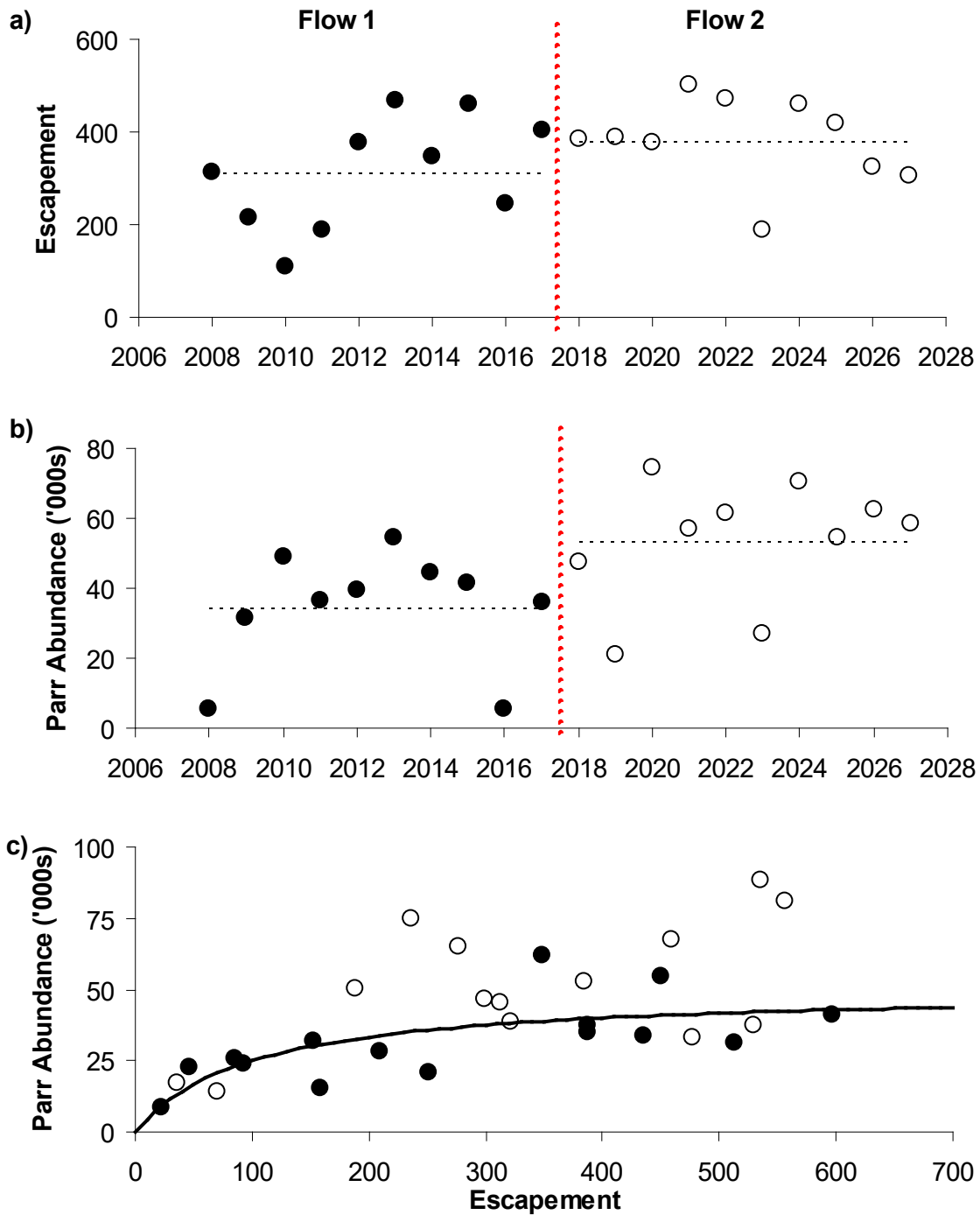


Figure 1.3. Theoretical responses of escapement (a) and parr abundance (b) under two flow regimes, with 10 years of data collected under each regime, and the stock-recruit relationship between these life stages over the two periods (c). Solid and open circles represent data collected under flow regimes 1 and 2, respectively. Dashed horizontal lines in a) and b) represent the mean abundances over these periods. The solid line in c) represents the best-fit stock-recruitment curve under flow regime 1. Evidence for the effect of flow increases from a) to c) by reducing the confounding effects of marine survival (b) and the effects of both marine survival and density dependence (c).

Reporting Year	Calendar Year	Season	Escapement	Juvenile Ages			Events
				Age-0	Age-1	Age-2	
	2005	Spring					
		Fall					
	2006	Spring					← WUP Flow Regime Begins
		Fall					
	2007	Spring			2 yr smolt		← WUP Monitoring Begins
		Fall					
2008 (1)	2008	Spring			2 yr smolt	3 yr smolt	pilot sampling
		Fall					
2009 (2)	2009	Spring			2 yr smolt	3 yr smolt	
		Fall					
2010 (3)	2010	Spring			2 yr smolt	3 yr smolt	
		Fall					
2011 (4)	2011	Spring			2 yr smolt	3 yr smolt	
		Fall					
2012 (5)	2012	Spring			2 yr smolt	3 yr smolt	← WUP Phase I Monitoring Ends
		Fall					
2013 (6)	2013	Spring			2 yr smolt	3 yr smolt	
		Fall					
2014 (7)	2014	Spring			2 yr smolt	3 yr smolt	
		Fall					
2015 (8)	2015	Spring			2 yr smolt	3 yr smolt	
		Fall					
2016 (9)	2016	Spring			2 yr smolt	3 yr smolt	
		Fall					
2017 (10)	2017	Spring			2 yr smolt	3 yr smolt	← WUP Phase II Monitoring Ends
		Fall					
2018 (11)	2018	Spring			2 yr smolt	3 yr smolt	← WUP Monitoring Extension
		Fall					
2019 (12)	2019	Spring		No juvenile	2 yr smolt	3 yr smolt	
		Fall					
2020 (13)	2020	Spring			2 yr smolt	3 yr smolt	
		Fall					

Figure 1.4. Life history table for the freshwater life stages of Steelhead in the Cheakamus River in relation to annual and seasonal monitoring periods, WUP assessments and reporting periods, and implementation of the WUP flow regime. This report covers reporting years 1-12 (with the exception of some escapement data from 2019 currently not processed, and age-0 data to be collected in the fall of 2019). Each color tracks the cohort from individual broods (year of spawning) through the freshwater residency period. Note that an age-0 fish sampled in spring (April) is just less than one year old from the date of fertilization. An age-1 parr enumerated in early spring during the surveys (e.g., March) can potentially smolt in the same calendar year in late spring (e.g., May) as an age-2 smolt or the next year as an ag-3 smolt.

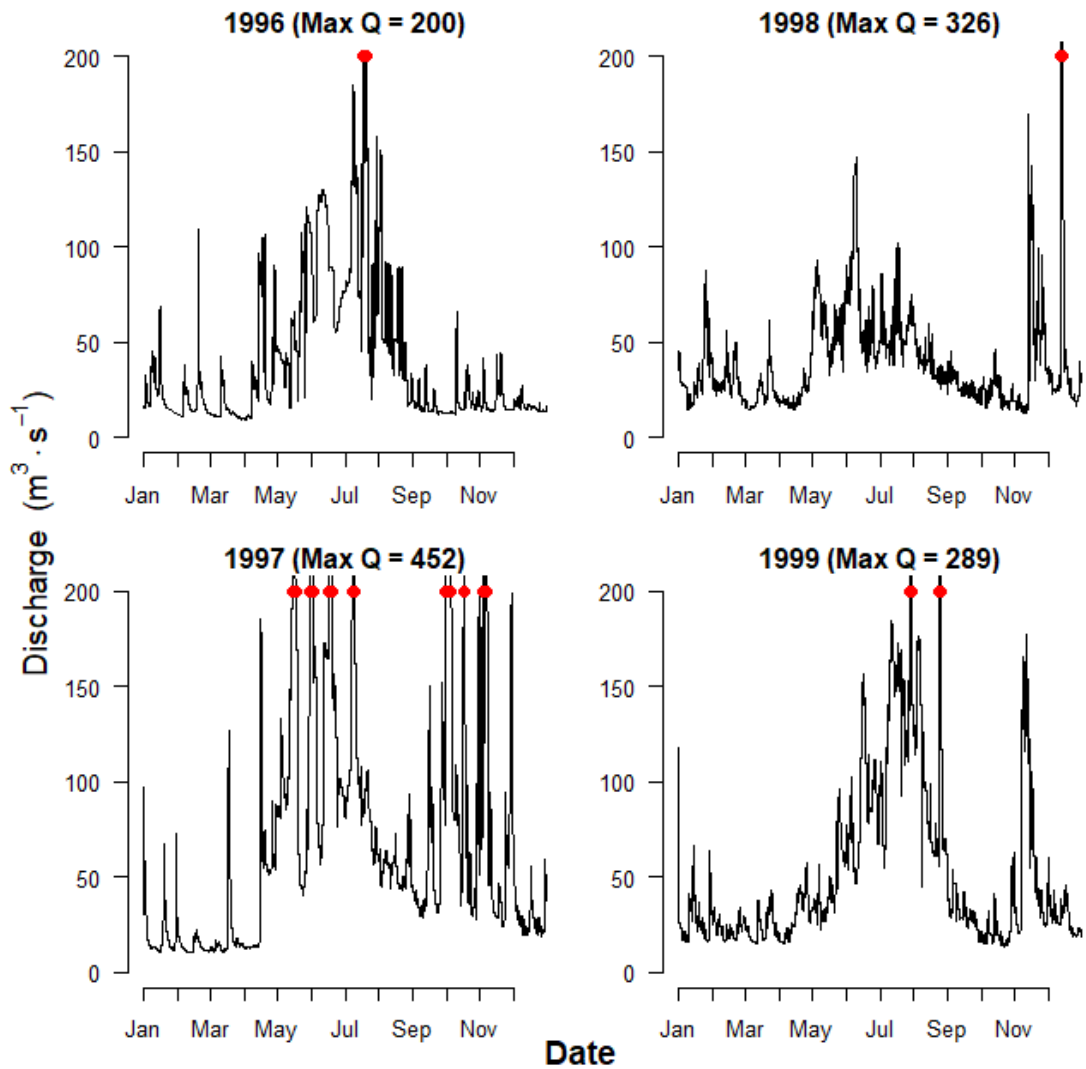


Figure 2.1. Hourly discharge at the WSC Brackendale gauge 1996-2018. Red points denote hours when discharge exceeded the y-axis maxima of $200 \text{ m}^3 \cdot \text{s}^{-1}$.

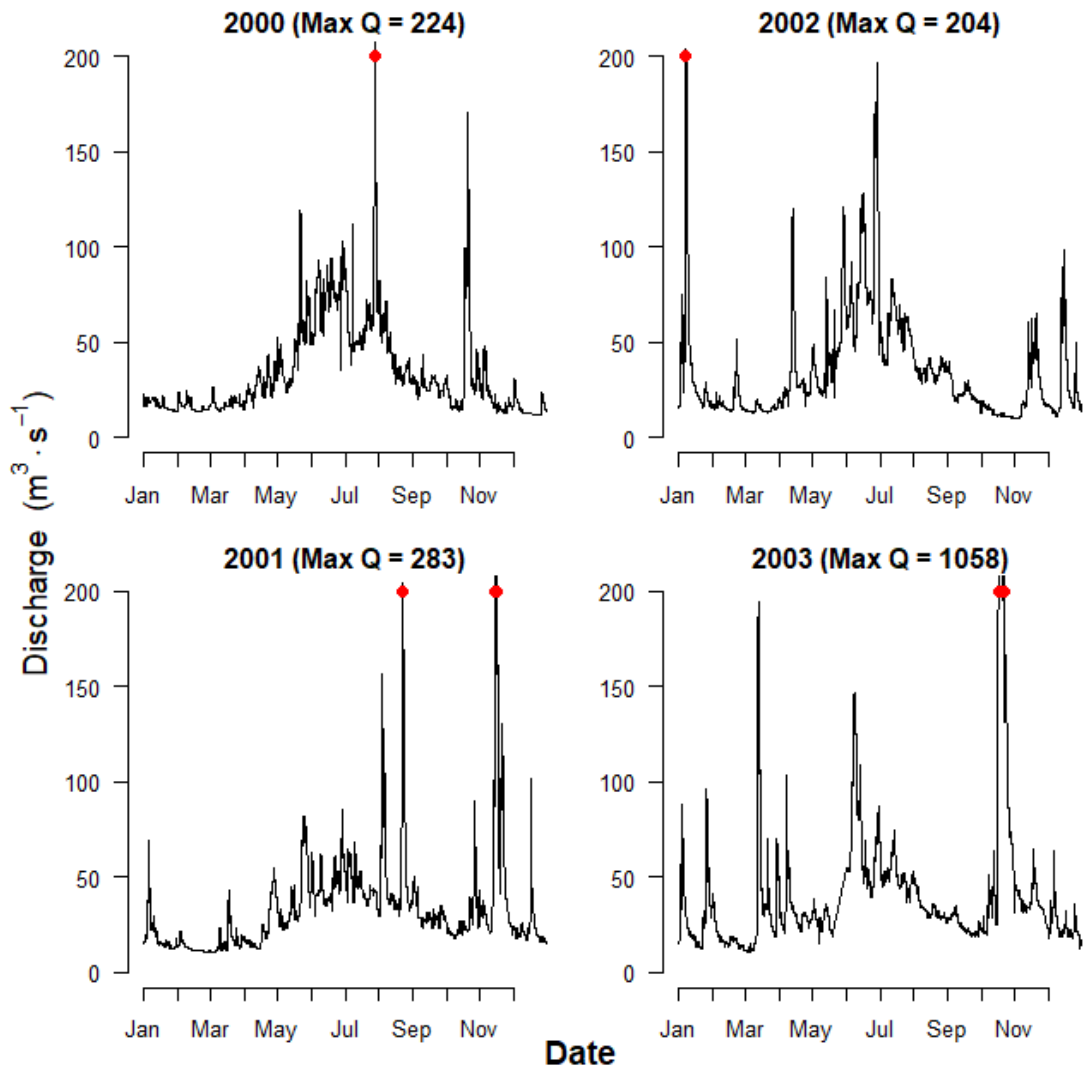


Figure 2.1. Con't.

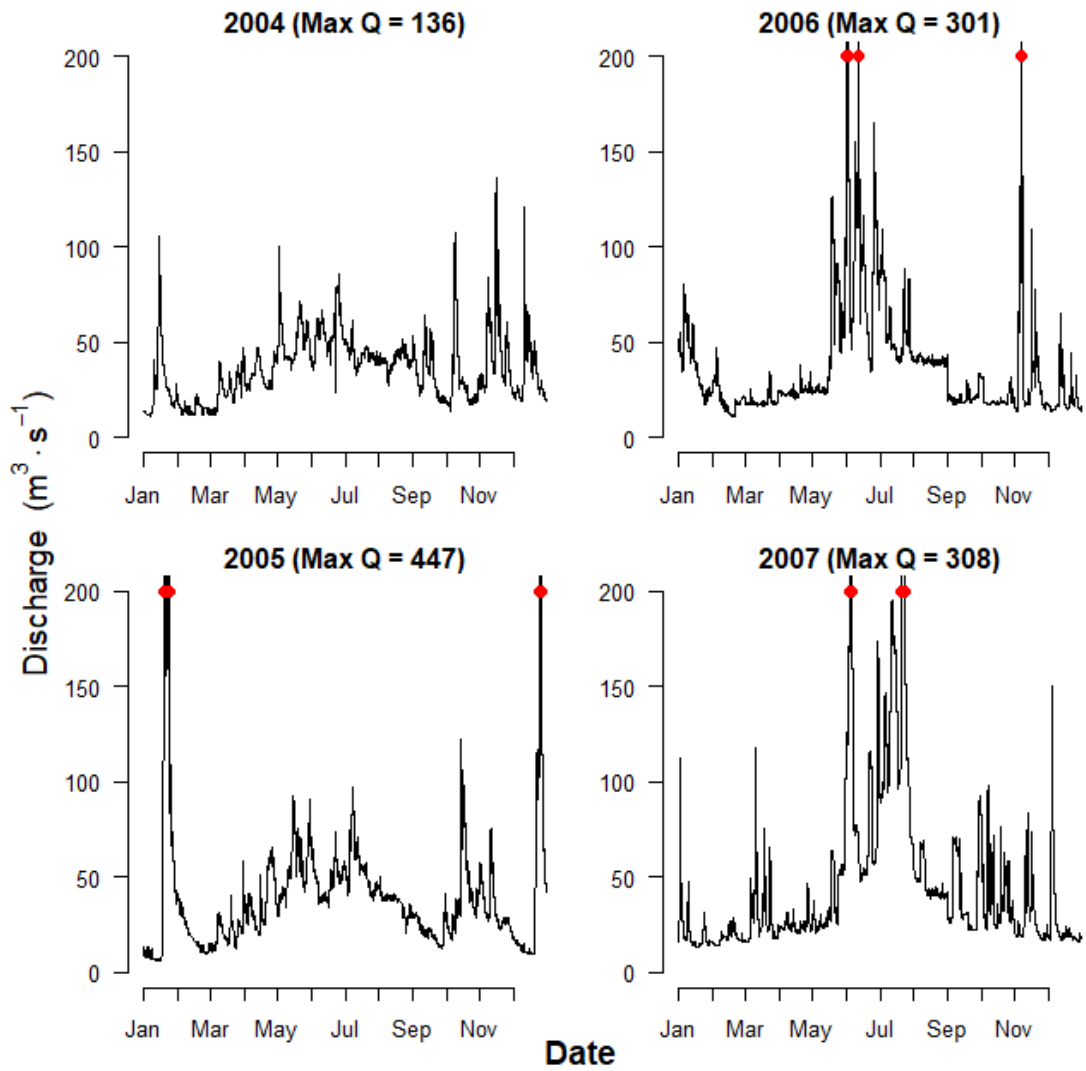


Figure 2.1. Con't.

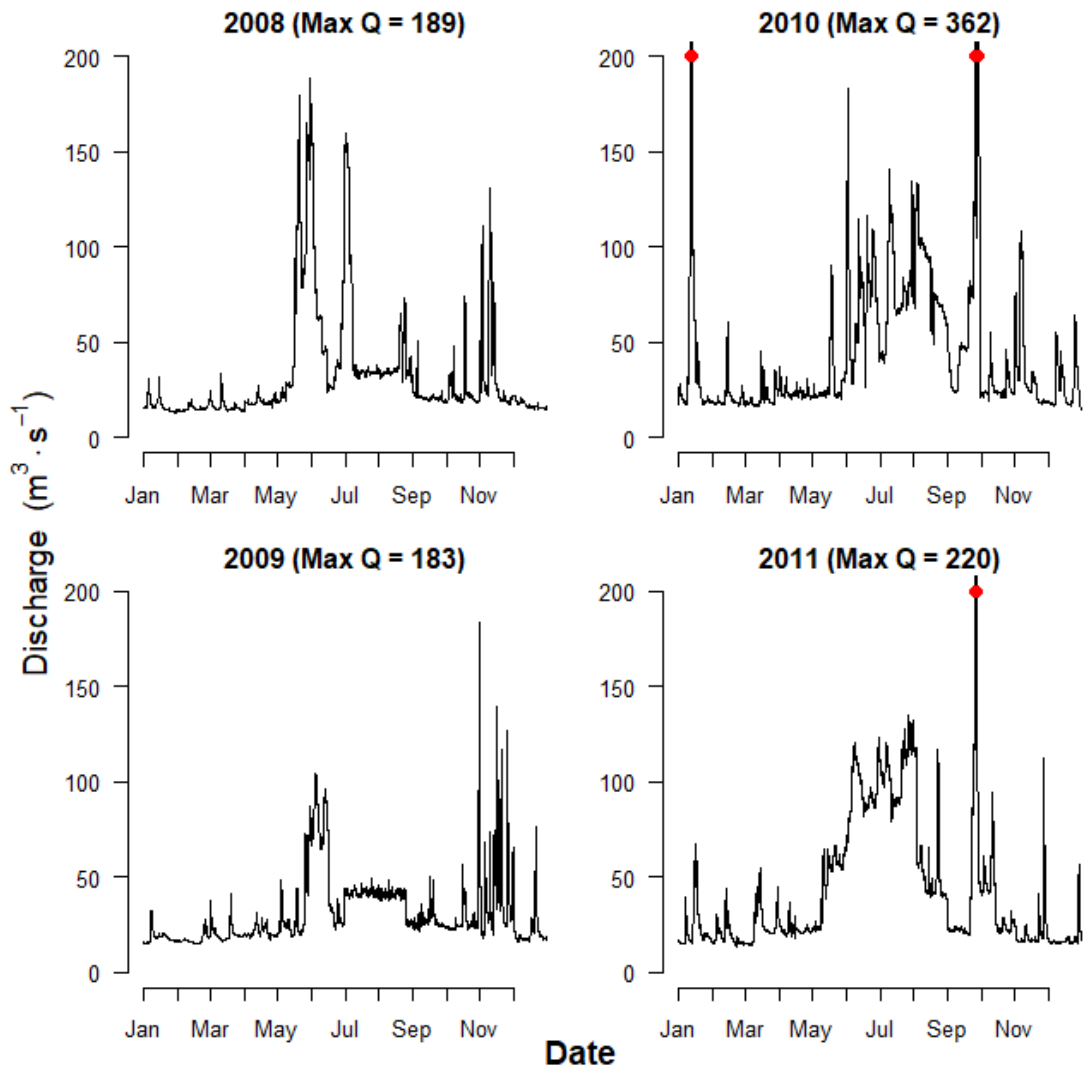


Figure 2.1. Con't.

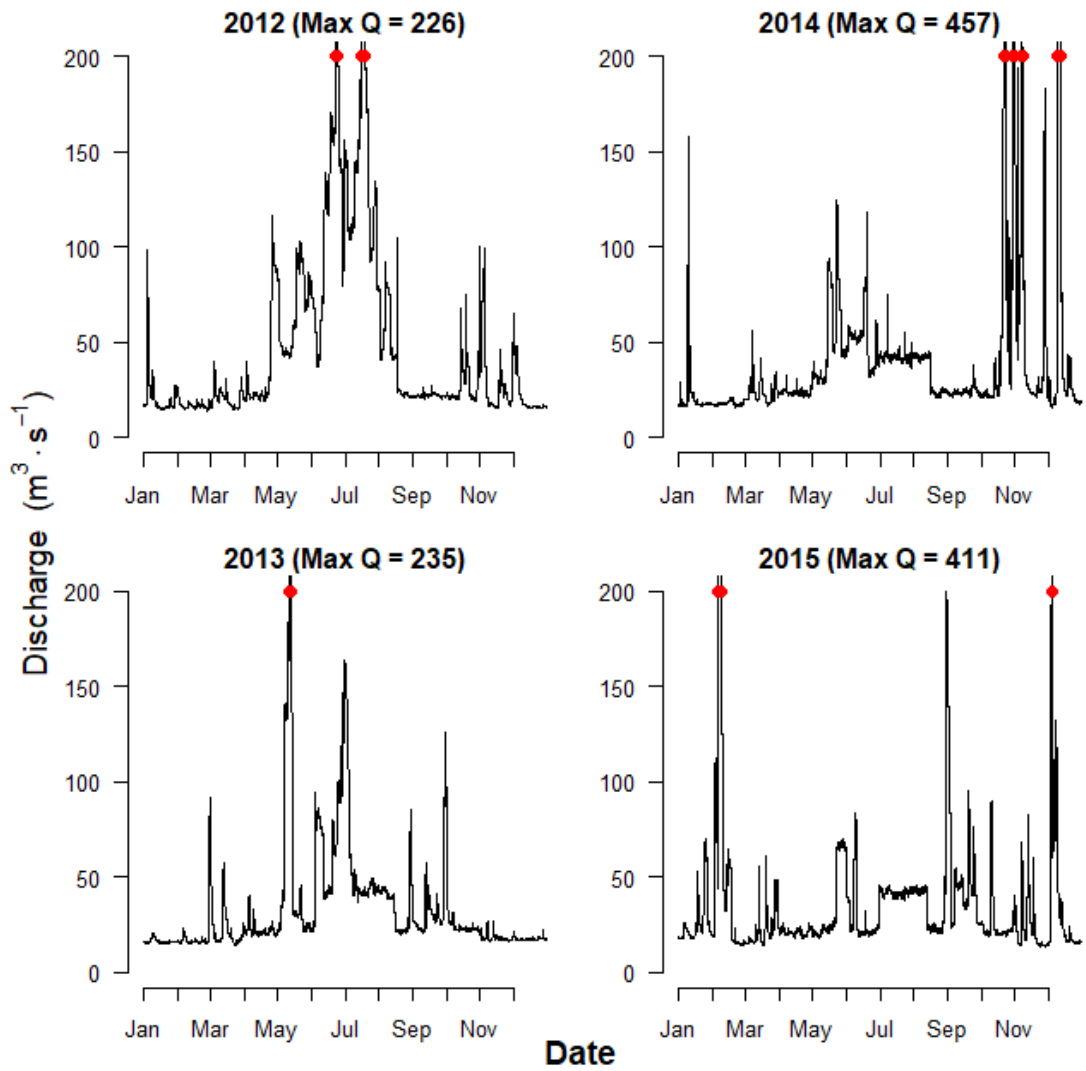


Figure 2.1. Con't.

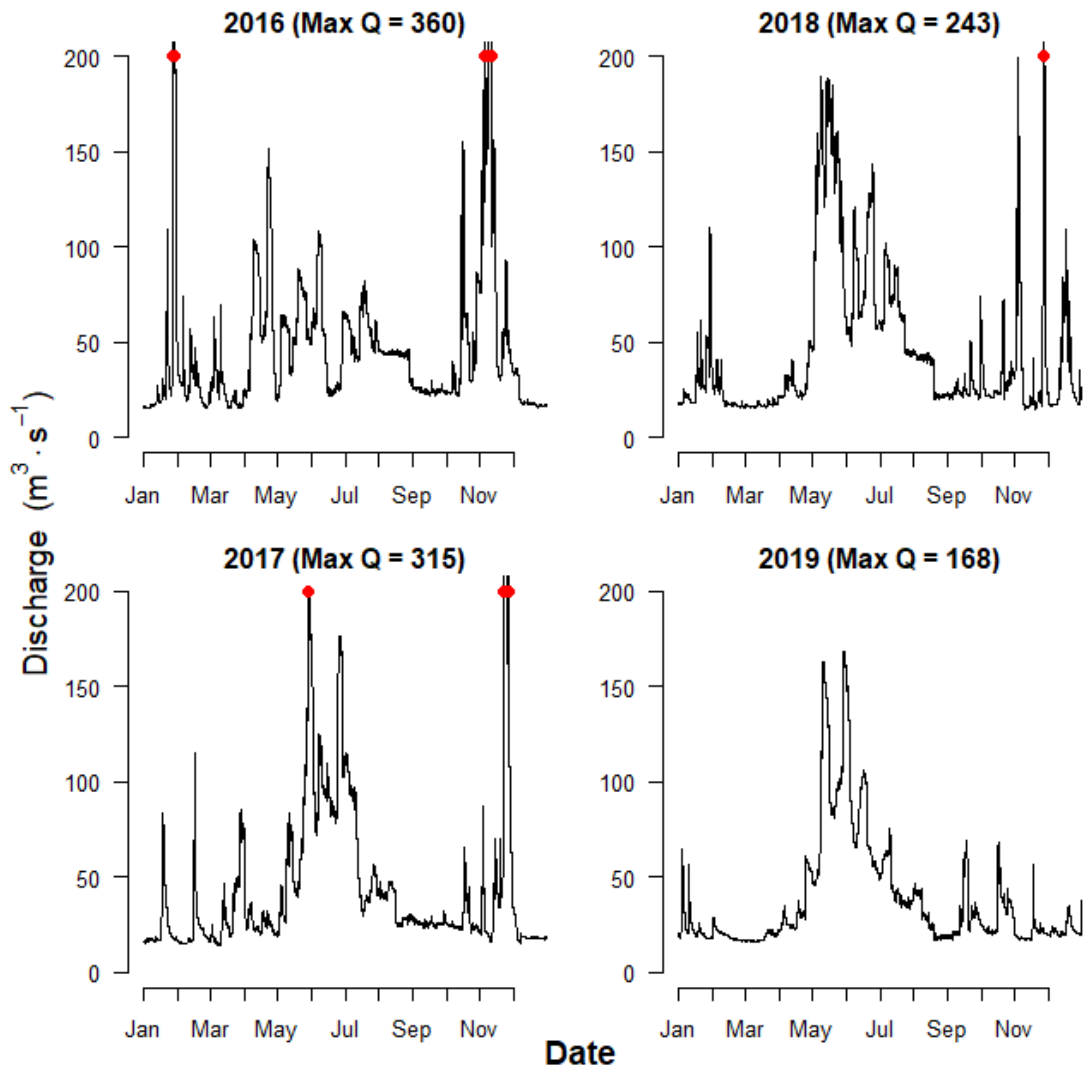
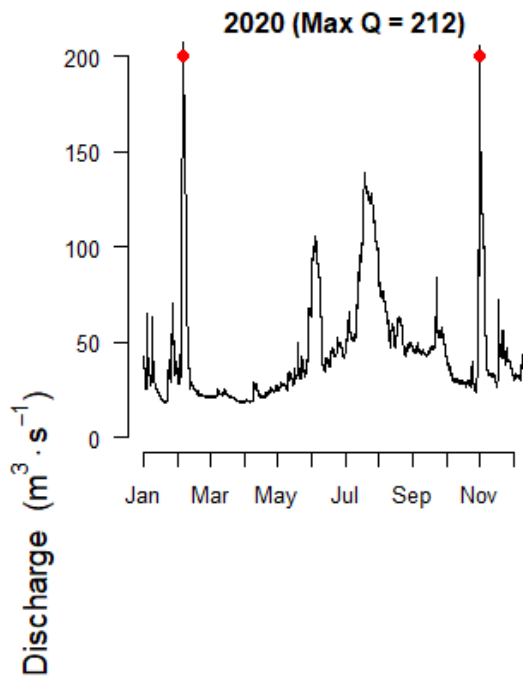


Figure 2.1. Con't.



Date

Figure 2.1. Con't.

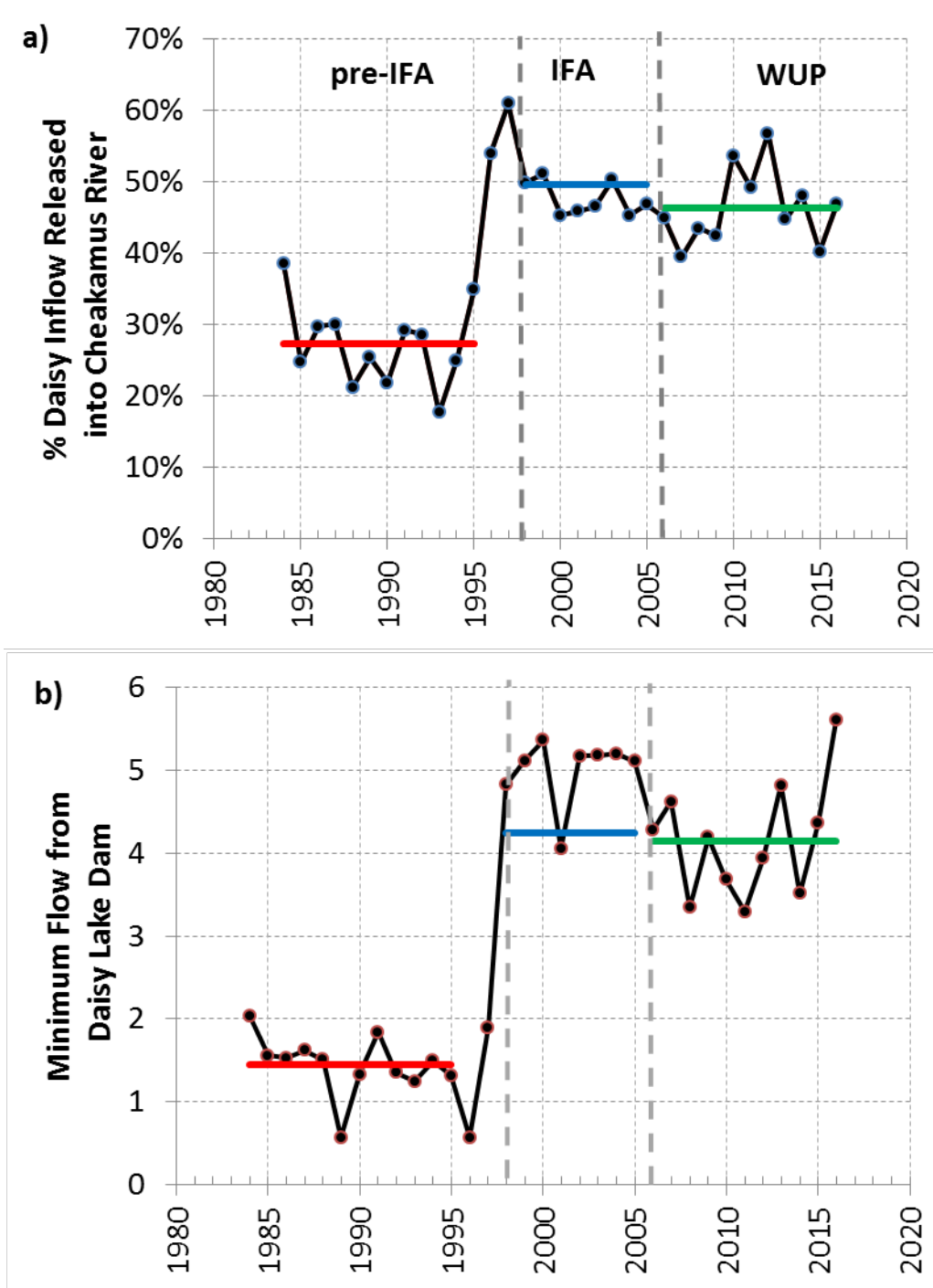


Figure 2.2. Annual percentage of inflow to Daisy Lake Reservoir released into the Cheakamus River from Daisy Lake Dam(a), and minimum flow releases from Daisy Lake Dam. Red, blue and green lines show the average levels prior to the Instream Flow Agreement (IFA), during the IFA period, and during the Water Use Planning (WUP) period through 2016, respectively.

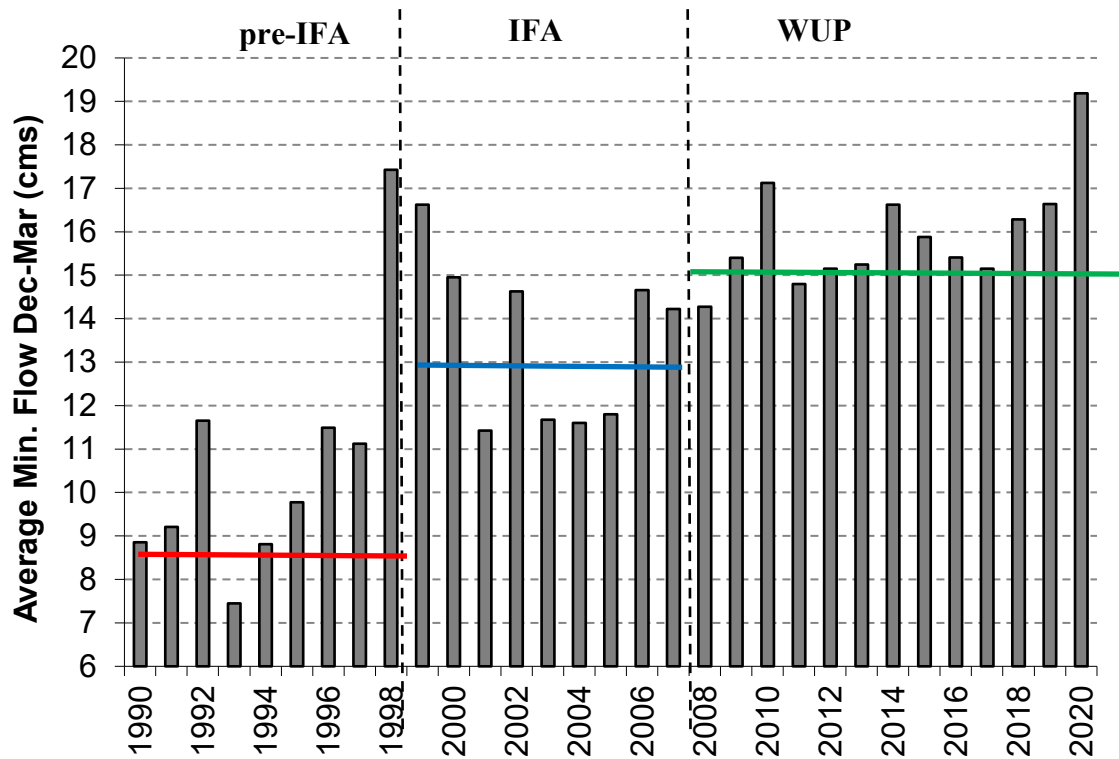


Figure 2.3. The average minimum flows during the winter at the Brackendale gauge on the Cheakamus River, 1990-2020. The average minimum flow between December and March was computed as the average of the minimum flow in December from the previous year (based on average daily flows), and the minimum flows in January, February, and March for the current year (specified on x-axis). Labels at the top of the graph identify the pre-Instream Flow Agreement (pre-IFA), IFA and current Water Use Plan (WUP) flow regime periods. Red, blue and green lines show the average minimum flow over these three periods.

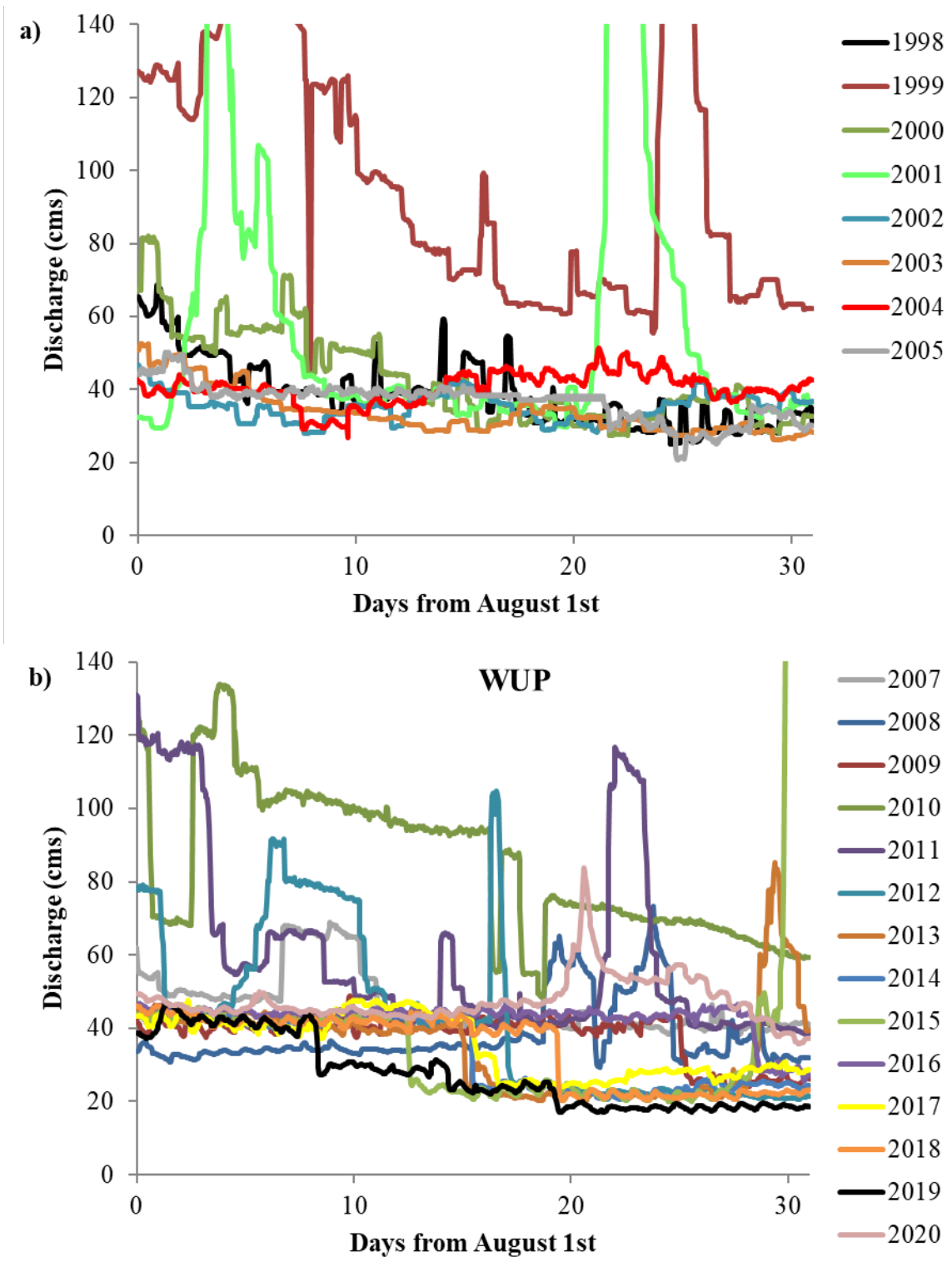


Figure 2.4. Hourly discharge at the Brackendale gauge on the Cheakamus River in August (a) during the IFA (a) and WUP (b) flow regime periods.

Multi-yr telemetry data

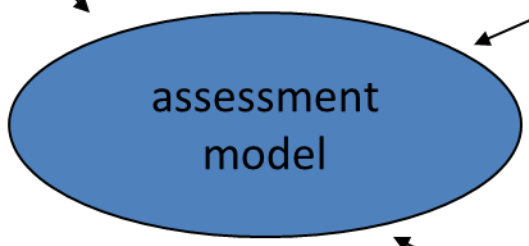
Survey life – entry date

Departure schedule

Observer efficiency

Annual swim count data

of marked and unmarked fish observed by survey



Angling data

Catch of wild and hatchery steelhead by survey

Age data

Returns by brood year (stock-recruitment)

Escapement by origin (trend)

Figure 3.1. Diagram showing how different data sets contribute to the model that estimates Cheakamus River steelhead escapement and the spawner-adult stock-recruitment model.

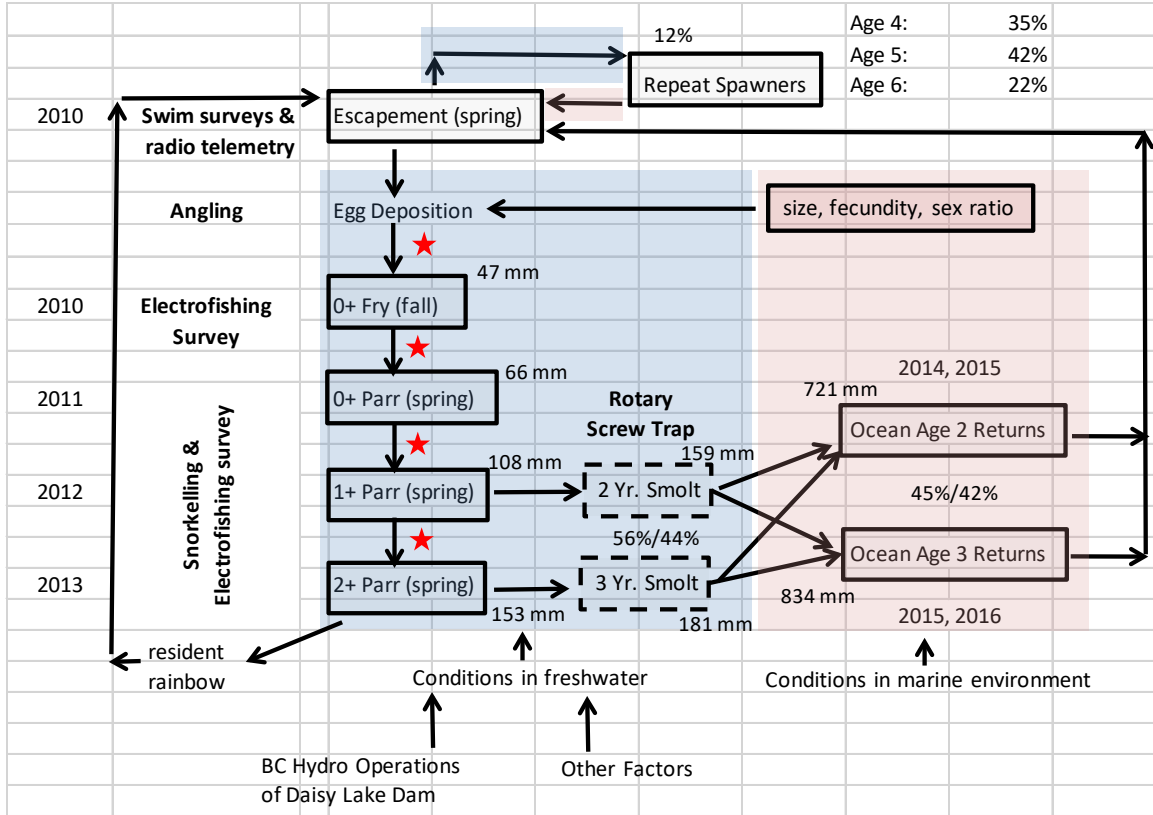


Figure 4.1. Summary of Steelhead life history in the Cheakamus River in relation to WUP monitoring activities. The years on the left of the diagram track the timing of a cohort spawned in 2010 to a 3 yr. smolt exiting the Cheakamus River in 2013 or remaining in the river as a resident trout. The average size of each life stage and proportion of fish by age class are also shown. Blue and pink shaded boxes identify life stages effected by freshwater and marine conditions, respectively.

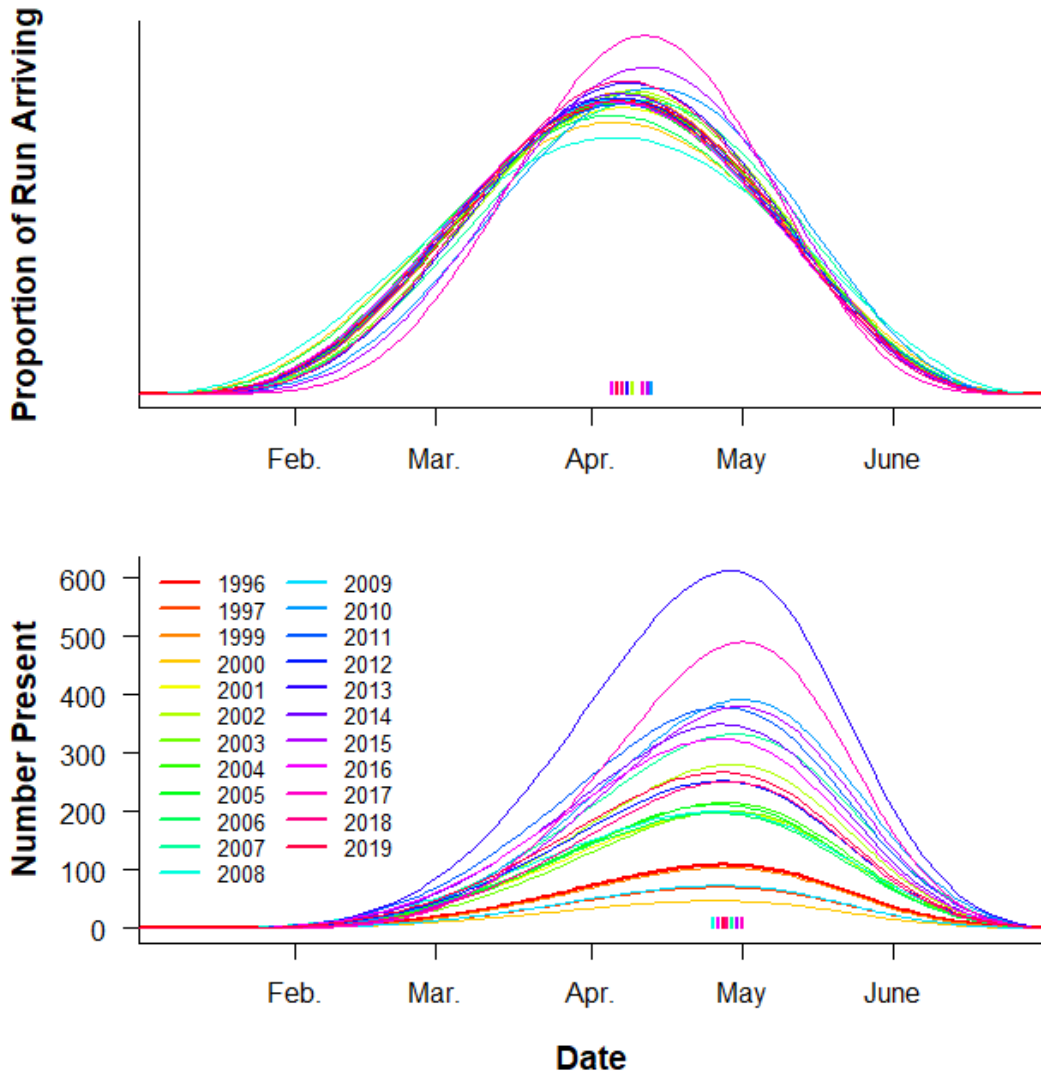


Figure 4.2. Arrival-timing of Steelhead spawners to the Cheakamus River upstream of the Cheekye-Cheakamus confluence. The top plot shows arrival timing as a proportion of the total run arriving by date. The bottom plot shows the number present in the survey area by date, which is the difference between estimates of the numbers that have arrived and the number that have departed by date. Variation in the height of the curves in the bottom plot reflects variation in annual escapements. Vertical lines at the bottom of each plot highlight the dates where the proportion arriving peaks (top) or where the maximum number of fish are present (bottom). These results are based on output from the escapement model.

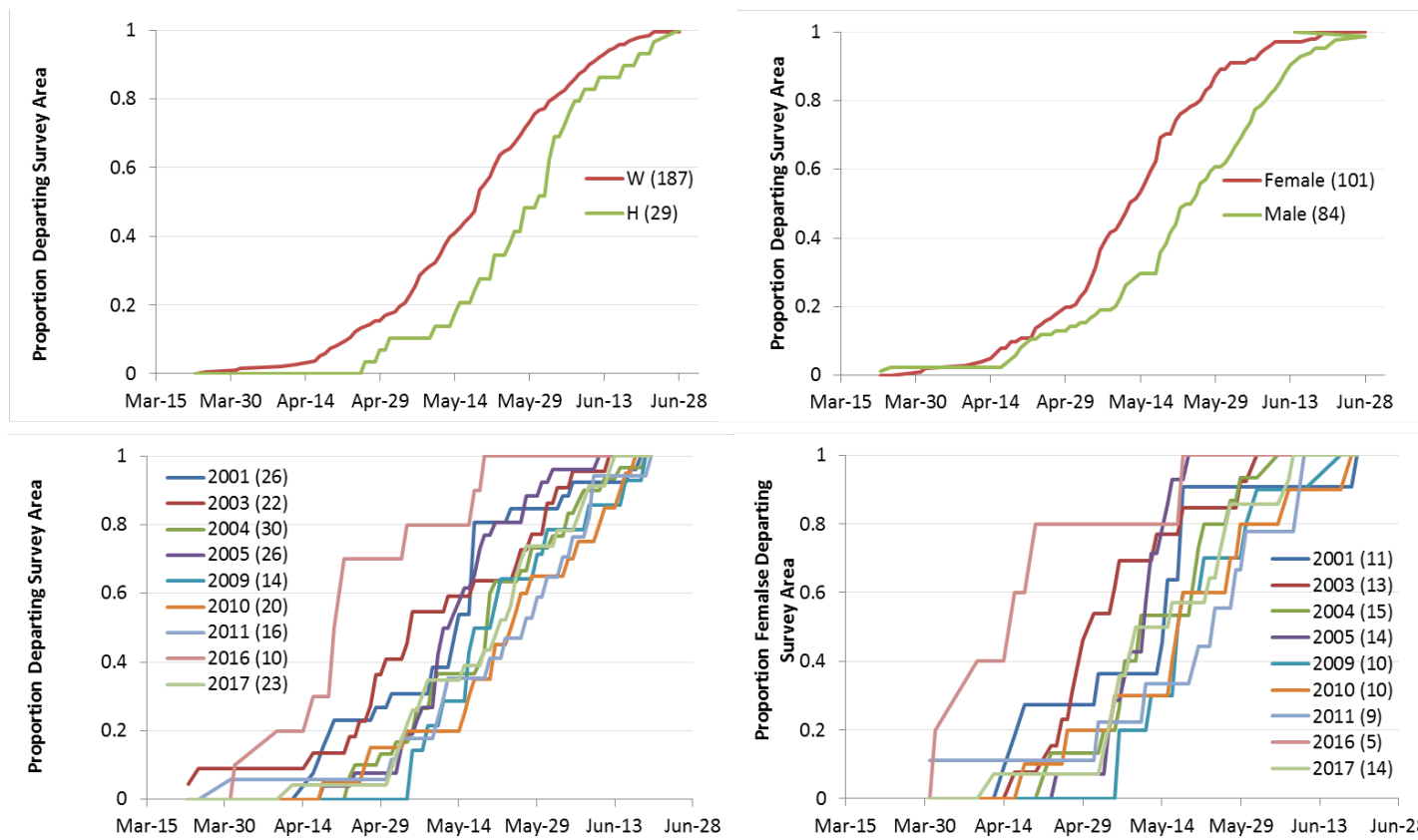


Figure 4.3. Cumulative proportion of radio tagged steelhead departing the Cheakamus River survey area by date based on data from all years radio telemetry was conducted. Numbers in parentheses in the legends denote the sample size (# of tagged fish recorded as departing at the fixed station). Owing to differences in departure timing of wild- and hatchery-origin steelhead, hatchery fish are only included in the Hatchery-Wild comparison plot.

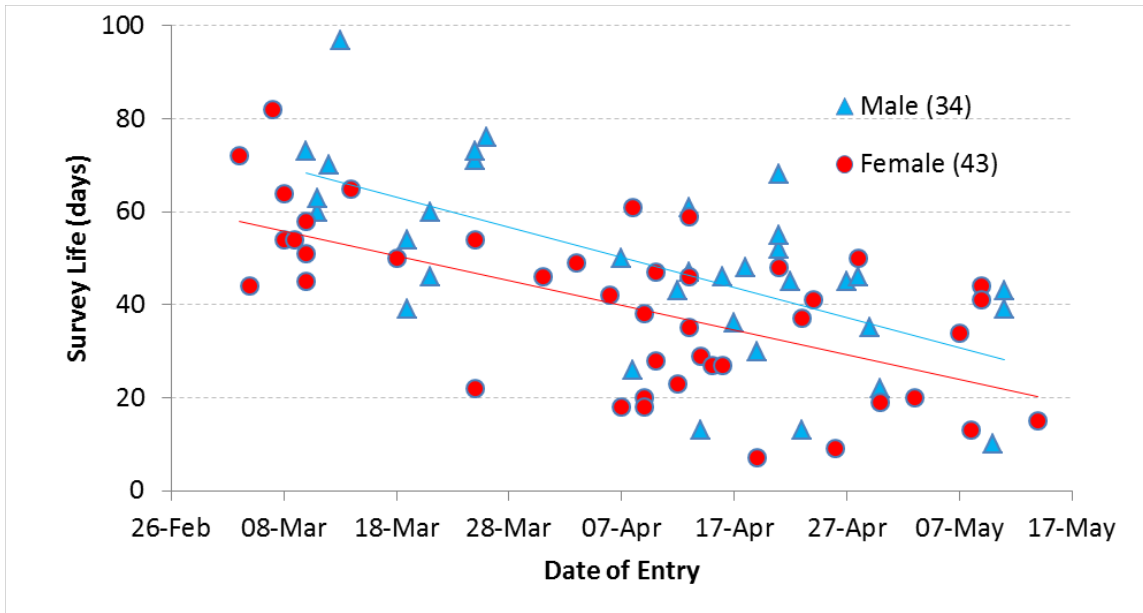


Figure 4.4. Relationship between date of entry and duration of time spent in survey area (upstream of the Cheekye-Cheakamus confluence) for male and female steelhead in the Cheakamus River based on data from all years telemetry was conducted. Numbers in parentheses in the legend denote the sample size.

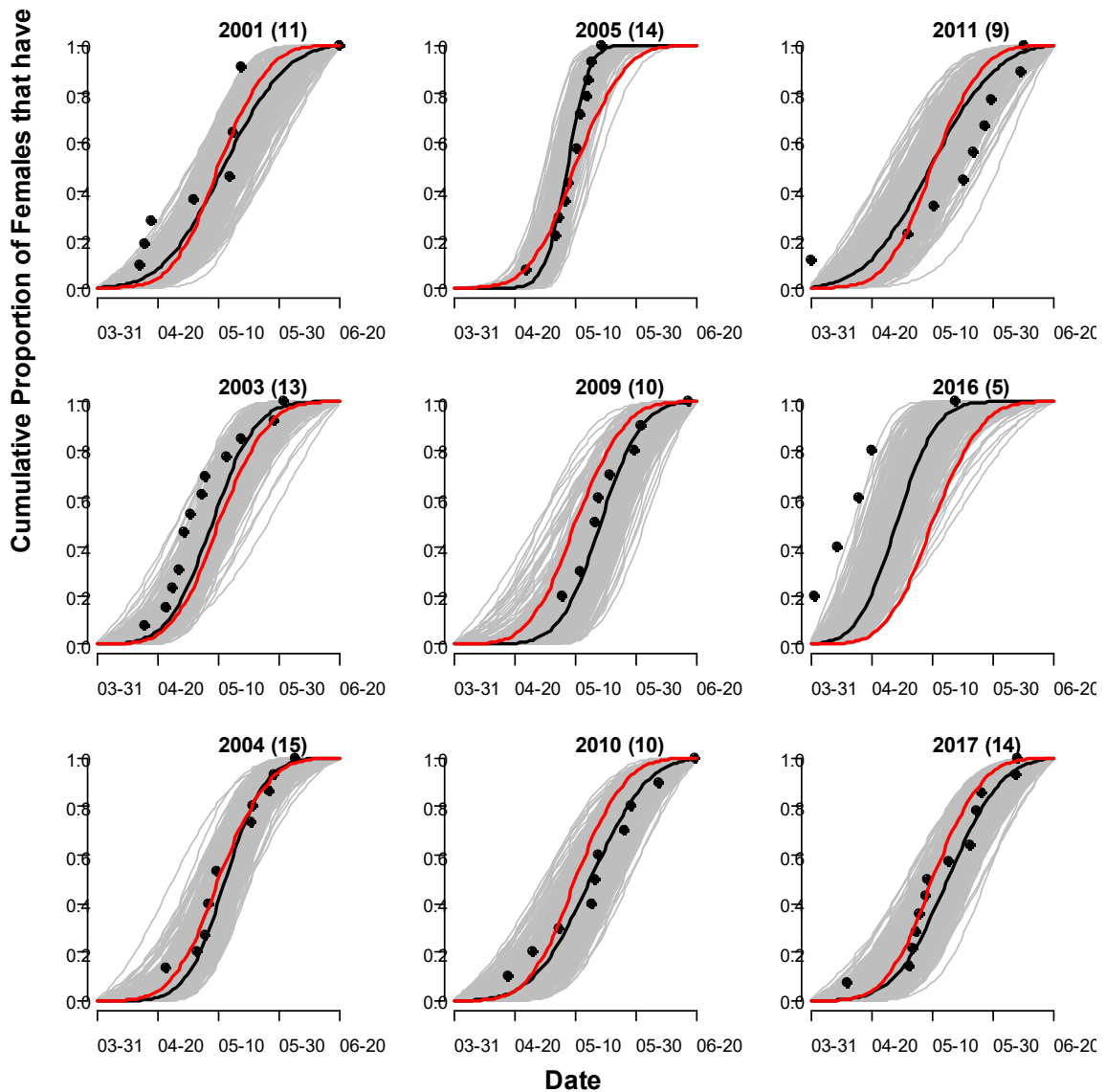


Figure 4.5. Estimated timing of departure from the Cheakamus River survey area for radio tagged female steelhead. Points show the cumulative proportion of females that have departed by date. The black curves show the best-fit departure timing for that year and the red curve shows the average curve across years. Light grey curves show the extent of uncertainty in year-specific predictions based on a random sample from posterior distributions of departure-timing parameter estimates. Numbers in parentheses at the top of each plot denote the number of observations (# of radio tagged females where departure date was determined).

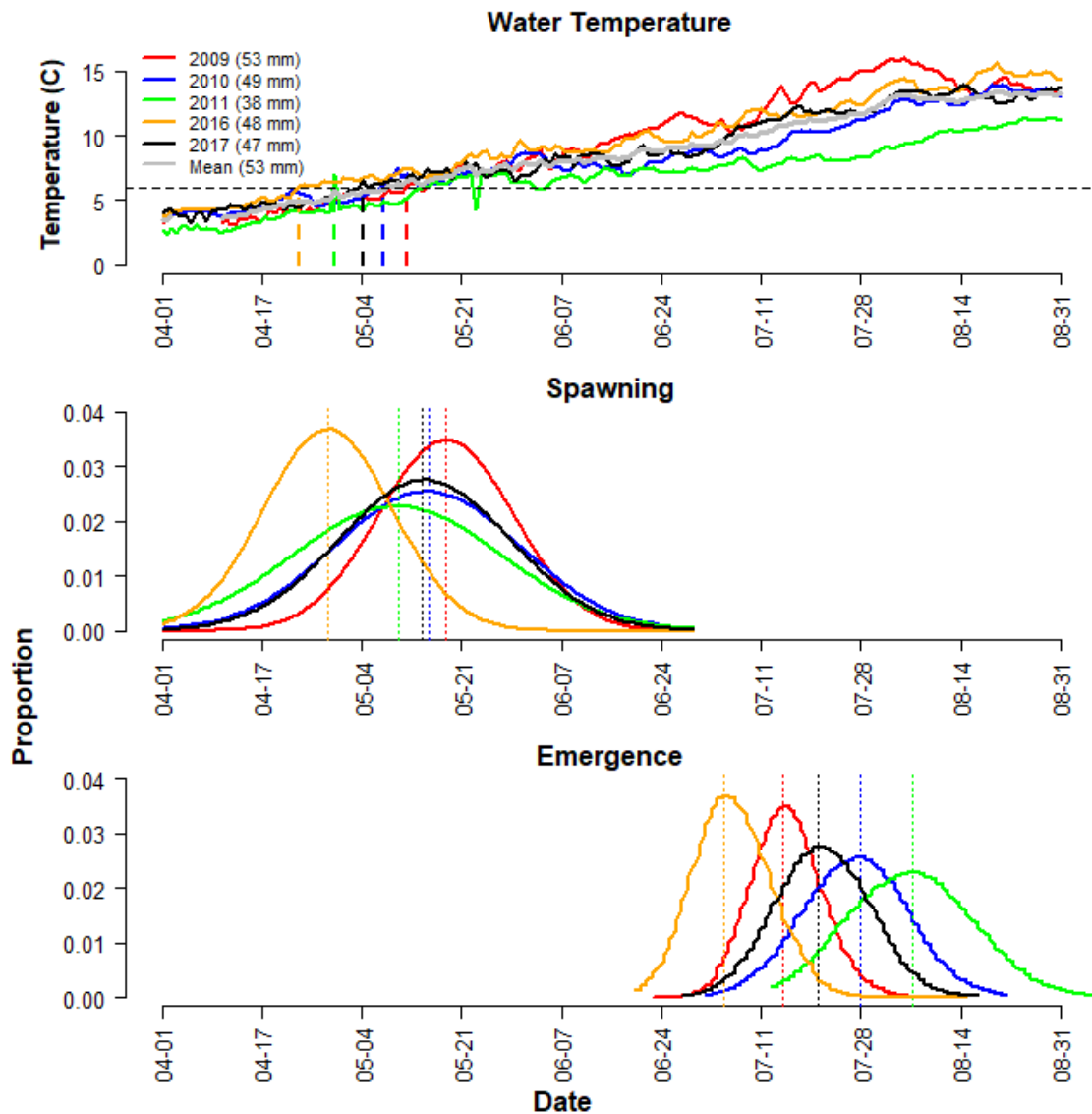


Figure 4.6. Predicted emergence timing of Steelhead in the Cheakamus River based on water temperature and spawn-timing distributions. Water temperatures over the potential incubation period are shown in the top panel. The middle panel shows spawn-timing distributions based on the modelled departure date of radio-tagged female steelhead (see Fig 4.5). Emergence timing is shown in the bottom panel. Vertical dashed lines denote the median spawn and emergence dates for each year. The horizontal line in the top panel denotes 6 °C (the minimum temperature for spawning), and the thick vertical lines below it show the date when this limit is first exceeded. The date this temperature limit is reached precedes the date of peak spawning shown by the vertical dashed lines in the middle panel.

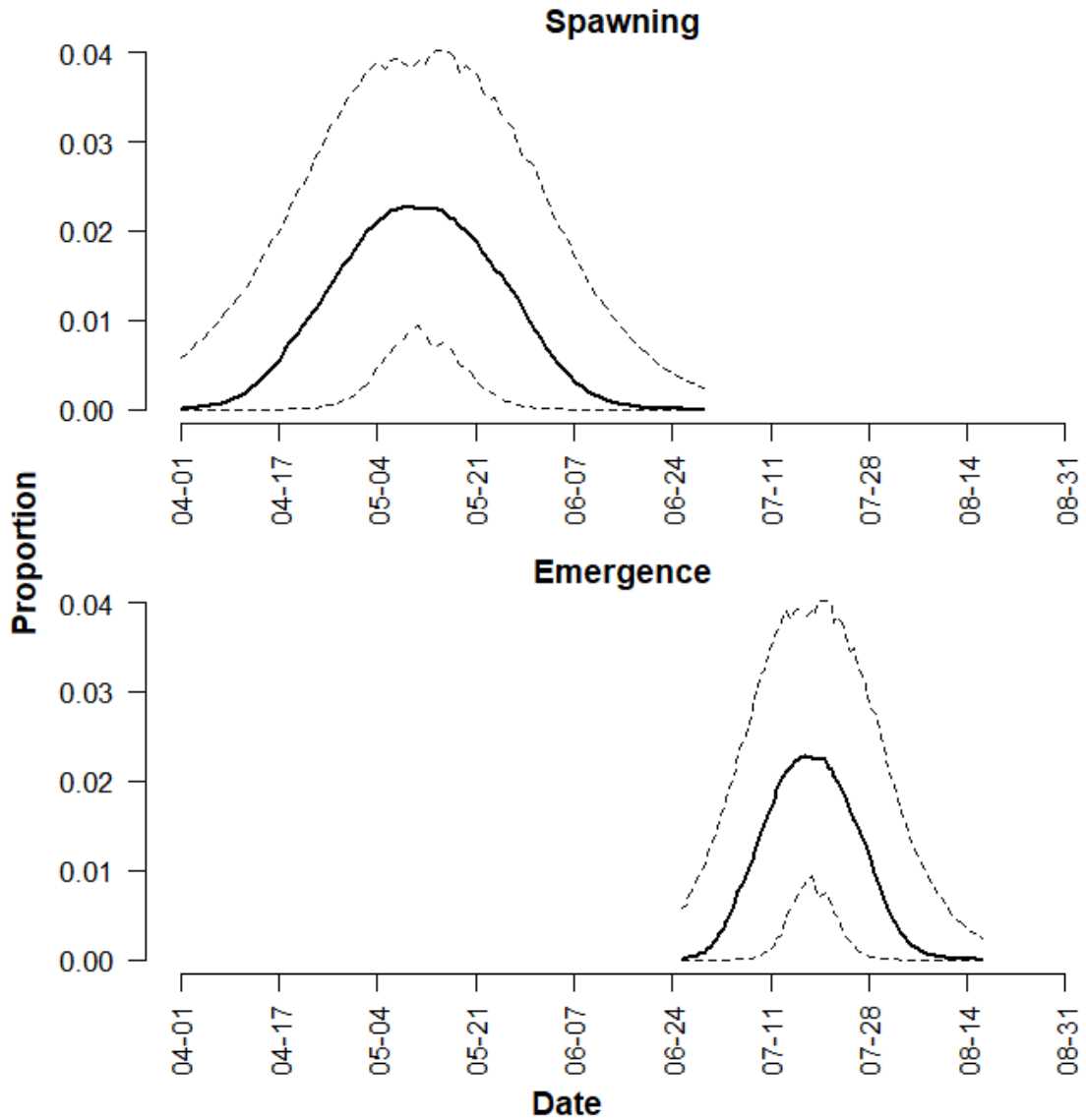


Figure 4.7. Average spawning and emergence timing for Steelhead in the Cheakamus River. Solid and dashed lines show the mean and 80% credible interval. Results are based on the average number of days to emergence by day between 2008 and 2017 (based on water temperatures) and the average spawn timing curve based on data from all years when telemetry was conducted (red curves in Fig. 4.5).

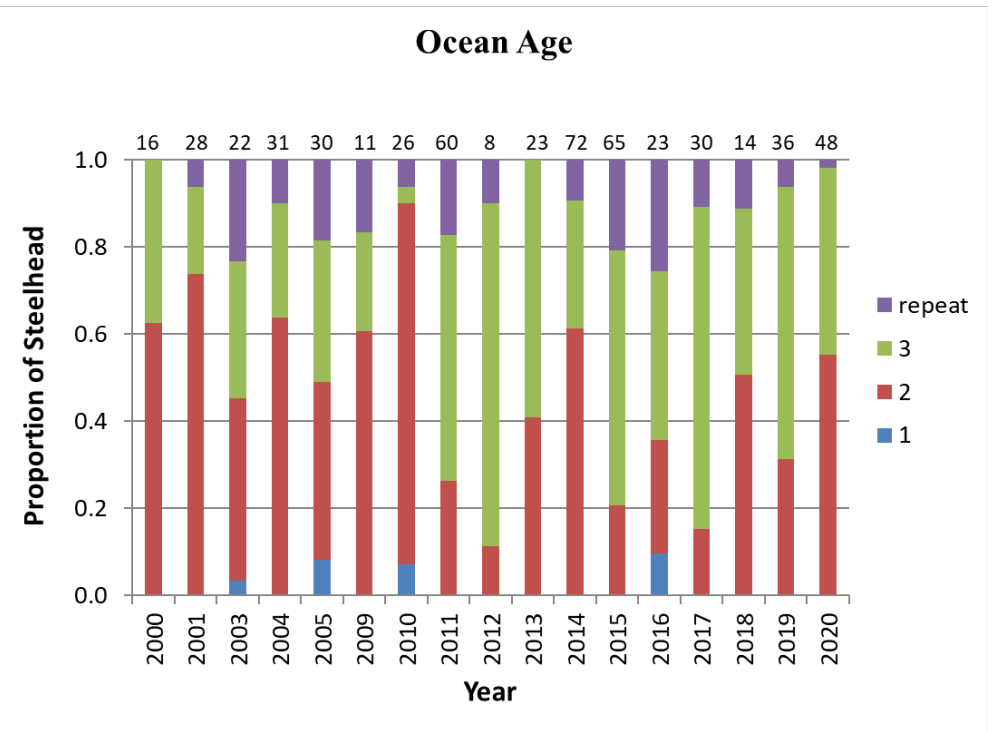
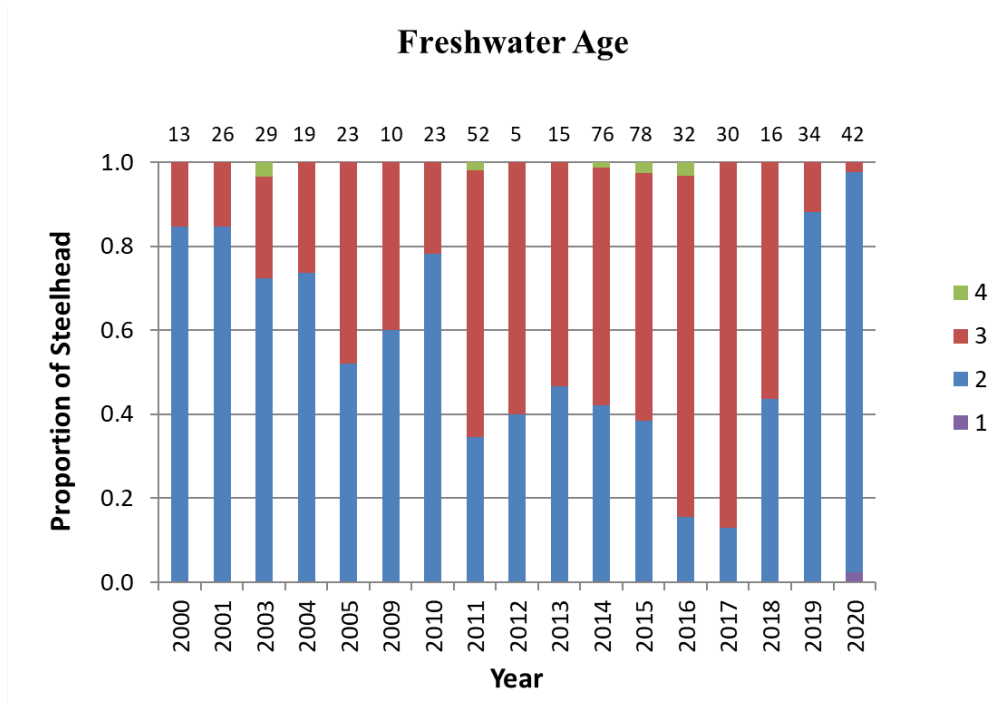


Figure 4.8. Proportion of Steelhead in the Cheakamus River by freshwater and ocean age as determined by scales collected from returning spawners. Freshwater and ocean age (in years) is the number of winters spent in freshwater and the ocean, respectively. Year on the x-axis denotes the year that scales were collected from returning spawners. Numbers at the top of each bar denote the number of scales where a freshwater or ocean age could be determined (sample size).

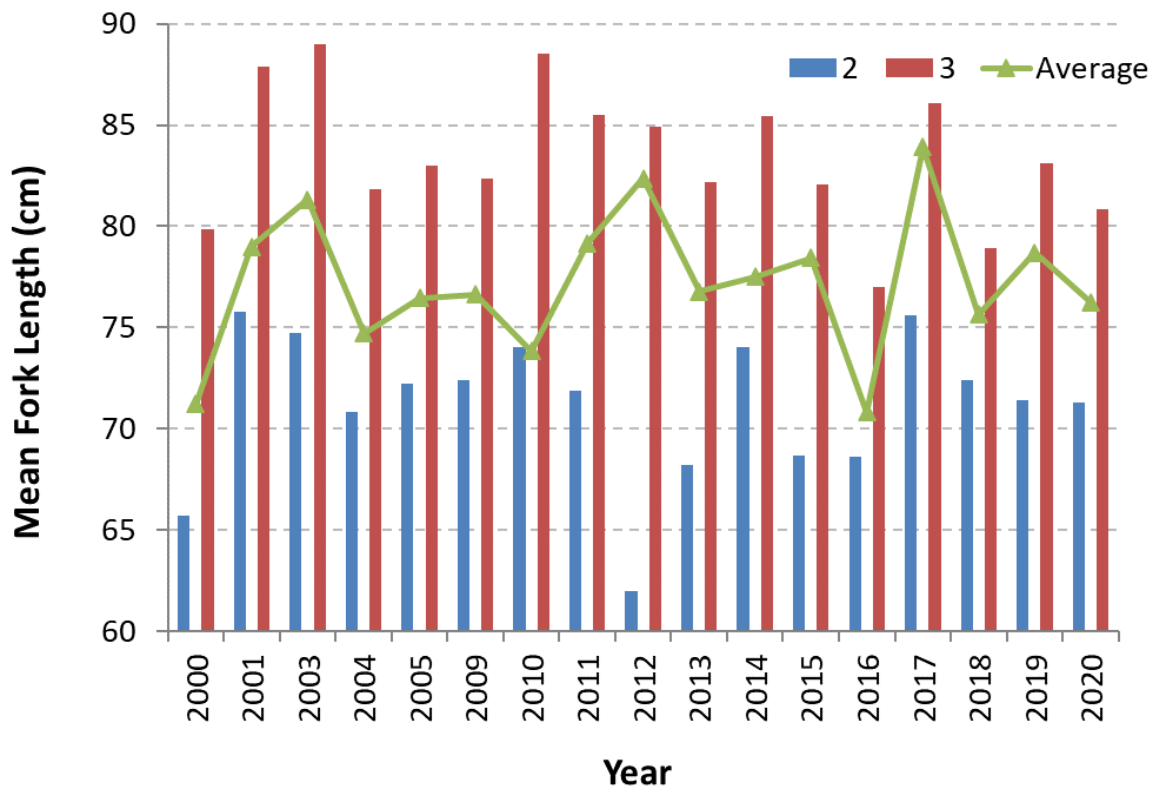


Figure 4.9. Mean size of returning Steelhead spawners by ocean age in years (bars), and the average size for all fish captured (green line).

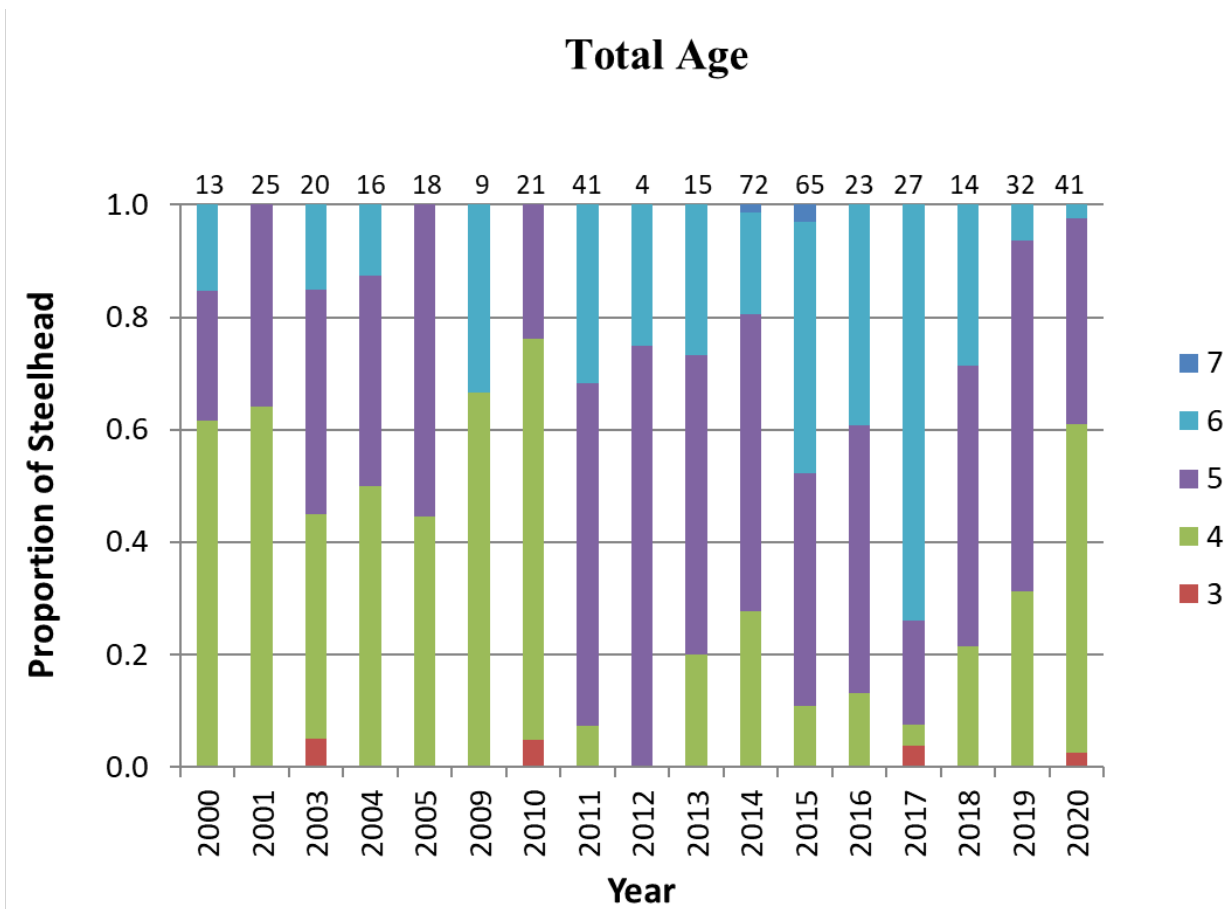


Figure 4.10. Proportion of Steelhead spawners returning to the Cheakamus River by total age. Numbers at the top of each bar denote the number of fish where a total age could be determined (i.e., both freshwater and ocean age could be determined).

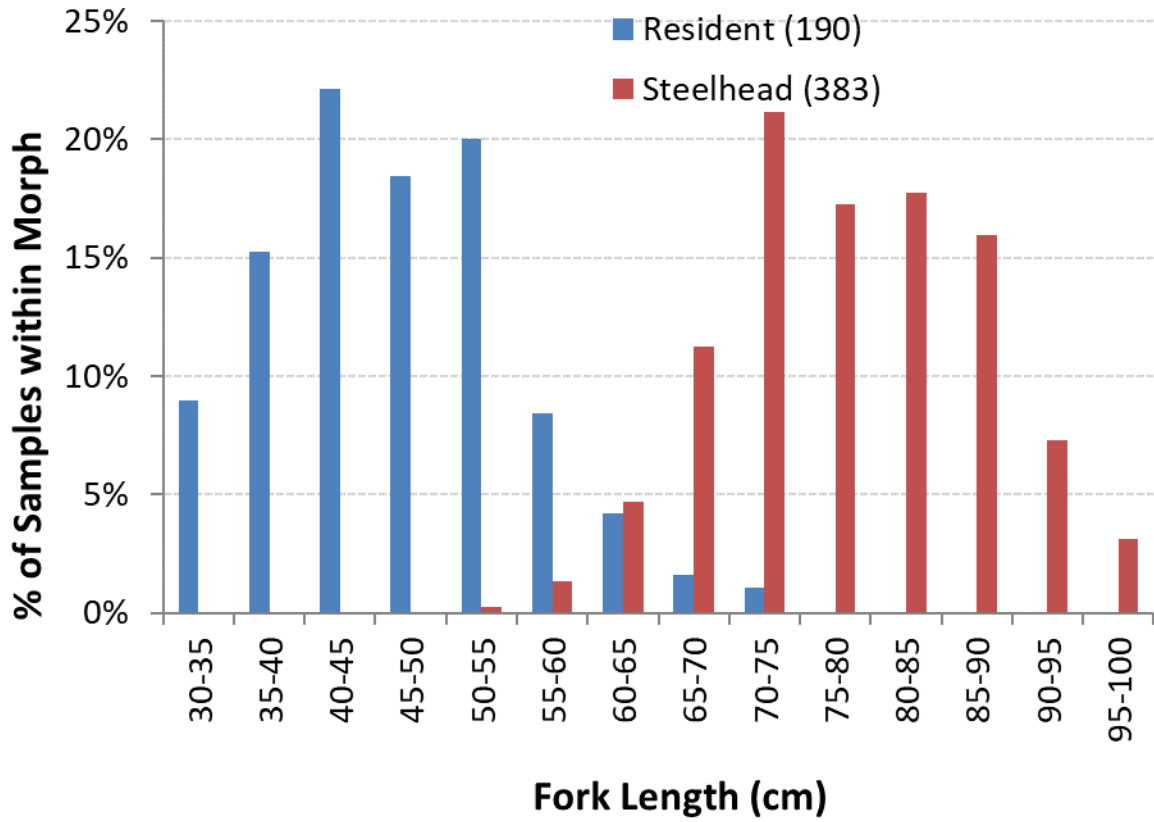


Figure 4.11. Comparison of size distribution of resident rainbow trout and steelhead in the Cheakamus River based on collection of 388 scales between 2009 and 2020.

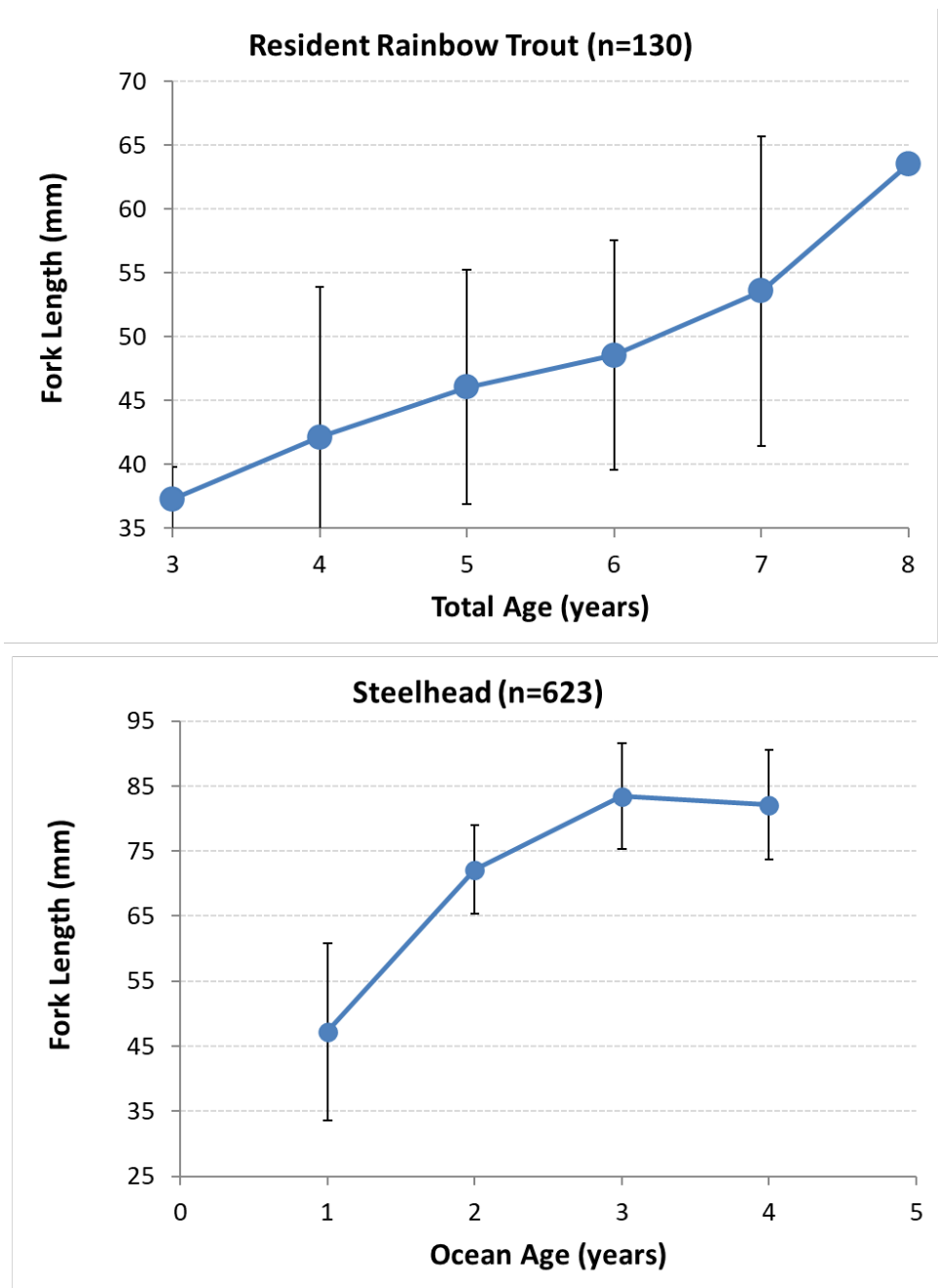


Figure 4.12. Mean fork length of resident rainbow trout (top) and steelhead (bottom) captured in the Cheakamus River between 2000 and 2019. Error bars show 1 standard deviation around the mean. ‘n’ at the top of each plots denotes the sample size (# of fish where total age could be determined).

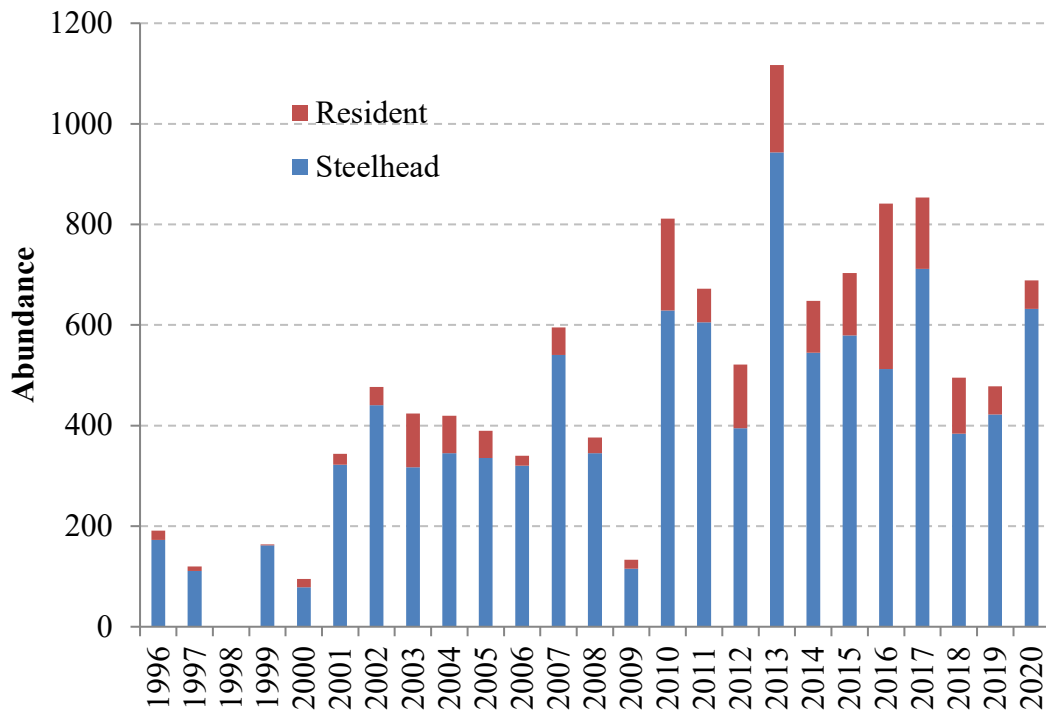


Figure 4.13. Abundance of returning steelhead (escapement) and resident rainbow trout in the Cheakamus River. The total height of the bar is the combined abundance of Steelhead and resident trout.

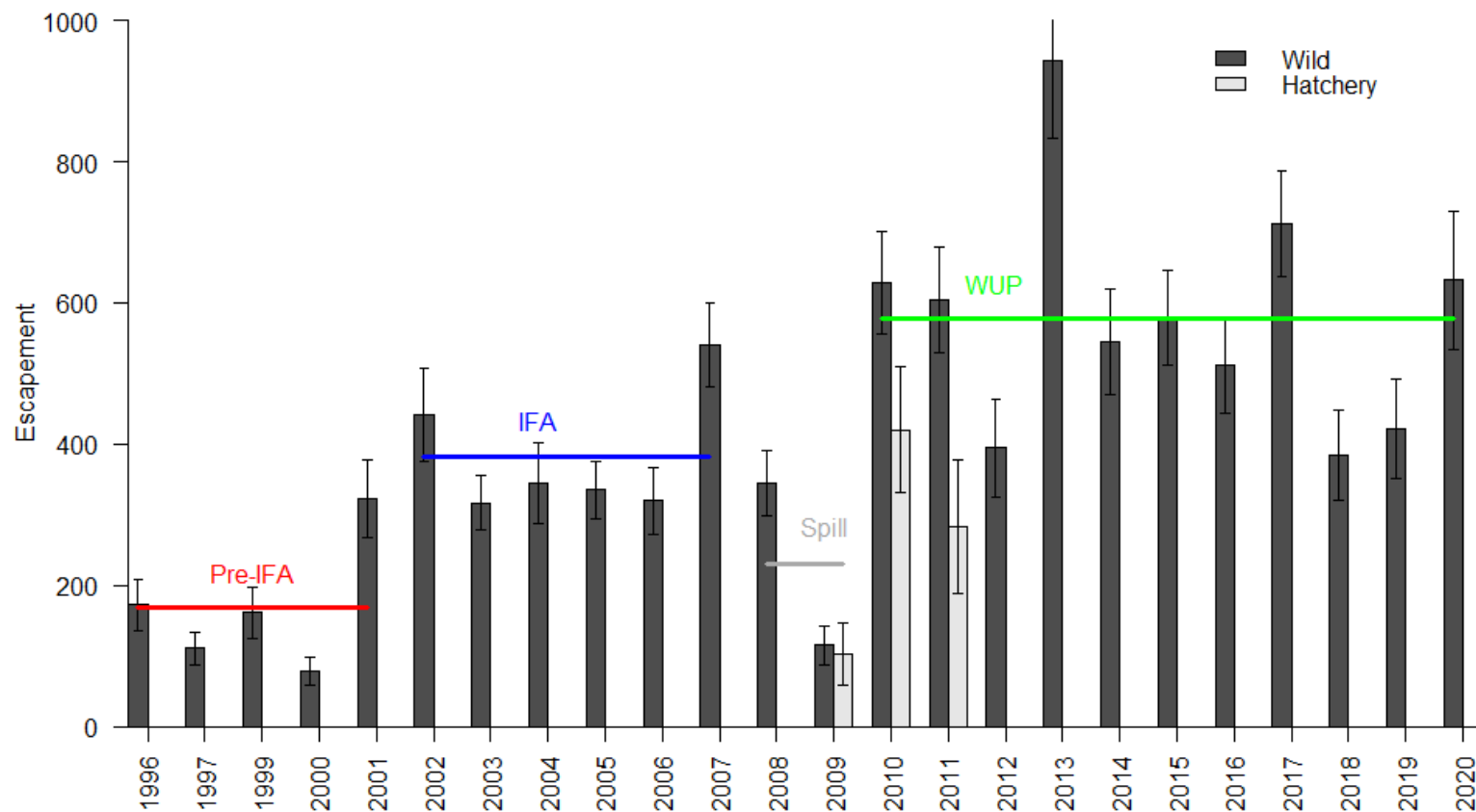


Figure 5.1. The Steelhead escapement trend in the Cheakamus River, 1996-2018. The height of the bars and error bars show the most likely escapement estimates and 80% credible intervals, respectively. The colored horizontal lines show the average escapement for years where the returns had reared as juveniles before and after the Interim Flow Agreement (pre-IFA and IFA, respectively) and under the Water Use Plan flows (WUP), respectively. Also shown are years where returns were reduced due to the CN caustic soda spill.

a) Beverton-Holt

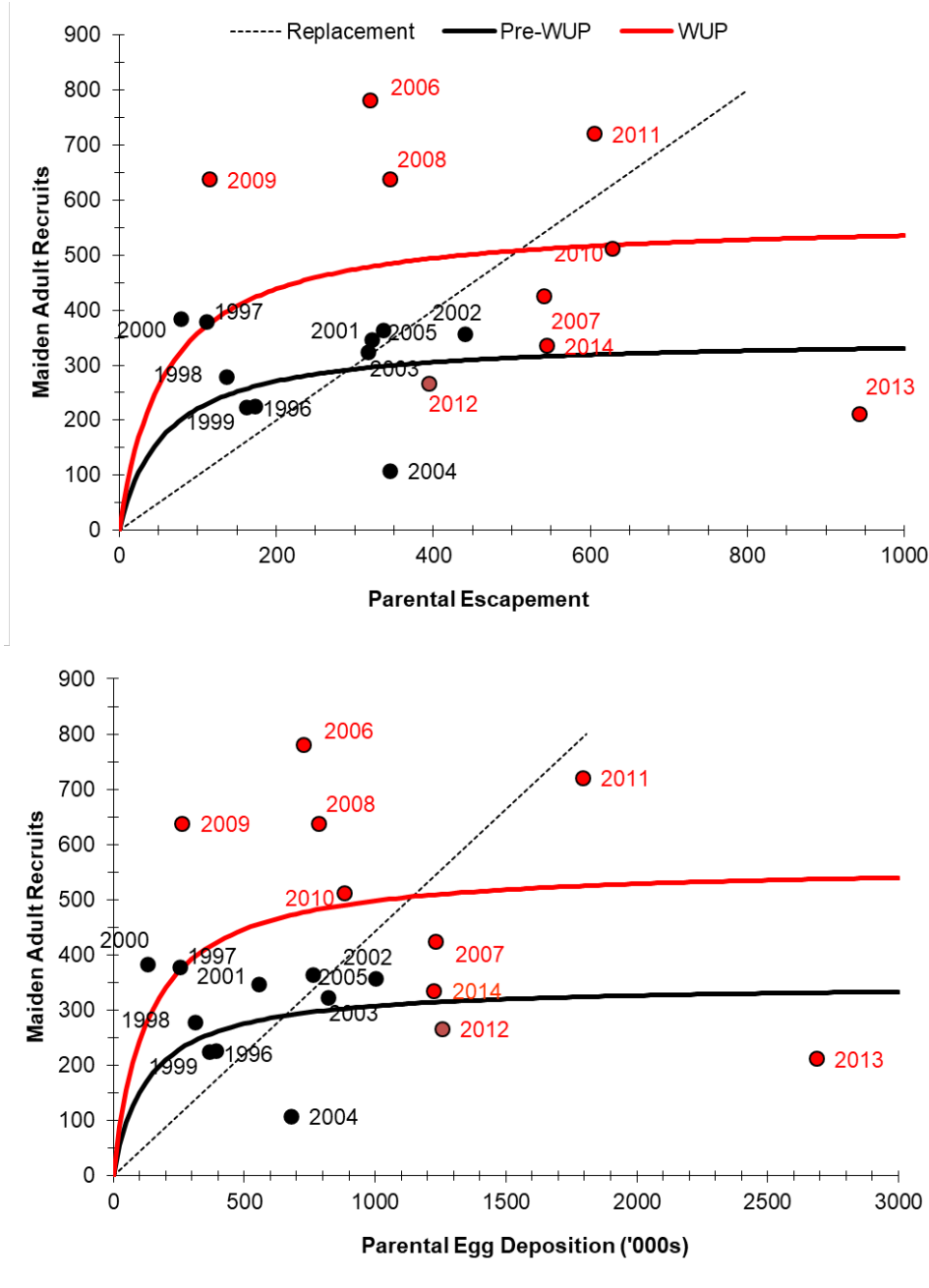


Figure 5.2. The relationship between the number of Steelhead spawners (top) and egg deposition (bottom) and the resulting maiden adult returns (total returns less repeat spawners) to the Cheakamus River. Recruitments based on fish that reared in the river under WUP flows are highlighted in red. The year beside each point represents the brood year (year of spawning). The dashed lines show the number of spawners or egg deposition required for replacement (spawners=recruits). Black and red lines represent fits based on a productivity-constrained Beverton-Holt stock-recruit relationship under pre-WUP and WUP flows (a) and the unconstrained Ricker model fit (b).

b) Ricker Model

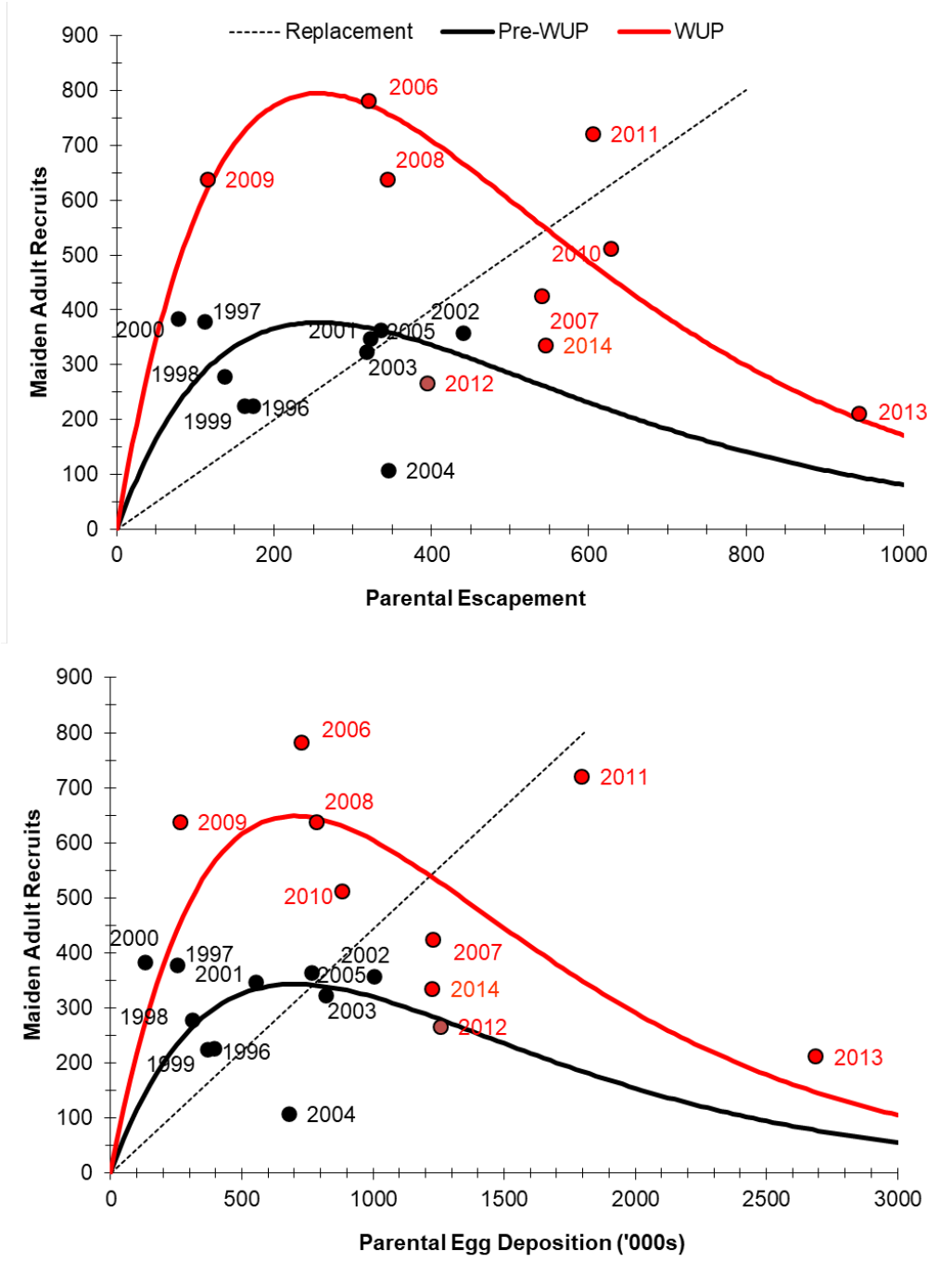


Figure 5.2. Con't.

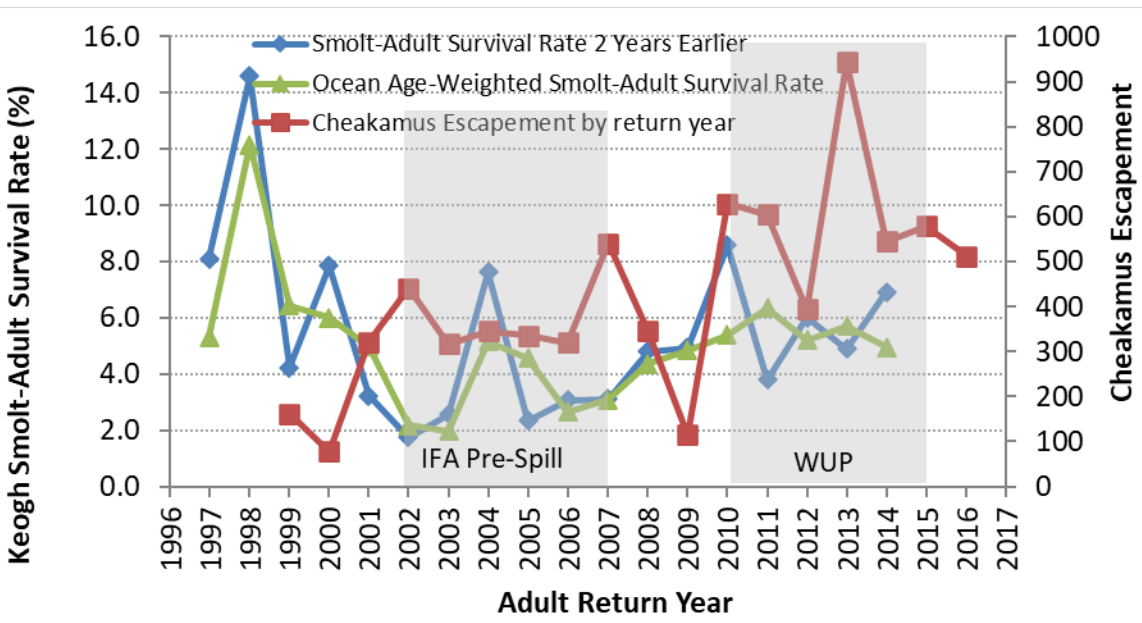
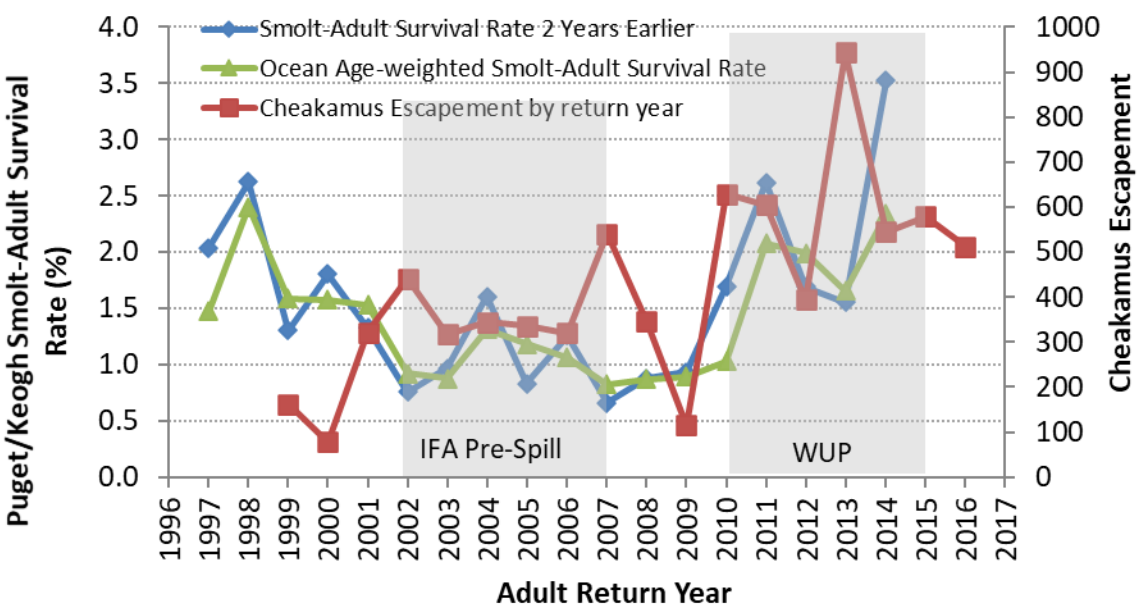


Figure 5.3. Trends in Steelhead escapement to the Cheakamus River by return year in comparison to the smolt-adult survival rate (SAR) for smolts outmigrating two years earlier, and the ocean-age proportion-weighted SAR, for wild winter run stocks in Puget Sound and the Keogh River (top) as compiled by Kendall et al. (2017), and using the Keogh River data alone (bottom). The shaded grey boxes identify the years used to compute average escapement and SAR for IFA (pre CN-spill effected years) and WUP periods.

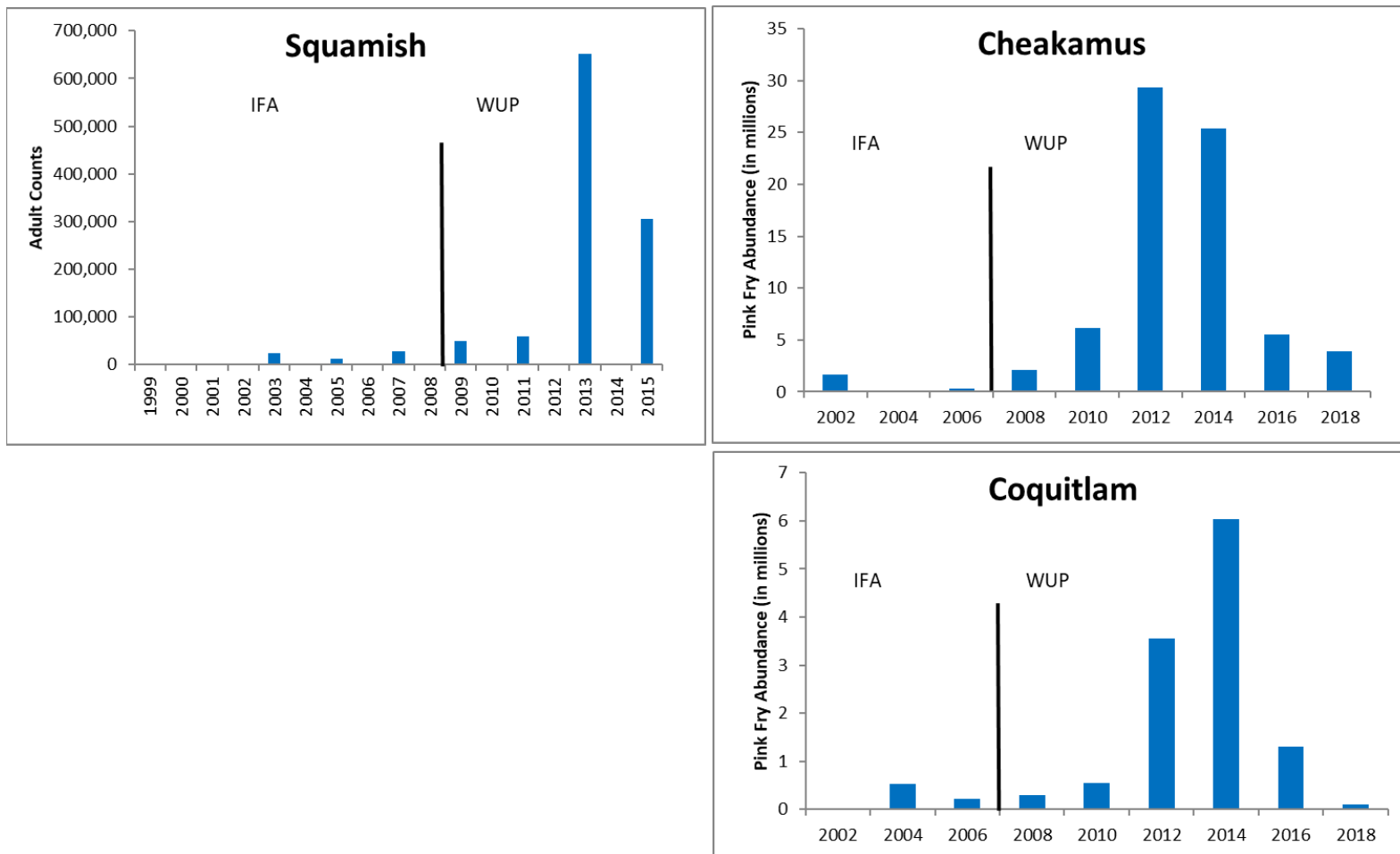


Figure 5.4. Trends of pink salmon abundance. The top-left panel shows the index of pink salmon returns in the Squamish watershed as a whole from stream walks conducted by the Squamish First Nation (SFN). Plots on the right show run size of pink salmon fry from Rotary Screw Traps which occur in the calendar year following the return year.

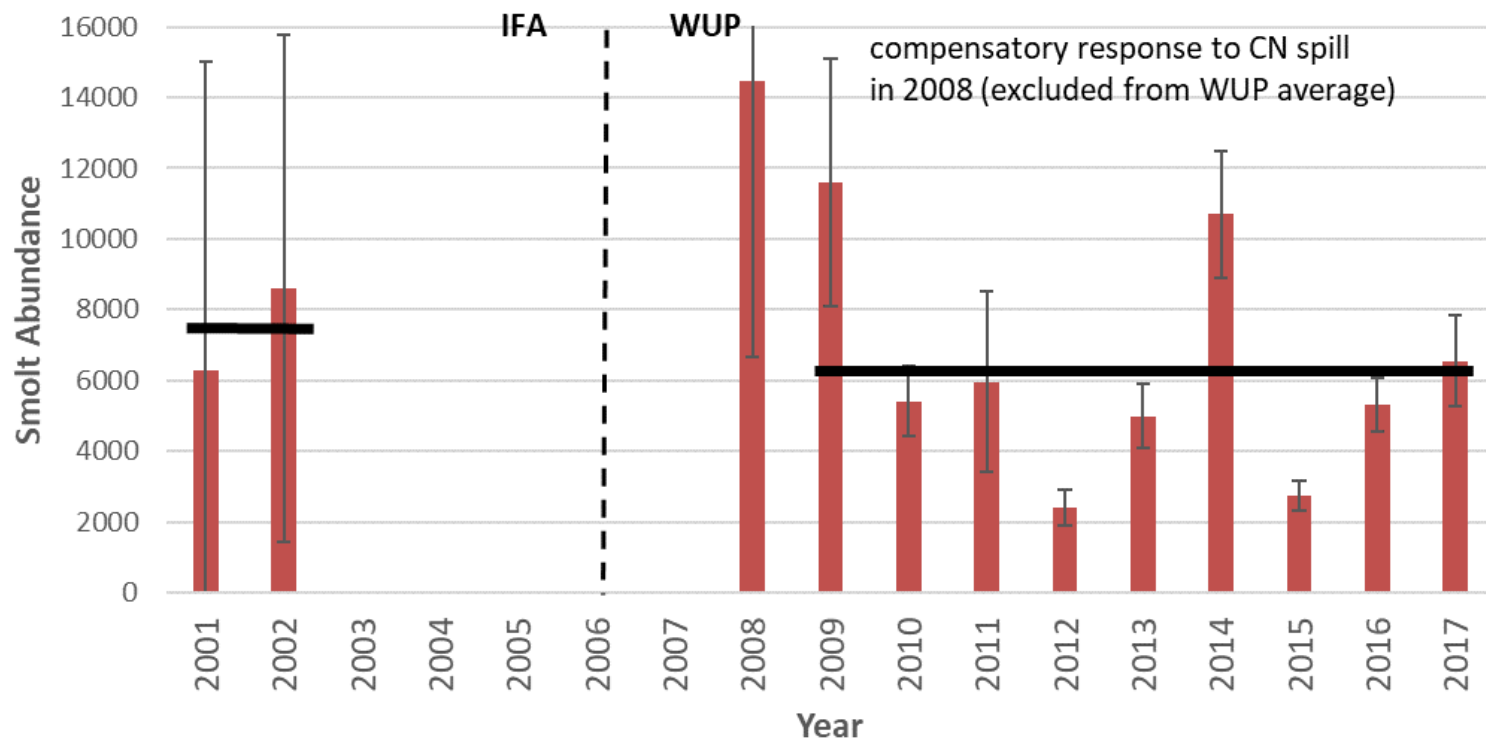


Figure 5.5. Estimates of Steelhead Trout smolt abundance at the Rotary Screw Trap on the Cheakamus River. Black horizontal. Black horizontal bars show the mean abundance during the IFA and WUP periods. 2008 was excluded from the WUP period due to CN caustic soda spill effects. The height of the bars represent the means of the posterior distributions, and the error bars represent one standard deviation.

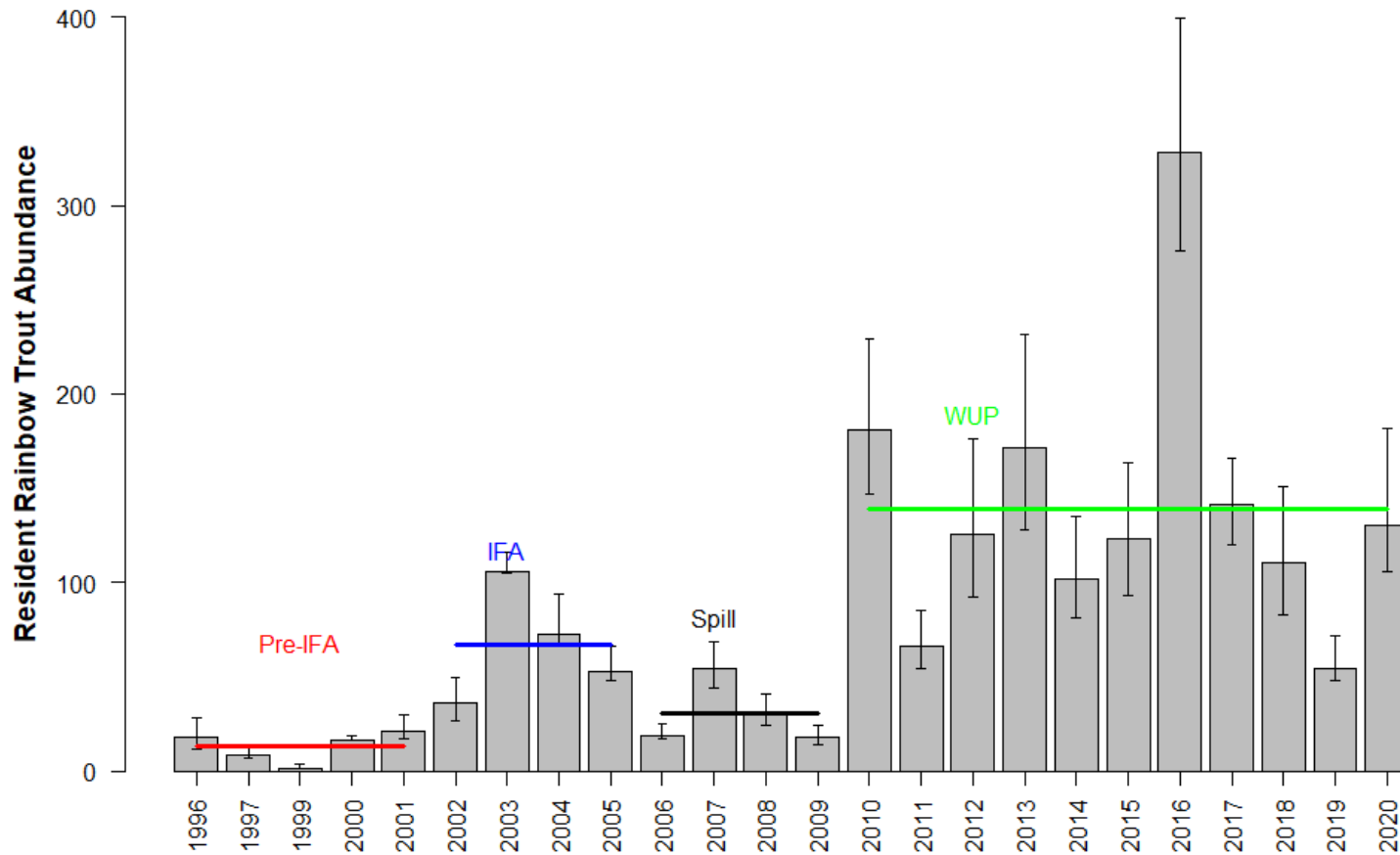


Figure 5.6. Resident rainbow trout abundance estimates in the Cheakamus River upstream of the Cheekye River confluence. The height of bars and error bars represent the mean and 95% credible intervals for annual estimates. Assuming a minimum age of 4 yrs, old, the average abundance for resident trout rearing in the Cheakamus River prior to the Instream Flow Agreement (pre-IFA), during the IFA period (IFA), during the period effected by the CN spill (Spill), and during the WUP period (WUP) are shown by red, blue, black, and green horizontal lines, respectively.

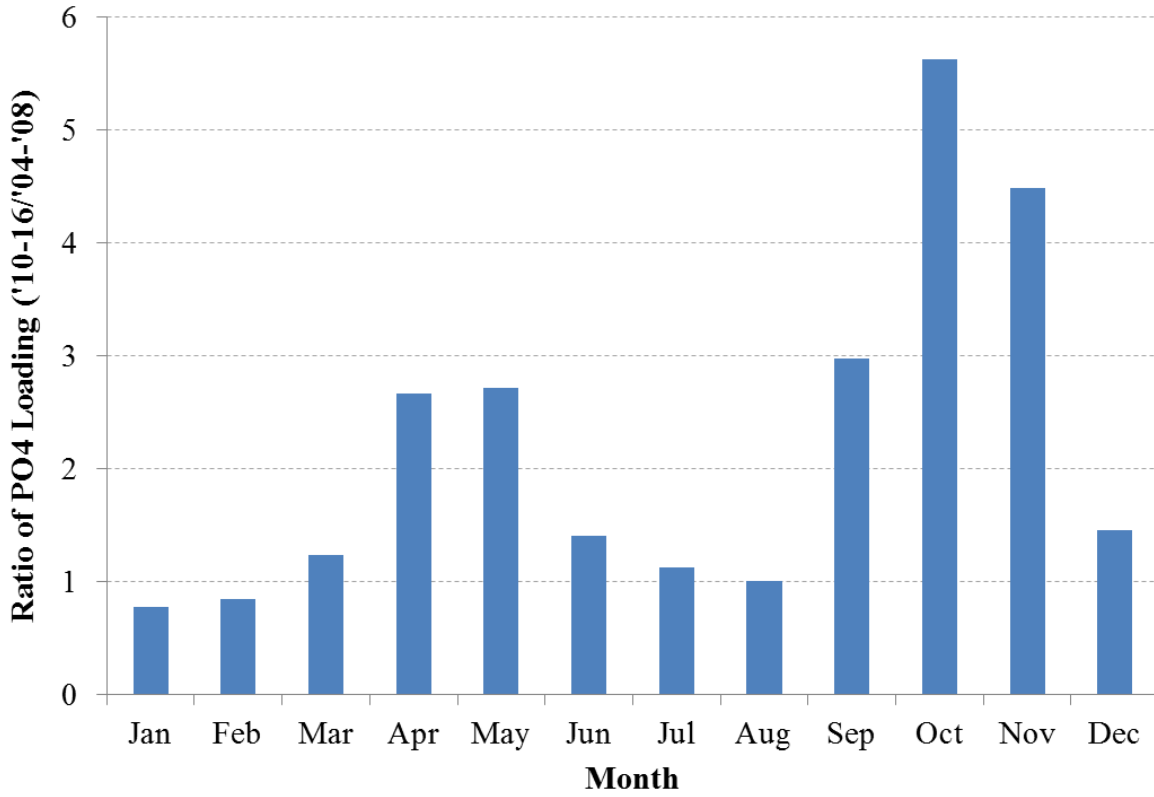


Figure 5.7. Change in phosphate loading from the Whistler sewage treatment plant. The height of the bars is the ratio of the average phosphate loading (kg/day) by month from 2010-2016 to the average from 2004-2008.

a) Cheakamus River

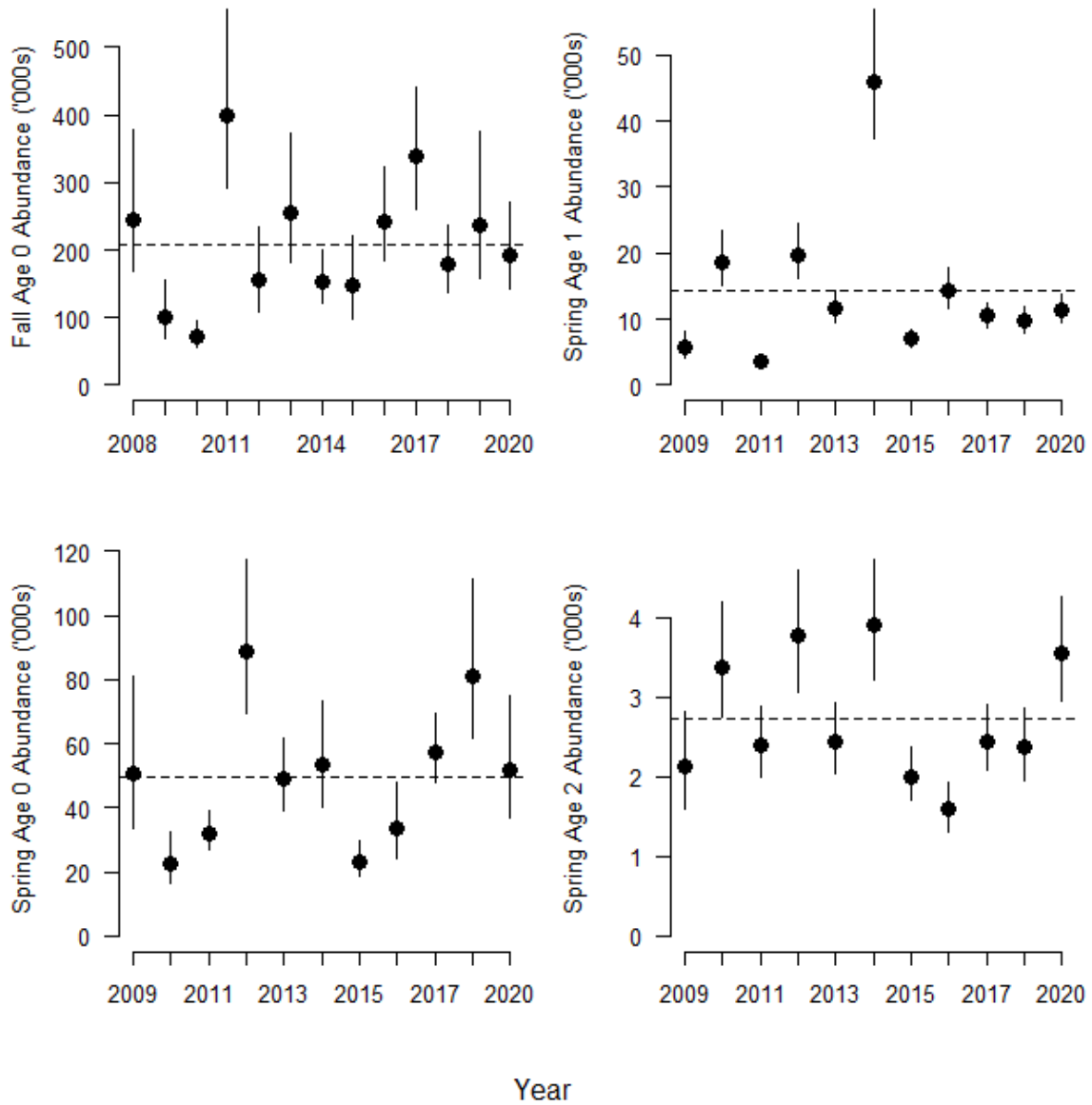


Figure 6.1. Juvenile steelhead abundance estimates in Cheakamus (a) and Brohm (b) Rivers. The height of bars and error bars represent median values and the 95% credible intervals, respectively. The dashed horizontal lines show the across-year averages.

b) Brohm River

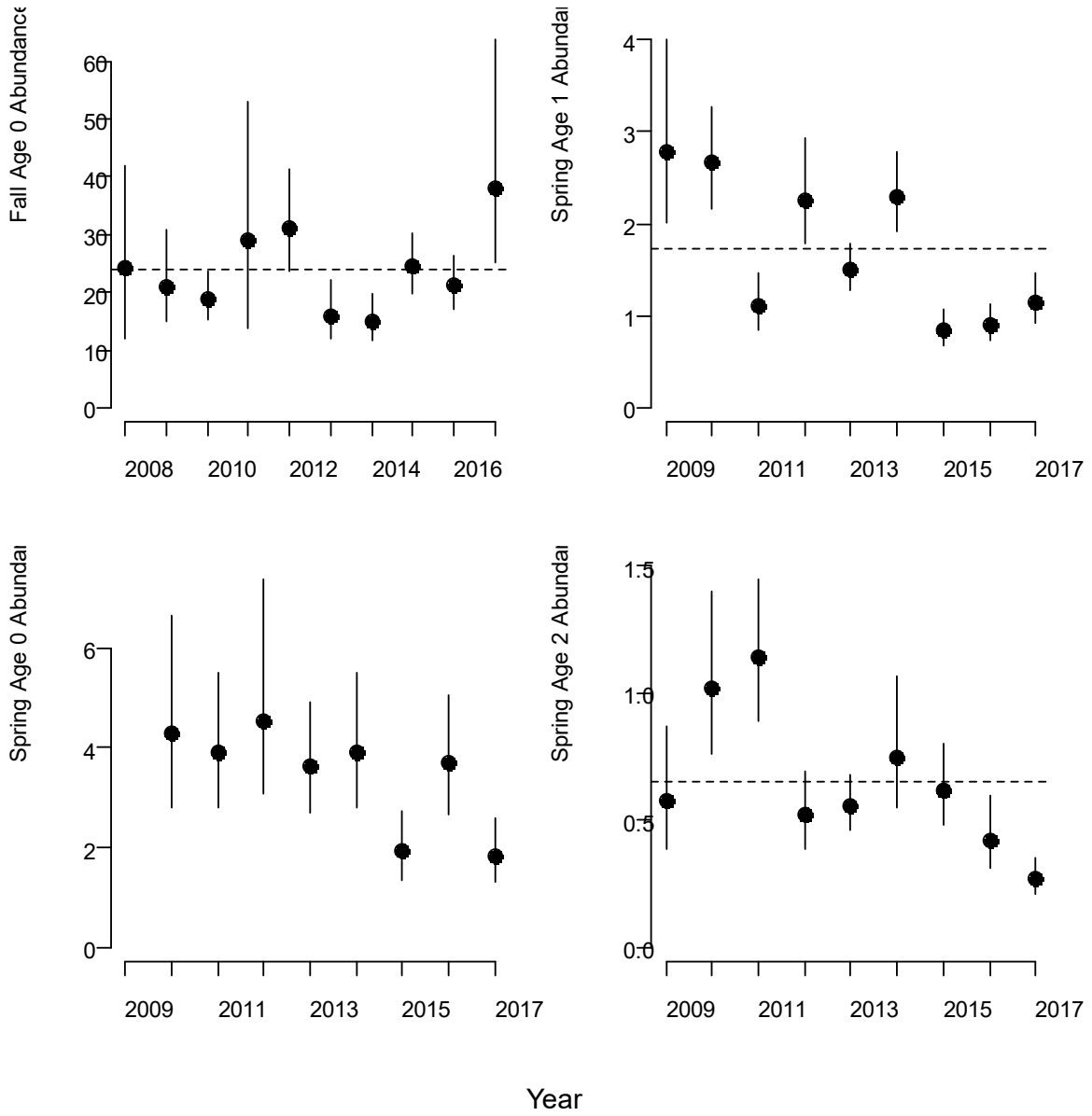


Figure 6.1. Con't.

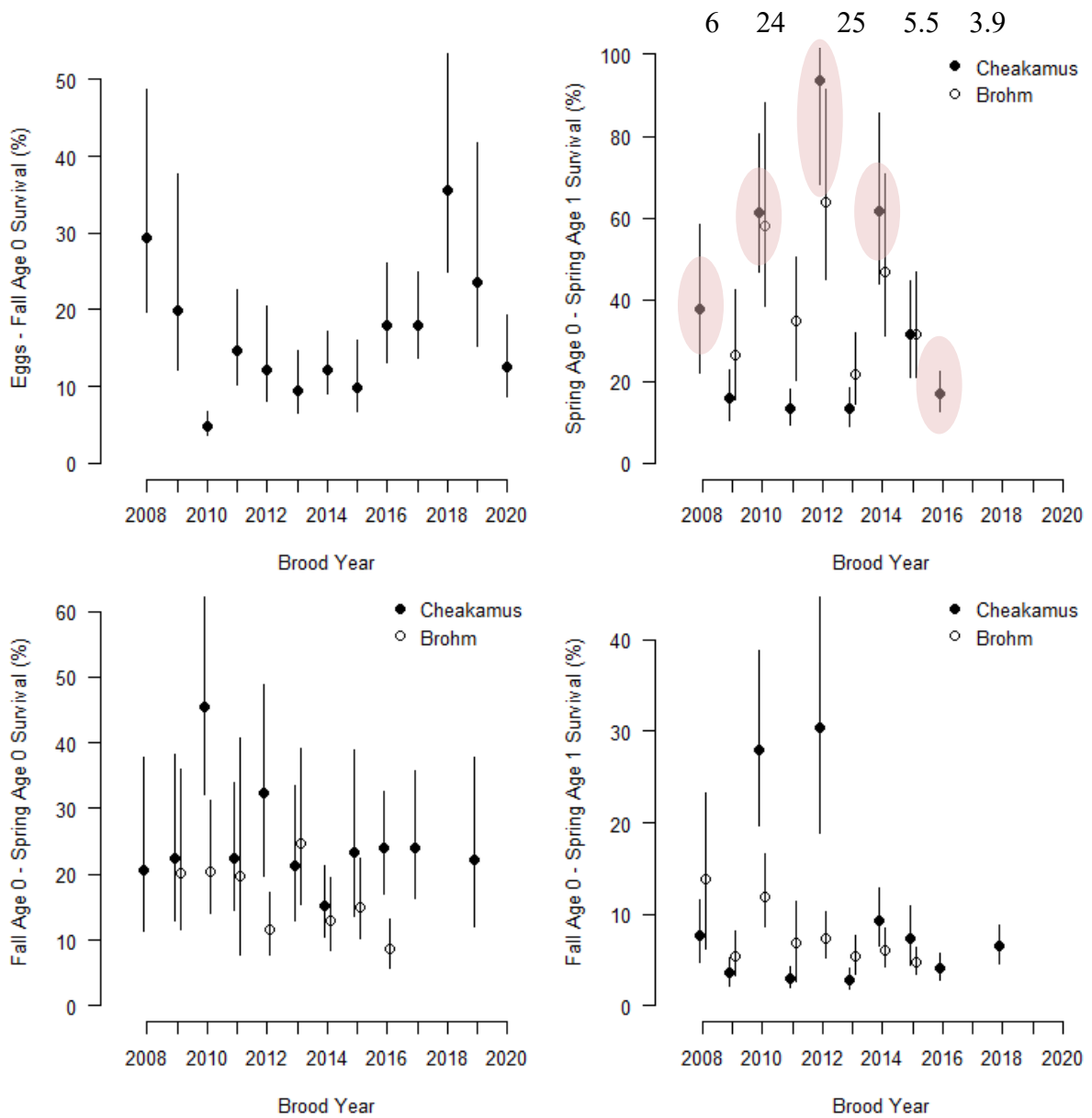


Figure 6.2. Survival for different Steelhead life stages in the Cheakamus and Brohm Rivers by brood year. Points and vertical lines denote means and 95% credible intervals, respectively. Numbers at the top of the top-right panel are estimates of the number of outmigrating pink salmon fry (in millions) for each pink salmon return year (identified by pink ovals).

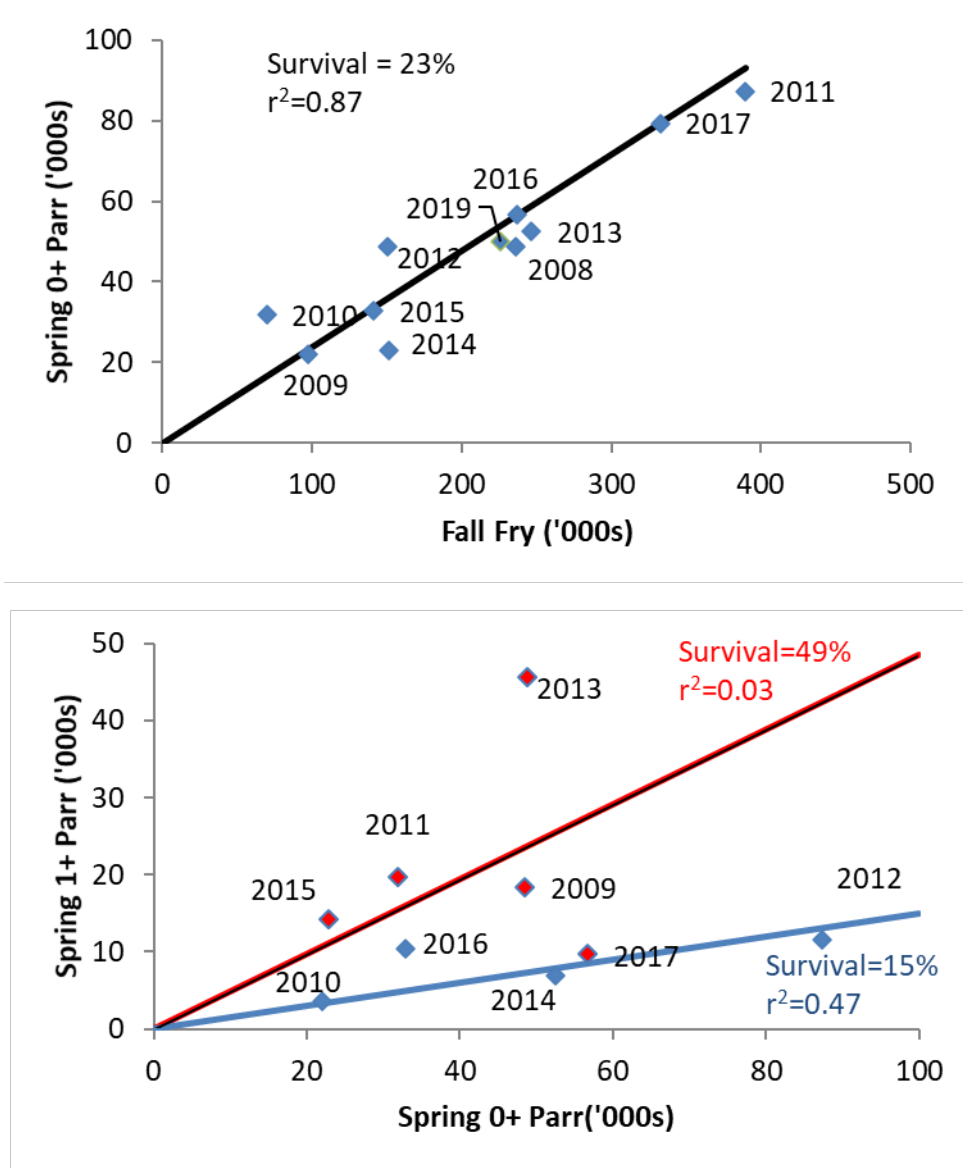


Figure 6.3. Relationship between Steelhead fall fry abundance and 0+ parr abundance the following spring (top), and between 0+ parr abundance in the spring and 1+ abundance the following spring (bottom). Separate relationships in even and odd years are used to highlight differences in survival in odd and even years. Labels beside each point in the top panel denote the brood year (year fall fry abundance was estimated). Labels beside each point in the bottom panel denote the year age-0+ abundance was determined (brood year + 1).

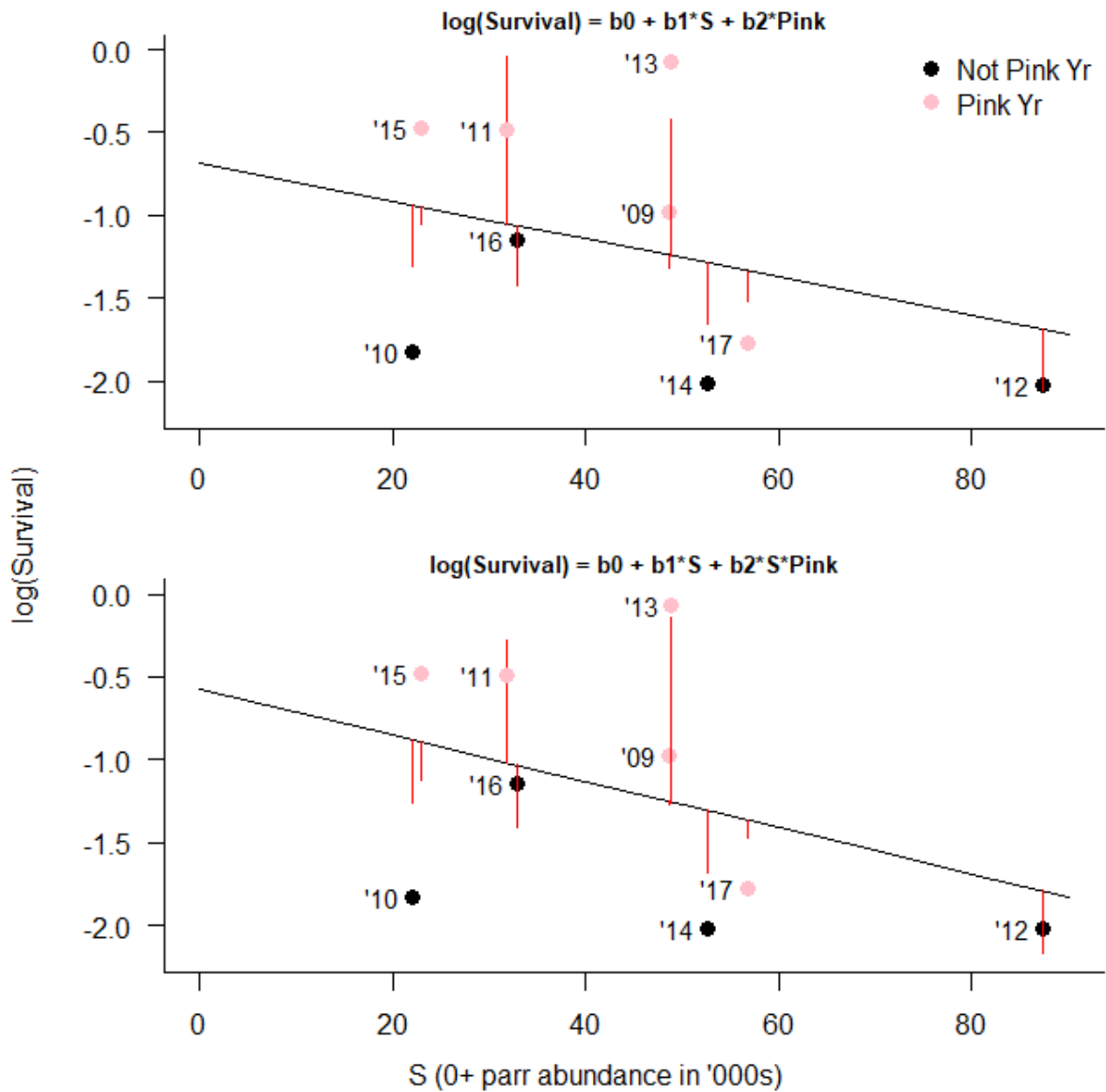


Figure 6.4. Relationships between the log of annual parr survival rates (0+ spring to 1+ spring) and 0+ parr abundance in the Cheakamus River. The labels beside each point denote the year of 0+ parr abundance estimates with pink points denoting odd years when pink spawners return to the Cheakamus River and black points denoting even years when they do not return. The black lines show predicted survival as a function of 0+ parr abundance and the mean pink abundance across years (top plot) or the production of 0+ parr abundance and pink abundance (bottom plot. Note that pink abundance is indexed by fry abundance estimates from the Rotary Screw trap in the following calendar year). The vertical red lines show the annual predictions that depend on annual estimates of 0+ parr abundance and pink fry abundance (top) or 0+parr*Pink fry abundance (bottom). [HBM/SurvPink_Regression.R].

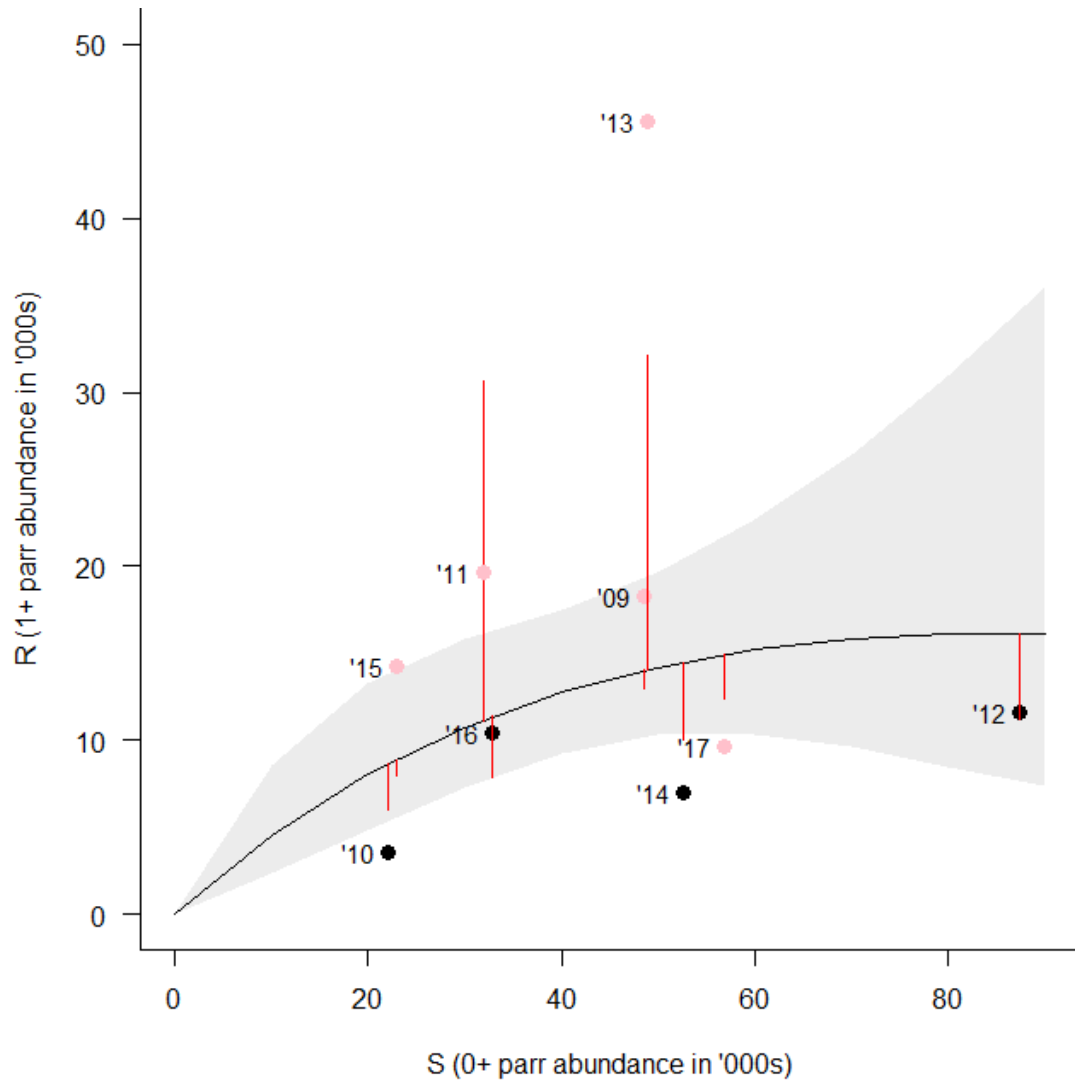


Figure 6.5. Relationship between abundance of 0+ Steelhead parr in the spring and abundance of 1+ parr one year later. Predictions are based on the model that uses 0+ parr abundance and pink fry abundance as independent variables ($R=S*\exp(b_0+b_1*S+b_2*Pink)$). The black line is the predicted parr abundance at the average pink fry abundance, and red vertical lines are the year-specific predicted abundances based on the year-specific fry abundances. The grey area represents the 95% confidence interval for the stock-recruit relationship at the average pink abundance.

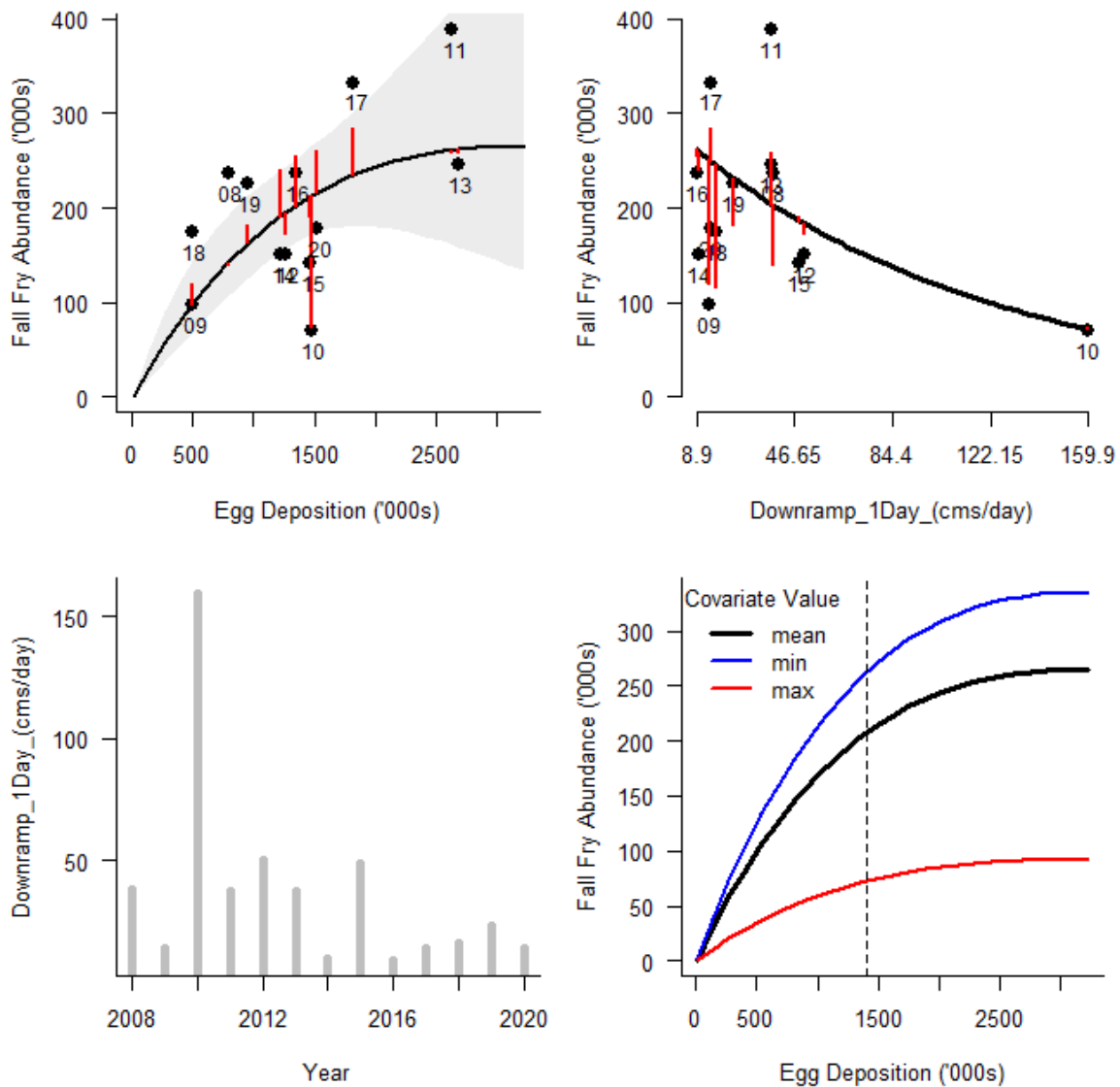


Figure 6.6. Fit of a Ricker flow covariate model to Steelhead egg deposition – fall fry abundance in the Cheakamus River. This model shifts the stock-recruitment curve each year based on the maximum decrease in discharge over 1 day (downramp rate). The upper-left panel shows the data (points) with text denoting year. The average stock-recruit relationship at the mean covariate value is shown by the black line, and predicted deviations for each year which depend on year-specific flow covariate values, are shown by the vertical red lines. The shaded grey area shows the 95% credible interval at the mean covariate value. The top-right panel shows the relationship between the covariate and fry abundance at the average egg deposition over years (black line) and predicted deviations at each annual level of egg deposition (vertical red lines). Also shown are the annual covariate values (bottom left) and the effect of the covariate on the recruitment curve based on the minimum, mean, and maximum covariate values across years (bottom right). For reference, the average egg deposition is shown by the vertical dashed line.

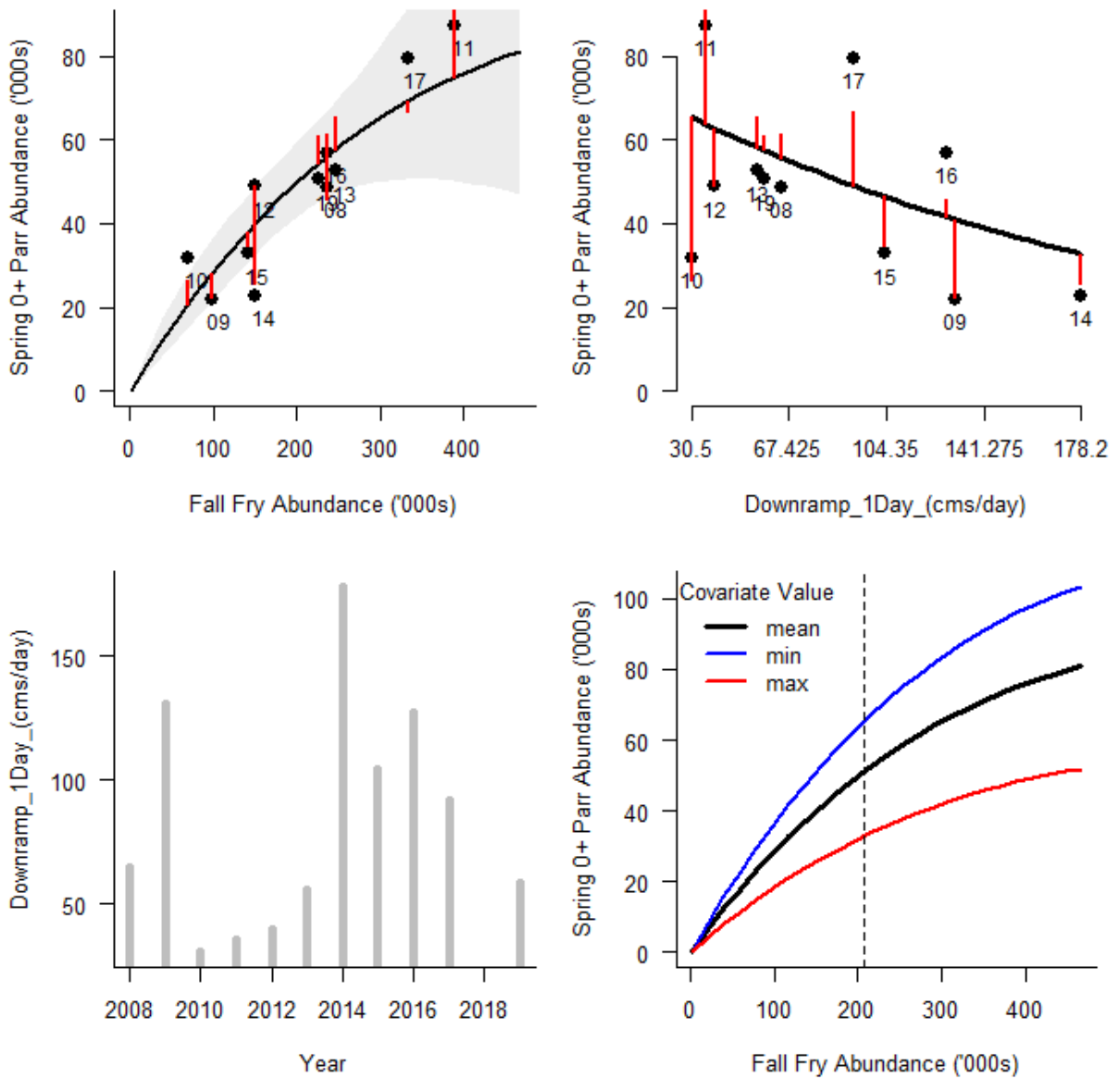


Figure 6.7. Fit of a Ricker flow covariate model to Steelhead fall fry – spring age 0 parr abundance in the Cheakamus River. See caption for Figure 6.6 for additional details.

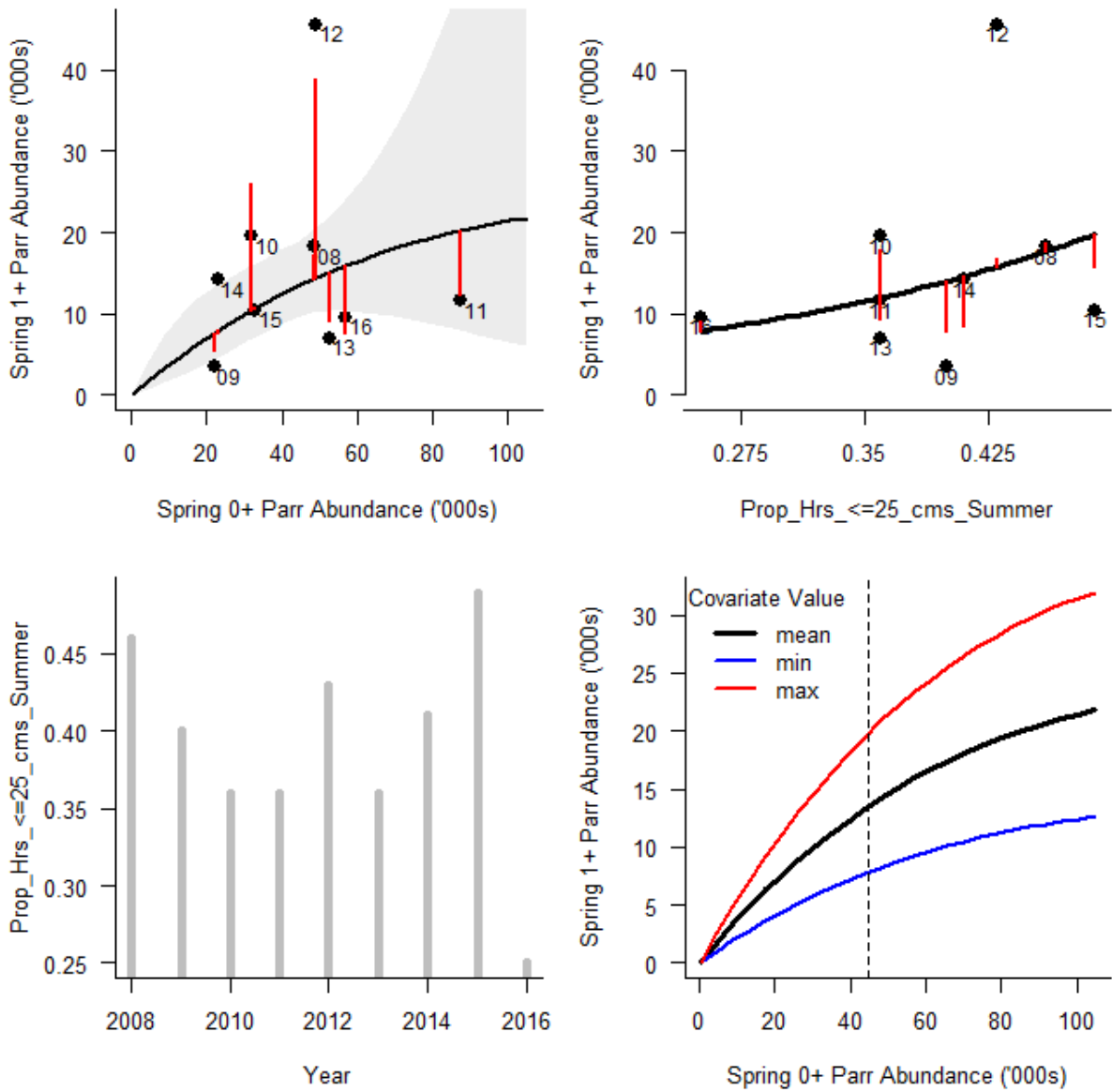


Figure 6.8. Fit of a Ricker flow covariate model to Steelhead spring age 0 – spring age 1 parr abundance in the Cheakamus River. See caption for Figure 6.6 for additional details. Predicted values are made at the average pink fry abundance.

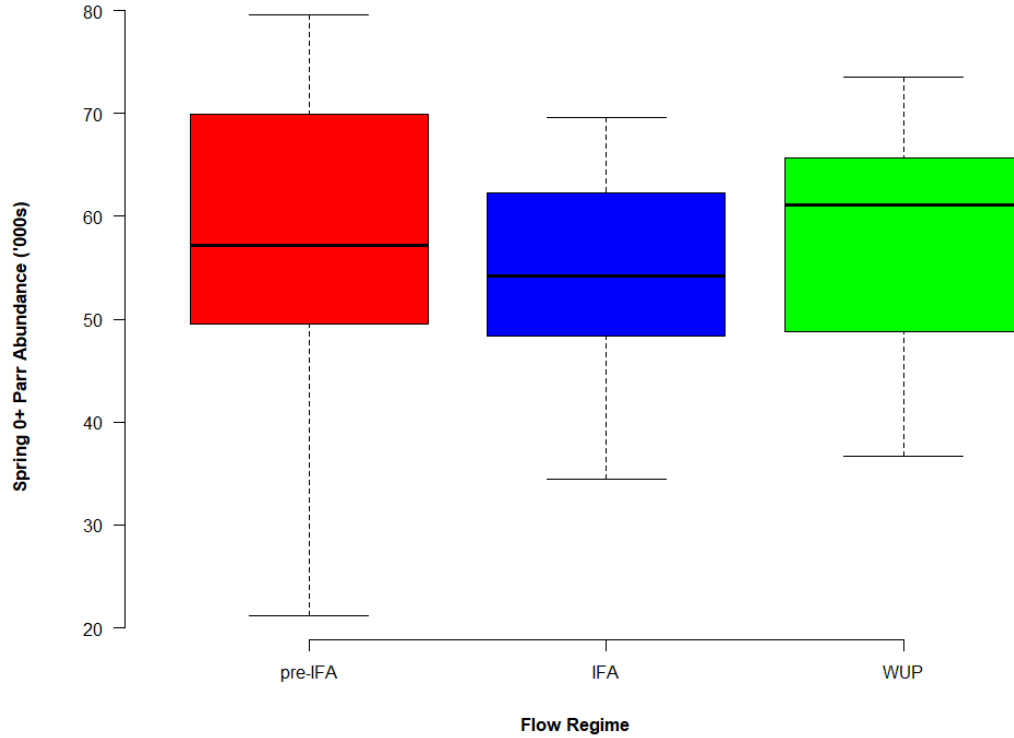


Figure 6.9. Box plots showing the distribution of predicted spring 0+ parr abundance across years from the integrated model applied to historical daily discharge during pre-Instream Flow Agreement (pre-IFA), IFA, and WUP flow regimes. The center line in each box plot shows the median and the box shows the central 50% of the distribution as defined by the 25th (Q1) and 75th (Q3) percentiles (the interquartile range, IQR). Error bars show $Q1 - 1.5 \cdot IQR$ and $Q3 + 1.5 \cdot IQR$.

Do WUP Flows Limit Steelhead Production in the Cheakamus River Relative to Production under IFA?

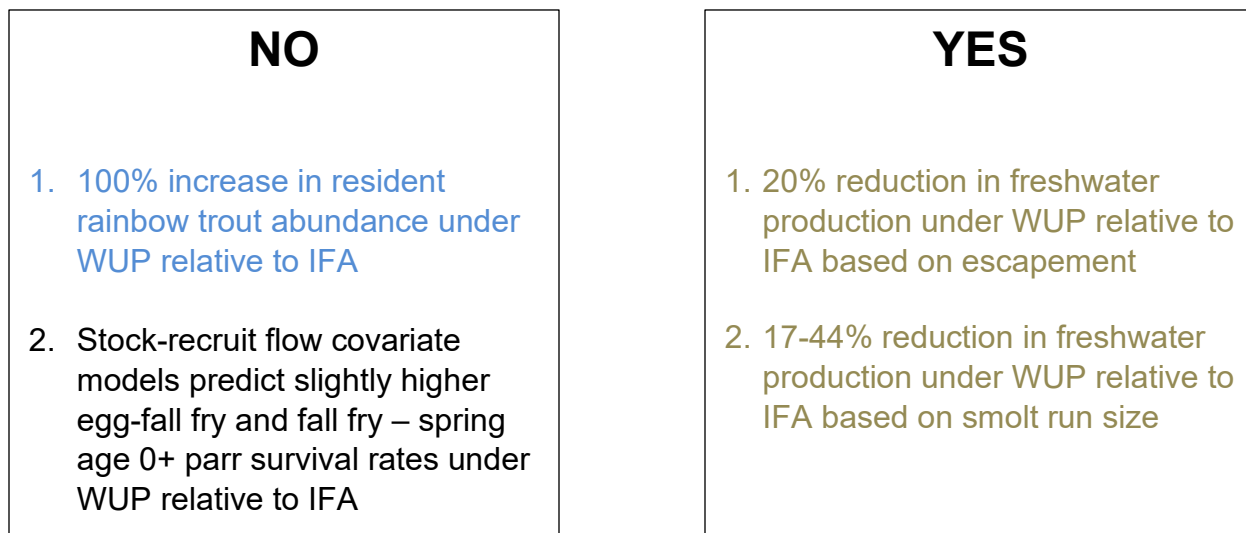


Figure 7.1. Weight-of-evidence that the WUP flow regime is limiting Steelhead production in the Cheakamus River. Lines of evidence indicating the answer is ‘YES’ or ‘NO’ are presented in the respective text boxes. Text colour indicates uncertainty level: brown, black and blue text indicates that statements that are *highly uncertain*, *moderately uncertain*, and *slightly uncertain*, respectively.