Cheakamus Project Water Use Plan

Project: Final Report of Adult and Juvenile Data to Evaluate Effects of the WUP Flow Regime on Steelhead in the Cheakamus River

Implementation Year 10

Reference: CMSMON-03

Study Period: Fall 2008 – Fall 2017

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May 4th, 2018

Executive Summary

We provide a synthesis of Steelhead information from the Cheakamus River to address three questions about the effects of flow and dam operations on Steelhead production identified during the 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 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 escapement (adult returns or spawners), juvenile abundance and survival rates, and smolt production. Escapement estimates are available from 1996 to 2017 and are based on repeat swim count data combined with radio telemetry information. Juvenile abundance estimates for various life stages (fall fry, age 0+, 1+, and 2+ parr in spring) are available from fall 2008 to fall 2017 based on electrofishing and snorkeling surveys. Survival rates between juvenile life stages are computed from abundance estimates. Abundance and survival rates from these programs are likely unbiased and are quite precise. Steelhead smolt production estimates from a Rotary Screw Trap (RST) program, available from 2001-2017, have variable precision.

Steelhead escapement was low (average 170) prior to the Interim Flow Agreement (the pre-IFA period as characterized by returns from 1996-2001). The average escapement produced under IFA flows was twice as high (386, escapement from 2002-2007) as the average for the pre-IFA period and this difference was statistically significant (p=0.002). The average escapement since 2010, which was produced from juveniles which reared in the river under WUP flows, was 1.6-fold higher (618) than during the IFA period and the difference was statistically significant (p=0.004). Escapement trends are affected by changes in both freshwater and marine survival rates and differences in escapement between pre-IFA, IFA, and WUP periods are therefore not solely caused by changes in flow regime. Based on information from other Steelhead rivers in southern BC and Washington, marine survival was 1.4- to 2-fold higher in WUP years than IFA years. The ratio of WUP to IFA average escapements, corrected for this

difference in marine survival, was 0.8-1.2, indicating that there has either been a modest decrease or increase in freshwater production under the WUP flow regime. Steelhead juvenile data from the Cheakamus River showed that annual survival rates of parr were more than 4-fold higher in odd years with elevated returns of pink salmon. These high returns only occurred in WUP years and were caused by broad-scale changes in marine survival. Removing the beneficial effect of pink salmon returns during WUP years led to a WUP/IFA adjusted escapement ratio of ~0.4-0.55. This result suggests that freshwater production of Steelhead in the Cheakamus River declined by 45-60% under the WUP flow regime.

Survival between age-0+ fry in the fall and age-0+ parr in spring, which quantifies their overwinter survival rate, averaged about 25% across years, was very consistent across years, and was not effected by fall fry abundance. These patterns indicate that that survival over winter was not density-dependent or effected by considerable variation in the magnitude and frequency of peak flow events across years. We speculate that overwintering behavior of this juvenile stage makes them largely invulnerable to flow effects. Annual survival rates of parr were four-fold higher in odd years compared to even years. This difference occurred because Steelhead parr consume pink salmon eggs which are a rich lipid source but are only available in odd years. There was no evidence of density-dependence in survival rates and they were relatively constant within odd and even years. Limited sample size, strong pink salmon effects, and the inability to estimate seasonal survival rates, make it challenging to evaluate effects of flow on parr survival.

There was density-dependence in Steelhead egg- fall fry survival rates and negative effects of high flow and rapid change in flow during summer months. Models that included effects of rapid changes in discharge from July through mid-September explained ~75% of the variation in the log of fall fry abundance. Models that included effects of high discharge in August (proportion of time > 60 or 80 $m^3 \cdot s^{-1}$) explained about 60% of the variation in log fry abundance. These flow covariate models provided a substantial increase in predictive ability relative to a model that only included density-dependent effects, which explained only 25% of the variation in log fry abundance. Egg-fry survival rates declined with higher levels of discharge and large and rapid changes in discharge. The latter model predicted that egg-fry survival rates in IFA years would only have been about 12% higher than in WUP years. This occurred because ramping rates under the IFA regime were lower and largely determined by natural rates of inflow. This predicted change in survival rates was not large enough to explain the ~ 50% decline in freshwater production under the WUP estimated from the analysis of escapement data. This indicates the models are either under-predicting the positive effects of IFA flows, or the escapement analysis over-predicted negative WUP effects. Stock-recruitment flow covariate models should be considered preliminary owing to limited sample size and partial confounding among different flow covariates.

Reductions in ramping rates specified in the current WUP flow order should be considered during the WUP order review. Ramping rates specified in the WUP flow order are 4- to 7-fold higher than the 2.5 $cm \cdot hr^{-1}$ guidelines from Fisheries and Oceans Canada. Models developed in this study predict that implementing ramping rates close to this guideline during July and August could result in substantive increases in Steelhead egg-fry survival rates. Our results also indicate that rafting flows of ~ 40 $m^3 \cdot s^{-1}$ are too low to have negative effects on Steelhead egg-fry survival rates but that flows higher than ~ 60 $m^3 \cdot s^{-1}$ in August should be avoided. Decision-makers should take into account that our predictions of negative effects of high flow and rapid flow change on egg-fry survival rates, while consistent with the literature, are still uncertain. Continued monitoring of Steelhead egg-fry survival rates is therefore warranted.

Acknowledgements

This project was supported through a contract from BC Hydro to Ecometric Research to provide data for the Cheakamus River Water Use Plan. Thanks to Mike Stamford, Jeff Sneep, and Jason Macnair for assisting in the fieldwork. Thanks to LJ Wilson, Cole Martin, Jennifer Buchanan, Stephanie Lingard and Caroline Melville at Instream Fisheries Research for conducting radio telemetry, scale-age analysis, and providing information from the Rotary Screw Trap program. Thanks to Mark Sherrington and Alexis Hall at BC Hydro for providing administrative support. We are very grateful for the considerable efforts of some dedicated anglers who captured resident rainbow trout and steelhead for radio tagging and scale collection.

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1.0 General Introduction

The Cheakamus River is a productive tributary of the Squamish River that supports populations of Steelhead (*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 (Fig. 1.1). Only the lower 17.5 kilometers of this river are accessible to anadromous salmon and Steelhead. 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 fish populations.

The Cheakamus River supports a relatively productive wild winter-run Steelhead population and a well-known Steelhead fishery. Although adult Steelhead returns are likely much smaller today, 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 other juvenile salmonids 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 Instream Flow Agreement (IFA) regime, which mimicked natural inflow patterns, argued that both seasonal and daily elements of the hydrograph could be important to juvenile salmonid production and that higher flows 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). Much of the debate focused on Steelhead, which is a highly valued species in the watershed and hypothesized to be more susceptible to flows than other salmonids because of their longer freshwater rearing period.

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 higher flows during and shortly after the Steelhead fry emergence period (July and August), that 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 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 currently supports 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 enumerates 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 provides estimates of the abundance of returning adult Steelhead spawners (escapement), abundance of various juveniles life stages rearing in the river, and survival rates among life stages. The central objectives of these programs are 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 consistently 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 large 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 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 will be 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 age-0 Steelhead, but this mortality may not affect subsequent age-1+ 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, and provide recommendations on future monitoring to improve our understanding of how flow effects steelhead production in the Cheakamus River. The objectives of this report are to clarify relationships between flow and Steelhead production in the Cheakamus River based on available data, to determine if the data are sufficient to address critical uncertainties regarding steelhead-flow relationships identified during the WUP, and to determine if monitoring programs are on-track and will be able to address these uncertainties in the future.

This 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 Instream Flow Agreement and WUP regimes. Chapter three provides a brief summary of the methods used to estimate abundance and survival rates, and analytical methods used in this report. Chapter four summarizes basic 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 is the only information 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. Chapter 7 provides final conclusions and recommendations.

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 Instream Flow Agreement (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). We also analyze flow records provided by BC Hydro on discharge from Daisy Lake Dam, turbines flows, and back-calculated inflow to Daisy Lake.

Discharge in the Cheakamus River is characterized 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, which affects flow downstream of the dam, have varied considerably since impoundment (Fig. 2.2). From 1958-1994, diversions were largely driven by power generation within the constraints of the original water license. However historical operations did not always follow the original water license which specified that a minimum of 45% of inflows to Daisy Lake Reservoir be released into the Cheakamus River from Daisy Lake Dam, with the remaining 55% 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). The IFA specified that the greater of 5 $m^3 \cdot s^{-1}$ or 45% of the previous seven days average inflow be released downstream (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 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.

As many of the historical and current operating rules focus on minimum flows, the effect of operations on flow in the Cheakamus River are greatest during winter when inflows are lowest (when the diversion is a greater proportion of the inflow). 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 local inflows, which often exceed the storage capabilities of the reservoir and the capacity of the tunnels (~65 $m^3 \cdot s^{-1}$) 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 and when diversions were reduced (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 rate of flow increases and decreases in August was substantially less 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 as releases from Daisy Lake Dam were determined by the previous weeks` inflow. As a result, rates of increase and decrease were less than under the WUP regime, where flow variation is only limited by ramping rates.

Ramping rates controlling the rate of change in discharge from Daisy Lake Dam specified in the WUP (Table 2.1) greatly 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 $m^3 \cdot s^{-1}$ to 60 $m^3 \cdot s^{-1}$. 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 $m^3 \cdot s^{-1} \cdot hr^{-1}$ (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 $cm \cdot hr^{-1}$ at the Brackendale gauge site and the pedestrian bridge, respectively. These ramp rates are about 4-fold greater than the FOC 2.5 $cm \cdot hr^{-1}$ 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 $m^3 \cdot s^{-1}$ to 72 $m^3 \cdot s^{-1}$ in one hour and to 56 $m^3 \cdot s^{-1}$ in two hours. This is equivalent to a stage change of about 15 $cm \cdot hr^{-1}$. Another common flow reduction occurs when flows at the Brackendale gauge site in mid-August are reduced from minimum rafting levels of 38 $m^3 \cdot s^{-1}$ to the seasonal minimum flow requirement of 20 $m^3 \cdot s^{-1}$ (Table 2.1).This flow change would occur in about 1.5 hrs. given a downramp rate of 13 $m^3 \cdot s^{-1} \cdot hr^{-1}$ (Table 2.1), which would be the most likely scenario given that releases from Daisy Lake Dam are greater than 10 $m^3 \cdot s^{-1}$. This rate of flow change translates to a stage change rate of approximately 17 and 19 $cm \cdot hr^{-1}$, 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. For example, on August, 15, 2013 the switch from rafting to minimum flows occurred in about four hours, considerably longer than minimum time of 1.5 hours needed to meet allowable ramping rates. This resulted in an actual ramp rate of about 7 $cm \cdot hr^{-1}$. While lower than the 17-19 $cm \cdot hr^{-1}$ rate, it was still about 3-fold greater than the FOC guideline.

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 used in this synthesis report (Chapters 5 and 6).

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 records the number of steelhead, resident rainbow trout, and bull trout that are observed. These surveys have been conducted over 21 years between 1996 and 2017 (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 after 2000 increased and has averaged 12 swims/year. Since 2000, swims are typically conducted on a weekly or bi-weekly basis from early March through early May. High flow conditions during 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 fish 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. abundance = count/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 provide 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).

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 radio tracks 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 swims 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 habtiats 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.

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 can be computed.

3.3 Smolt Production

The abundance of steelhead, Chinook, and coho smolts, as well as 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 again move downstream, mixing with the unmarked fish as they move downstream, and some (along with unmarked fish) are captured at the same location as before. The recapture of the marked fish provides information on the capture efficiency of the RST(s) which is then used to expand the number of unmarked fish captured, to estimate the population of fish passing the 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 (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 21 years through 2017 (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 years recruitment 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 summer- and 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). 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 odd years when pink salmon returns were high. Pink salmon returns were likely low during the IFA period but elevated during the WUP period in odd years only. 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 across both even and odd years during the WUP period.

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 underseeded. This would give the false impression that the IFA regime had a negative effect on freshwater production. The effects of underseeding 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 underseeding 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) $$R_t = E_{t+3}P_{t+3,3} + E_{t+4}P_{t+4,4} + E_{t+5}P_{t+5,5} + E_{t+6}P_{t+6,6},$$

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, 2013-2017) 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 (spawn more than once) species. 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 based the complete ageing dataset 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 a Beverton-Holt stock-recruit model to these data. The form of the Beverton-Holt model we fit was,

3.2) $$R = \frac{\alpha \cdot S}{1 + \frac{\alpha}{b} \cdot S} \cdot e^{\gamma \cdot X_t}$$

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 defined the pre-WUP stock-recruitment curve. As X is one in WUP brood years effected by WUP flows, γ shifts recruitment up (γ>0) or down (γ<0) by the same amount for any stock size in WUP years. Parameters were estimated by maximum likelihood in Excel using solver assuming that error in the log of predicted and observed recruitment was 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 were not available. Stock-recruitment models were fit using escapement and egg deposition as the measures of stock size (S in eqn. 3.2 = escapement or eggs). Productivity (α) was constrained to be 6 recruits/spawner or 2.6 recruits/1,000 eggs, respectively, to avoid unrealistically steep stock-recruitment curves owing to a paucity of very low escapement estimates (making it difficult to estimate the initial slope of the curve).

3.4.2. Emergence Timing

Determining the emergence timing of juvenile Steelhead 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-thermal unit 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 the 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 fertilization date. The spawn-timing curve for each year was then shifted 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.

We fit a Beverton-Holt stock-recruitment relationship to predicted fall fry abundance as a function of egg deposition that also accounts for potential flow effects using,

3.3) $$R_t = \frac{\alpha \cdot E_t}{1 + \frac{\alpha}{b} \cdot E_t} \cdot e^{\gamma \cdot X_t}$$

where R_t is the fall age-0+ (fry) abundance in year *t*, E is the estimated egg deposition in that year (in thousands of egg), α is the maximum fry per 1000 eggs (productivity), β is the maximum number of fall fry (capacity) , 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 evaluated use hourly discharge records at the WSC Brackendale gauge. These include measures of rapid discharge change (maximum discharge increase and decrease over 6 hours and 1 hour), average discharge, variation in discharge, and the proportion of time discharge was below 40, 50, 80, and 100 $m^3 \cdot s^{-1}$. These metrics were computed for the months of July, August, and the July-August period. Time intervals for the rapid discharge change covariates for August and the July-August periods extended through 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.

Parameter estimates for α, β and γ were obtained by Bayesian estimation by assuming that observations of log fry abundance were normally-distributed random variables with means predicted by eqn. 3.3. Models were fit using WinBUGS (Spiegelhalter et al. 1999). Uninformative priors were used for all model parameters (source code provided below). 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 Rhat values <1.01). See Table 3.2 for the source code.

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 required 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, 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 anadramous form or morph of *Oncorynchus mykiss*. The non-anadromous morph of *Oncorynchus 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, which in turn can be used to evaluate whether Brohm River can act as a control population for the Cheakamus River, 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 21 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 distributions (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 (Fig. 4.4).

The 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^{o}C. Spawn timing was generally similar among years, except in 2016 where it peaked two-three weeks earlier (Fig. 4.6 middle panel) when water temperatures by mid-April were noticeably higher in (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 data distribution (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. 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-15 to Aug-5, and last dates of emergence ranged from July-20 to Aug-13. These ranges indicate that July and August are potentially flow-sensitive months that will effect 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. Beginning in return year 2011, an average of 66% 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. 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 effect the relative survival rate of smaller (age 2 yr.) or larger (age 3 yr.) smolts.

Steelhead fry collected during our fall surveys in September had an average fork length of 47 mm between 2007 and 2017 (Table 4.2). They grew an average of 18 mm to reach a size of 65 mm by the spring (following April session). They grew an average of 34 mm over the summer to reach a size of about 100 mm by the next fall. Winter growth of these 1+ fish was on average 8 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 10 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 4.3). 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 due to only the larger 1+ parr smolting. 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.

Freshwater age as determined by scales collected from spawners since 2011 has been dominated by 3 yr smolts (66%). This conflicts with age frequency from the RST data, which indicates that only 44% of the smolts have been age 3 year 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 higher survival rates for age 3 yr smolts, which is feasible given their larger size at outmigration (Table 4.3).

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 = 29%, 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 15% 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 (76 mm) and after (78 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 based of returns to the Cheakamus River averaged 4.9 years, with an obvious shift from younger (mean age 4.5 yrs) to older fish (5.2 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 enter into the swim count data and be included in the resident trout estimate in 2006. 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 this change. 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 successfully radio-tagged a total 51 individuals based on captures in 2016 and 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.

Resident rainbow trout make a negligible contribution to the abundance of juvenile mykiss in the Cheakamus River that we sample during fall and spring juvenile surveys. The average escapement of Steelhead over 14 years when scale information has been collected was 521 fish, compared to 110 resident rainbow trout (Table 4.4). Of 112 resident trout that have been captured by angling, 47% have been females. There is no indication that the resident population in the Cheakamus River is dominated by males as in other systems (e.g. Thompson River). Owing to lower abundance and fecundity of residents compared to spawning Steelhead, they contributed on average only 4% of the total egg deposition for mykiss. Annual contributions have been as low as 1% (2009) and as high as 9% (2016).

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% (n=4) or 100% (n=16) in the reach between Culliton and Cheekye confluences, or downstream of the Cheekye confluence, respectively. 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 from residents to the juvenile population (~1%) compared to the otolith analysis. However samples from the otolith analysis were from a systematic upstream-downstream sampling program. 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.

4.6 Contribution of Brohm River

Brohm River is a tributary of the Cheekye River that flows into the Cheakamus River (Fig. 1.1). On average, 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 spawns 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 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 20% in the Cheakamus and 16% in Brohm (Table 4.5). These rates are similar and do not suggest that there is substantive loss of fish from Brohm River into the Cheakamus River over their first winter. Annual survival from age-0+ parr in the spring to age-1+ the following spring was 36% and 47% in Cheakamus and Brohm Rivers, respectively. Assuming that survival rates in Brohm River are not much larger 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 independent.

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

The historical escapement trend for Steelhead in the Cheakamus River can be used to make inferences about the effects of flow on freshwater production. 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 (386, escapement from 2002-2007) and this difference 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 ($p=0.063$). The average escapement since 2010, which was produced from juveniles which have reared in the river under WUP flows, was 1.6-fold higher (618) than during the IFA pre-spill period and this difference was statistically significant ($p=0.004$).

Greater Steelhead production in the Cheakamus River during the WUP period was also seen in the stock-recruitment analysis (Fig. 5.2). Brood years that reared in the Cheakamus River under WUP flows had recruitments more than double those of brood years 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 under WUP flows. More than 99% of the γ estimates from the posterior distribution were greater than zero, indicating a highly significant increase in recruitment for a given stock size under WUP flows. This model explained about 60% of the variation in log recruitment relative to 5% under a model which did not allow recruitment to very across regimes. Thus there is strong evidence for greater recruitment for a given stock size under the WUP regime. 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. Escapements during that period were lower than under the WUP regime but were still sufficient to result in full seeding as estimated by the stock-recruitment curves.

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, may not be caused by changes in flow. Our goal here is to use differences in Steelhead escapement in the Cheakamus River as effected by IFA and WUP flow regimes to make inferences of the effects of these regimes on freshwater survival rates. To do this we need to remove effects of marine survival and non-flow related effects on freshwater survival rates. 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 rivers in Puget Sound and the Keogh River. We took the average of these survival rates in outmigration years associated with freshwater production during IFA and WUP periods. The adult return year range for the IFA period was 2002-2007 (4 years after IFA flows were implemented in 1998) which corresponds to an outmigration year range of 2000-2005 assuming that the majority of returns spend two winters at sea (Fig. 4.8). The return year range for the WUP period was 2010-2017, corresponding to an outmigration year range of 2008-2015. The average Puget Sound-Keogh smolt-adult survival rates were 1.1% and 2.1% over these 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 almost-two fold under the WUP regime relative to the IFA regime. As a result, the 1.6-fold increase in escapement under WUP flows relative to IFA flows must be reduced by almost two-fold, resulting in a WUP/IFA marine survival-adjusted ratio of 0.82. This estimate suggests that freshwater production of Steelhead in the Cheakamus River dropped by almost 20% under the WUP flow regime. We repeated this calculation using smolt-adult survival rates from the Keogh River only. Average marine survival rates were 4.2% and 5.7% over the IFA and WUP outmigration year periods, respectively, resulting in a marine survival adjustment of 1.35 (Table 5.2). This reduced the WUP/IFA escapement ratio from 1.6-fold to ~1.2-fold. This correction suggests freshwater production increased by about 20% over the WUP period. A similar set of computations comparing pre-IFA and IFA periods resulted in an IFA/pre-IFA marine-survival adjusted escapement ratio of 3.2 and 3.5 based on Puget-Keogh and Keogh-only adjustments, respectively. These estimates indicated that freshwater production increased by more than 3-fold under the IFA flow regime relative to the pre-IFA regime.

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 odd years when pink salmon returned in large numbers, compared to even years when virtually no pink salmon returned (see Chapter 6). This pink salmon adjustment must be reduced by 50% to account for the fact that the higher survival due to pink salmon returns only occurs every second year, resulting in an average pink salmon-adjustment of just over two-fold (Table 5.2). 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 Rive has provided a very reliable index of the run size of outmigrating pink salmon fry since 2002 which presumably reflects in part 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. 5.3). 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. 5.2). 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. Marine survival- and pink salmon-corrected WUP/IFA Steelhead escapement ratios in the Cheakamus River were 0.38-fold and 0.55-fold using the smolt-adult survival rates for the Puget-Keogh and Keogh only rates, respectively (Table 5.2). These low ratios suggest that the WUP flow regime has reduced freshwater survival rates for juvenile steelhead by ~45-60% relative to production under the IFA regime.

The trend in abundance of resident rainbow trout generally followed the trend seen for steelhead (Fig. 5.4). 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 2.25-fold higher than under the IFA regime and this difference was also statistically significant (p=0.031). 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. 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 plan increased substantially in late 2009 owing to a change in their treatment process (Fig. 5.5). We speculate that this increase may have effected 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. 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

Reliable abundance estimates for juvenile life stages of Steelhead in the Cheakamus and Brohm Rivers are available beginning in fall of 2008. Although we 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 vast majority of mykiss juveniles are born of 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 was 200,000 (Fig. 6.1a). We quantify the abundance of age-0 parr the following spring through a combination of electrofishing and snorkel surveys. The abundance of this life stage across study years was ~45,000. 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 abundance due to poor capture probability. However, we can provide reliable indices of abundance of 1+ and 2+ parr in the spring, whose abundances averaged about 15,000, and 3,000 across study years, respectively. Abundance estimates are also available for Brohm River (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. Plots of the abundance of one life stage as a function of the abundance of the previous life stage can be used to determine if there is density-dependent mortality and can also quantify the average survival rate between life stages. The abundance of age-0+ parr in the spring increased linearly with the abundance of fry the previous fall (Fig. 6.3 top). This strongly linear relationship indicates that survival rate is constant as density increases. In other words, survival rate is not density-dependent. The slope of the relationship represents the average survival rate between fall fry and spring age-0+ parr, which was 0.25. Fall fry abundance explained 84% of the variation in spring age-0+ abundance, indicating little inter annual variation in the overwinter survival rate between these life stages. This was surprising as there was considerable inter annual variability in the number and intensity of fall and winter storm events (Chapter 2). Some of these events were large enough to have caused noticeable changes in river morphology which indicates substantial bedload movement. The strong relationship between fall fry abundance and age-0+ abundance the following spring indicates that the events are not impacting overwinter survival rates. It is likely that overwintering behavior (concealment in substrate) reduces their vulnerability to high flow events.

We also saw linear relationships between the abundance of age-0+ parr and age-1+ parr one year later (Fig. 6.3 bottom) indicating no density-dependence mortality between these life stages. For a given age-0+ abundance, age-1+ abundance the following year was on average four-fold higher in odd years than even years. This indicates that annual survival rates (slopes) were 0.15 and 0.64 in even and odd years respectively. Higher survival rates of parr in odd years was almost certainly 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 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. We speculate 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).

There was a saturating relationship between Steelhead egg deposition and abundance of fry in the fall which indicated considerable density-dependence (Fig. 6.4, top). A Beverton-Holt model was fit to the data and explained 25% of the inter annual variation in log-recruitment. As there was no evidence for density-dependence in subsequent life stages (Fig. 6.3), a saturating relationship between egg deposition and age-1+ parr abundance in spring was similar to the one for egg-fall fry (Fig. 6.4). We fit the egg-parr relationship using an odd-even year offset (eqn. 3.2) to account for the much higher age-0+ to age-1+ parr survival rate in odd years (which are even brood years). This model explained 49% of the variation in the log of age-1+ parr abundance. The e^{γ} offset estimate was 3.6, which indicates that parr production was 3.6-fold higher for even year broods than odd year broods after accounting for effects of density-dependence.

We evaluated effects of flow in the Cheakamus River on juvenile Steelhead egg-fry survival rates using a stock-recruitment model that included a flow covariate effect. We only conducted this analysis for the egg-fall fry stage. There was no evidence of flow effects seen in overwinter survival rates for fry (fall fry to spring age-0+) as these survival rates were very consistent across years with very different peak flows. Evaluating flow effects for annual parr survival rates is challenging because the survival rate is affected by conditions over the entire year, making it difficult to evaluate specific flow hypotheses (e.g. low winter flows, high summer flows). In addition, any analysis of flow effects would have to be conducted separately for even and odd brood years owing to the strong pink salmon effect. This reduces the effect sample size down to 3 or 4 years which is too low to tease out potential flow effects.

The egg – fall fry stock-recruitment analysis examines the effects of a variety of flow metrics during the post-emergence stage (see section 4.2). We examined 11 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 six consecutive hours and over one hour. These four discharge change metrics were standardized so statistics show the rate of change in discharge over 1 hour ($m^3 \cdot s^{-1} \cdot hr^{-1}$). Other annual statistics include the average flow (Avg_Q), the standard deviation in flow (SD_Q), the maximum flow (Max_Q) and the proportion of hours with flows less than 40, 60, 80, and 100 $m^3 \cdot s^{-1}$ (Prop_Hrs<x). All 11 flow covariate statistics were computed using data in the months of July, August, and both months combined. The period over which discharge change statistics was computed was extended up the first date of sampling for fall fry in September for August only and July-August periods to ensure to

capture any rapid changes in flow that occurred up to the time when we estimate fry abundance. We also fit a model without any covariate effect as a baseline to judge potential improvements in predictions by including flow covariates. We fit a total of 34 models and compared them based on differences in the proportion of variance in log fry abundance explained and by differences in the Deviance Information Criterion (DIC).

Models that quantified rapid discharge change during the July-August period provided the best fits to the data and explained 73-77% of the variation in log fry abundance (Table 6.2, models 16, 24, and 27). These three models had DIC values that were separated by no more than about 2 units, indicating that they all had similar predictive abilities. Discharge change models using data from August only also provided very good fits to the data (models 13 and 14) and explained 67-71% of the variation in log fry abundance. Models based on the proportion of hours < 60 or 80 $m^3 \cdot s^{-1}$ in August (models 21 and 22) also fit the data well and explained a bit more than 60% of the variation in log fry abundance. These models had slightly higher DIC values than the rapid discharge change covariates and therefore had slightly weaker predictive ability. All these models provided a substantial increase in predictive ability relative to the model without a flow covariate effect (model 1), which explained only 25% of the variation in log fry abundance.

In the case of models that predicted fry abundance as a function of rapid discharge increases or decreases, the mean γ estimate was always negative, indicating that egg-fry survival rates decrease with increases in the magnitude of rapid discharge change (Fig.'s 6.5 and 6.6). The effect of rapid discharge change on egg-fry survival rates was large and predicted 2.5- to 3-fold increases in fry abundance across the range of observed rapid discharge changes over years (upper-right panels in Fig.'s 6.5 and 6.6). There was less than a 1% probability that $\gamma>0$ for almost all the rapid discharge change models (13, 14, 16, 24, 25, and 27) implying a very high probability (>99%) that greater rapid discharge change decreases egg-fry survival rates.

After correcting for marine survival and pink salmon effects on escapement, we estimated that freshwater production of juvenile steelhead under WUP flows declined by 45-60% relative to the production under IFA flows (Table 5.2). We evaluated whether our egg-fry rapid discharge covariate models could explain some of this decline. We computed the maximum discharge change over July and August (and extending to the average date of first sampling in September) for each year beginning in 1996 (Fig. 6.7). We then averaged these statistics across years representing the IFA (1998-2005) and WUP periods (2006-2012). We excluded years after 2012 from the WUP average since the escapement produced from fish that spawned in 2013 and later years is not yet known (given an average age at return in recent years of 5 years old). Differences in discharge change between IFA and WUP periods were modest (horizontal lines in Fig. 6.7). We calculated the average change in stock-recruitment curves using models 24 and 27 (Table 6.2) under IFA and WUP periods. These models predicted that egg-fry survival rates were 4% and 8% lower under the IFA regime due to rapid increases and decreases in discharge, respectively. The combined effect (12%) was well below the 45-60% decline in freshwater production estimated from the escapement analysis. This indicates we have either overestimated the impact of the WUP flow regime based on our analysis of the escapement data, have underestimated the effects of rapid changes in discharge on egg-fry survival rates, or there is another flow effect that we have not accounted for. Nevertheless, rapid flow change models predict that reduced ramping rates at Daisy Lake Dam has the potential to substantively improve Steelhead egg-fry survival rates. For example, reducing the maximum rapid flow increase from the maximum observed level between 2008 and 2017 (~23 $m^3 \cdot s^{-1} \cdot hr^{-1}$) to the minimum (~2 $m^3 \cdot s^{-1} \cdot hr^{-1}$) would lead to a 2.5-fold increase in egg-fry survival rates (Fig. 6.5 lower-left panel).

7.0 Conclusions and Recommendations for Future Monitoring

We estimated that freshwater production of Steelhead in the Cheakamus River under the WUP regime was about ~45-60% of production under the IFA regime based on an analysis of a 21-year record of escapement. The average Steelhead escapement produced from juveniles that reared in the river under WUP flows was actually 1.6-fold higher than the average escapement produced from juveniles that reared under IFA flows. However marine survival was higher during the WUP period, and large pink salmon returns during the WUP period, which were not caused by WUP flows, increased survival rates of Steelhead parr by more than two-fold. After accounting for these confounding effects, the escapement data indicate that freshwater production dropped substantively under the WUP regime. Models based on juvenile data did not find any evidence for flow effects on parr stages but did indicate that egg-fry survival rates are reduced when flows or rapid flow changes during July and August are high. The rapid flow change models predicted that egg-fry survival rates would have only been 12% lower under the WUP flow regime relative to the IFA regime. This difference was not great enough to explain the ~45-60% decline in freshwater production estimated from the escapement data. Thus, we have either overestimated the negative effect of the WUP regime on Steelhead from the escapement analysis, the analysis of juvenile data has underestimated the negative effects of ramping rates under WUP flows, or there is another effect of WUP flows that is lowering production. Nevertheless, both escapement and juvenile analyses indicate that the WUP regime has reduced survival rates of juvenile Steelhead, and the flow covariate models can be used to define lower ramping rates that could increase Steelhead egg-fry survival rates.

Our egg-fry flow covariate models indicate that rapid increases or decreases in discharge between July and early September, or higher discharges in August, has a negative effect on egg-fry survival rates. 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. Stock-recruitment relationships based on only ten years of data are uncertain, especially when the relationship includes an extra parameter to model flow effects. We simply do not have enough observations at high discharge or high rapid discharge change to quantify this effect more precisely. In addition, partial confounding between the magnitude of rapid flow change and the magnitude of discharge makes it difficult to separate these effects. The former can be completely controlled by ramping rates. The latter is largely determined by snow pack and air temperatures, but can be partially controlled by avoiding maintenance operations in July through early September. Our finding that rapid flow change and high flows during the post-emergent period result in substantive declines in egg-fry survival rates should therefore be considered preliminary and additional data collection is warranted if there is a desire to reduce this uncertainty.

Our prediction that high discharge or rapid changes in discharge reduce fry survival rates during the emergence and post-emergent periods is well-supported in the literature. 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 larger rivers like the Cheakamus, these microhabitats may be limited as they are only found in the immediate nearshore areas at river margins (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. 2008Young et al. 2011, Nagrodski et al. 2012, Gibeau et al. 2016). High flows can result in microhabitat velocities that exceed the limited swimming capacity for small post-emergent fry and can cause catastrophic displacement (Nislow and Armstrong 2012). Due to these factors a number of studies have shown that emergence and post-emergence periods 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 post-emergent fry stage can have negative effects on survival and growth. Maximum ramping rates at Daisy Lake Dam increase with discharge and are very high (Table 2.1). The recommended ramping rate from Fisheries and Oceans Canada is 2.5 $cm \cdot hr^{-1}$. Rates of change in the Cheakamus River specified in the WUP flow order are 3-

to 7-fold higher than this FOC guideline. Thus our prediction that the observed rapid changes in flow in the Cheakamus River due to operations at Daisy Lake Dam reduce Steelhead egg-fry survival rates should not be surprising given how fast they occur relative to recommended rates.

We saw very limited variation in overwinter and annual survival rates for Steelhead parr in the Cheakamus River, aside from a very strong effect of pink salmon returns. The larger size and mobility of parr relative to fry gives them 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 (see review in Nislow and Armstrong 2012). Parr spend more time hiding and less time foraging than fry. Shelter availability (interstitial pore space) has been shown to effect growth and survival rates of parr, thus the primary impact of flow regimes on parr may be through its effects on streambed composition. In the Cheakamus River, natural high inflow events are sufficient to regularly mobilize the bed (KWL 2014) and maintain sufficient interstitial space for Steelhead parr. 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. This may explain why our study showed little interannual variation in survival rates for parr life stages except for the food-mediated effect of pink salmon in odd years. The lack of density-dependence in parr stages indicates that higher mortality at the fry stage due to high flows or rapid discharge changes will not be compensated by lower mortality due to reduced densities in later life stages, and this was confirmed by the egg deposition-parr stock-recruit relationship. However, the sample size is limited, so our conclusions that there is no density-dependence in the parr stage, and no effect of flow, should be considered preliminary. We recommend continuing with annual estimates of parr abundance in spring to reduce this uncertainty.

Our estimated flow effects on post-emergent Steelhead fry in the Cheakamus River are also informative with respect to concerns raised during the WUP planning process that flows of ~40 $m^3 \cdot s^{-1}$ 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 flows >

60 or 80 $m^3 \cdot s^{-1}$ potentially reduce post-emergent survival rates, but we saw no effect for flows < 40 $m^3 \cdot s^{-1}$. These results indicate that there is currently no evidence that egg-fry survival rates are lower at 40 $m^3 \cdot s^{-1}$ than at 20 $m^3 \cdot s^{-1}$. Thus there is no evidence for a conflict between rafting flows and survival of Steelhead fry, and therefore no apparent risk of flows of 40 $m^3 \cdot s^{-1}$ being maintained through August if inflows are sufficient.

Overall, our results suggest that reductions in ramping rates specified in the current WUP flow order should be considered. Our egg-fry flow covariate model could be used to define new rates. For example, egg-fry survival rates increase rapidly at the low end of rapid discharge change range observed during the 2008-2017 assessment period (Fig.'s 6.5 and 6.6). A ramping rate of 1.5 $m^3 \cdot s^{-1} \cdot hr^{-1}$ meets the 2.5 $cm \cdot hr^{-1}$ FOC guideline at the Brackendale gauge at a discharge of 20 $m^3 \cdot s^{-1}$, and maintaining this ramping rate results in slower stage changes as discharge increases owing to the change in the shape of the channel cross-section (Fig. 6.8). Another alternative is to allow ramping rates to vary with discharge to maintain the FOC guideline, and in this case the ramping rate increases with discharge (Fig. 6.9). For example, the ramping rate at Daisy Lake Dam would need to be 1.3 $m^3 \cdot s^{-1} \cdot hr^{-1}$ if flows were 15 $m^3 \cdot s^{-1}$ at the Brackendale gauge, but would increase to 4.1 $m^3 \cdot s^{-1} \cdot hr^{-1}$ at a discharge of 100 $m^3 \cdot s^{-1}$ at Brackendale. These rates would typically be more than an order of magnitude lower than the FOC guideline rates, particularly the upramp rate.

Slower ramping rates could have impacts to hydropower generation and reduce flexibility for maintenance, however the impact may be modest if they are limited to the July-August post-emergent rearing period. This would be justified for Steelhead, but it is likely that similar dynamics are occurring for post-emergence Chinook and coho, so extending ramping restrictions from early spring through fall should be considered. Our predictions of negative effects of rapid flow change on Steelhead egg-fry survival rates, while broadly consistent with the literature, are still uncertain. Continued monitoring of Steelhead egg-fry survival rates is warranted as it will improve the predictive abilities of the egg-fry flow covariate models, especially if purposeful rapid flow changes during summer are implemented on an experimental basis in the next few years.

Given the intensity of Steelhead monitoring in the Cheakamus River over the last decade, it may be disappointing that more definitive statements about the effects of flow and the WUP flow regime cannot be made. From a scientific view point this outcome is not at all surprising. Three key elements are required to understand effects of flow on fish populations: 1) unbiased and precise monitoring programs; 2) adequate replication (multiple years of data for a given flow treatment); and 3) an informative experimental design (two or more treatments). Escapement and juvenile Steelhead abundance and survival estimates for the Cheakamus River are about as unbiased and precise as we can expect given the size of the river and hydrologic conditions. The recent reduction in monitoring effort (beginning in spring of 2018) will result in reduced precision of escapement and juvenile abundance and survival estimates, and increase the probability of bias. the length of available juvenile monitoring data is limited (10 years), and to date informative flow contrasts have not been provided even though they were recommended during the initial Cheakamus WUP planning process (Marmorek and Parnell 2002). Power analyses focused on detecting changes in freshwater productivity from monitoring of escapement and juvenile abundance indicate that experiments lasting 4-6 generations are needed to provide relatively unambiguous results (Parnell et al. 2003, Bradford et al. 2005). In the case of Cheakamus River Steelhead, this implies that a 20-30 year experiment given a generation time of ~5 years, with ~10-15 years for each treatment. The current juvenile program only provides 10 years of data under one treatment. The escapement data provides information on three treatments but there are only five-six replicates to characterize the pre-IFA and IFA flow regimes. Thus additional effort is required to provide definitive answers about the effects of flow on Steelhead in the Cheakamus River. This finding should not be surprising because it was identified in the power analysis conducted at the end of the initial WUP planning phase (Parnell et al. 2003). The need for extended monitoring periods and Adaptive Management was also identified in the WUP guidelines that were in place at the very beginning of the WUP planning process on the Cheakamus River (PBC 1998).

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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 ($m^3 \cdot s^{-1}$)
November 1 -December 31	3
January 1 - March 31	5
April 1 - October 31	7
	Brackendale Gauge Discharge ($m^3 \cdot s^{-1}$)
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 per hour

Discharge from Daisy Lake Dam ($m^3 \cdot s^{-1}$)	Maximum Rate of Increase ($m^3 \cdot s^{-1} \cdot hr^{-1}$)[2]	Maximum Rate of Decrease ($m^3 \cdot s^{-1} \cdot hr^{-1}$)
<10	52	1
10-62	52	13
>62	78	78

[1]Unless directed by the Comptroller to maintain flows at 38 $m^3 \cdot s^{-1}$.

[2]Upramp rates in the WUP order are specified in $m^3 \cdot s^{-1} \cdot 10\ min^{-1}$, $m^3 \cdot s^{-1} \cdot 15\ min^{-1}$, and $m^3 \cdot s^{-1} \cdot 60\ min^{-1}$ but are presented on an hourly timestep in this table so that upramp and downramp rates are directly comparable (see Table c below) for exact reproduction of WUP ramp rate table .

Table 2.1. Con't.

c) Ramping rates as specified in the Cheakamus flow order

Discharge from Daisy Lake Dam ($m^3 \cdot s^{-1}$)	**Maximum Rate of Increase**	**Maximum Rate of Decrease**
<10	13 $m^3 \cdot s^{-1} \cdot 15\ min^{-1}$	1.0 $m^3 \cdot s^{-1} \cdot hr^{-1}$
10-62	13 $m^3 \cdot s^{-1} \cdot 15\ min^{-1}$	13 $m^3 \cdot s^{-1} \cdot hr^{-1}$
>62	13 $m^3 \cdot s^{-1} \cdot 10\ min^{-1}$	13 $m^3 \cdot s^{-1} \cdot 10\ min^{-1}$

d)

e) **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 $m^3 \cdot s^{-1} \cdot hr^{-1}$ 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 ($m^3 \cdot s^{-1}$)	**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 ($cm \cdot hr^{-1}$)		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 ($cm \cdot hr^{-1}$)		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])
	lgsdDep[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_lgsdDep<-sum(lgsdDep[])/Nyrs	#mean of lg SDs
for(iyr in 1:Nyrs){
	SSQ[iyr]<-pow(lgsdDep[iyr]-mu_lgsdDep,2)
}
sd_lgsdDep<-pow(sum(SSQ[])/(Nyrs-1),0.5)
```

Table 3.2. Source code for WinBUGS model estimating stock-recruitment parameters for Steelhead egg deposition-fall fry abundance flow covariate model.

```
#prior on productivity
alpha_log~dnorm(0,1.0E-03)
alpha<-exp(alpha_log)

#prior on capacity
beta_log~dnorm(bprior,1.0E02)
beta<-exp(beta_log)

#prior on precidion for likelihood
tau~dgamma(0.01,0.01)

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

#predict recruitment given stock-recruit parameters, egg deposition, and standardized
#covariate value X
for(i in 1:Nrecs){
	lgPred_Rec[i]<-log(alpha*E[i]/(1+alpha/beta*E[i])*exp(gamma*X[i]))

	#normal likelihood on log predicted and observed
	lgREC[i]~dnorm(lgPred_Rec[i],tau)
}
```

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. 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).

	Mean Fork Length (mm)					
Year	**0+ fall**	**0+ spring**	**1+ fall**	**1+ spring**	**2+fall**	**2+ spring**
2007	52		108		149	
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
Average	47	65	100	108	143	153

	Growth (mm)				
Year	**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
Average	18	34	8	34	10

Table 4.3. 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).

	Mean Fork Length (mm)		
Year	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 4.4. 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.

	Steelhead								**Resident Trout**		
Year	**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%	79	131	1.7		17	7	5%
2001	27	756	4,219	41%	324	556	1.7		22	9	2%
2003	33	801	5,016	52%	319	825	2.6		107	43	5%
2004	36	769	4,431	44%	347	684	2.0		75	30	4%
2005	38	776	4,552	50%	337	768	2.3		54	22	3%
2009	27	735	3,864	59%	221	507	2.3		18	7	1%
2010	57	691	3,206	44%	1,061	1,492	1.4		182	73	5%
2011	107	794	4,885	61%	899	2,666	3.0		67	27	1%
2012	9	836	5,733	56%	396	1,263	3.2		127	51	4%
2013	24	794	4,883	58%	949	2,702	2.8		173	69	2%
2014	80	766	4,391	51%	548	1,232	2.3		103	41	3%
2015	88	780	4,640	55%	583	1,476	2.5		125	50	3%
2016	31	748	4,068	65%	514	1,350	2.6		331	132	9%
2017	26	806	5,116	50%	716	1,831	2.6		142	57	3%
Avg.	43	768	4,452	52%	521	1,249	2.35		110	44	4%

Table 4.5. Average abundance between 2008 and 2017 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.

	Abundance			Survival Across Stages	
Life Stage	**Cheakamus**	**Brohm**	**% Brohm**	**Cheakamus**	**Brohm**
0+ fall	205.3	20.7	10%		
0+ spring	40.4	3.4	8%	20%	16%
1+ spring	14.5	1.6	11%	36%	47%

Table 5.1. Steelhead escapement estimates to the Cheakamus River, 1996-2017. 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).

	Wild		Hatchery		Wild+Hatchery	
Year	**Mean**	**CV**	**Mean**	**CV**	**Mean**	**CV**
1996	174	0.17				
1997	112	0.16				
1999	163	0.17				
2000	79	0.19				
2001	324	0.13				
2002	443	0.12				
2003	319	0.09				
2004	347	0.13				
2005	337	0.10				
2006	322	0.12				
2007	544	0.09				
2008	347	0.11				
2009	116	0.19	105	0.34	221	0.19
2010	633	0.09	428	0.17	1,061	0.09
2011	608	0.10	290	0.26	899	0.11
2012	396	0.14				
2013	949	0.09				
2014	548	0.11				
2015	583	0.09				
2016	514	0.11				
2017	716	0.08				
Pre-IFA ('96-'01)	170	0.17				
IFA Pre-Spill ('02-'07)	386	0.11				
IFA Post-Spill ('08-'09)	231	0.15				
WUP ('10-'17)	618	0.10				

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 sare provided for the Puget Sound – Keoght 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 a 2.14-fold increase in freshwater survival rates due to higher pink salmon returns, whhich occurred during the WUP period only. See text for additional details.

		SAR Index	
	Escapement	**Puget-Keogh**	**Keogh Only**
Avg. pre-IFA ('98-'01)	170	1.5%	6.0%
Avg. IFA (pre-spill '02-'07)	386	1.1%	4.2%
Avg. WUP ('10-'17)	618	2.1%	5.7%
		Adjusted	
IFA/pre-IFA escapement ratio	2.27	3.14	3.21
WUP/IFA escapement ratio	1.60	0.82	1.18
Increase in parr survival (in odd years)	4.28		
Increase in parr survival (odd and even years)	2.14		
Marine survival- and Pink salmon-adjusted WUP/IFA ratio		0.38	0.55

Table 6.1. Juvenile Steelhead abundance and survivalfor 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.

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	814		
		0+	Fall-08	236.5	29%	
		0+	Spring-09	48.6	21%	
		1+	Spring-10	18.3	38%	8%
	2009	Eggs	Spring-09	507		
		0+	Fall-09	97.7	19%	
		0+	Spring-10	22.0	22%	
		1+	Spring-11	3.5	16%	4%
	2010	Eggs	Spring-10	1,492		
		0+	Fall-10	70.0	5%	
		0+	Spring-11	31.9	46%	
		1+	Spring-12	19.6	61%	28%
	2011	Eggs	Spring-11	2,666		
		0+	Fall-11	389.4	15%	
		0+	Spring-12	87.3	22%	
		1+	Spring-13	11.56	13%	3%
	2012	Eggs	Spring-12	1,263		
		0+	Fall-12	150.3	12%	
		0+	Spring-13	48.9	33%	
		1+	Spring-14	45.6	93%	30%
	2013	Eggs	Spring-13	2,702		
		0+	Fall-13	246.7	9%	
		0+	Spring-14	52.5	21%	
		1+	Spring-15	7.0	13%	3%
	2014	Eggs	Spring-14	1,232		
		0+	Fall-14	151.1	12%	
		0+	Spring-15	22.9	15%	
		1+	Spring-16	14.20	62%	9%
	2015	Eggs	Spring-15	1,476		
		0+	Fall-15	141.4	10%	
		0+	Spring-16	32.9	23%	
		1+	Spring-17	10.5	32%	7%
	2016	Eggs	Spring-16	1,350		
		0+	Fall-16	237.2	18%	
		0+	Spring-17	56.7	24%	
		1+	Spring-18	NA	NA	NA
	2017	Eggs	Spring-17	1,831		
		0+	Fall-17	332.8	18%	

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 alternative Beverton-Holt flow covariate models predicting Steelhead fall fry abundance as a function of egg deposition and flow covariates. 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 ΔDIC is the difference in the deviance information criteria for each model relative to the model with the lowest value (the best model). Rank specifies the rank order of each model based on DIC values (rank 1 = best model = lowest DIC). Dark- and light-grey highlighted rows identify models with strong (ΔDIC=0-2) and moderate (ΔDIC=2-4) support.

			Covariate Effect (γ)						
Model	**Month**	**Covariate (m^3/sec)**	**Mean**	**LCL**	**UCL**	**Prob>0**	**r^2**	**ΔDIC**	**Rank**
1	Jul	None					0.25	8.0	13
2		Upramp_6Hrs	-0.251	-0.569	0.083	7.2	0.53	7.1	9
3		Downramp_6Hrs	0.013	-0.387	0.450	50.2	0.25	11.3	31
4		Upramp_1Hr	-0.043	-0.433	0.384	38.8	0.27	11.0	24
5		Downramp_1Hr	0.013	-0.402	0.455	50.2	0.25	11.3	34
6		Avg_Q	0.029	-0.386	0.471	53.7	0.25	11.2	30
7		SD_Q	0.092	-0.316	0.532	67	0.27	11.1	27
8		Max_Q	0.049	-0.368	0.494	58.2	0.25	11.3	32
9		Prop_Hrs_<40	0.249	-0.089	0.607	93.9	0.44	8.5	15
10		Prop_Hrs_<60	0.061	-0.338	0.482	60.9	0.25	11.2	28
11		Prop_Hrs_<80	-0.047	-0.437	0.369	38.1	0.27	11.0	22
12		Prop_Hrs_<100	-0.005	-0.403	0.419	47.2	0.26	11.2	29
13	Aug	Upramp_6Hrs	-0.326	-0.578	-0.061	0.7	0.67	3.4	5
14		Downramp_6Hrs	-0.346	-0.584	-0.097	0.1	0.71	2.1	4
15		Upramp_1Hr	-0.215	-0.545	0.136	11.5	0.45	8.2	14
16		Downramp_1Hr	-0.369	-0.587	-0.147	0	0.77	0.0	1
17		Avg_Q	-0.247	-0.567	0.088	7.8	0.51	7.2	11
18		SD_Q	-0.150	-0.518	0.249	20.1	0.38	9.7	18
19		Max_Q	-0.182	-0.534	0.190	15.5	0.42	9.0	17
20		Prop_Hrs_<40	0.199	-0.170	0.587	87	0.35	9.8	20
21		Prop_Hrs_<60	0.338	0.065	0.626	98.6	0.61	4.7	8
22		Prop_Hrs_<80	0.340	0.072	0.626	98.7	0.62	4.6	7
23		Prop_Hrs_<100	0.280	-0.044	0.617	96.2	0.49	7.5	12
24	Jul-Aug	Upramp_6Hrs	-0.354	-0.589	-0.109	0.1	0.73	1.6	3
25		Downramp_6Hrs	-0.322	-0.576	-0.056	0.9	0.66	3.6	6
26		Upramp_1Hr	-0.249	-0.567	0.087	7.5	0.51	7.1	10
27		Downramp_1Hr	-0.363	-0.589	-0.127	0	0.75	0.7	2
28		Avg_Q	-0.081	-0.482	0.345	31.5	0.30	10.7	21
29		SD_Q	0.047	-0.370	0.492	57.6	0.25	11.3	33
30		Max_Q	-0.049	-0.458	0.384	37.2	0.28	11.0	23
31		Prop_Hrs_<40	0.242	-0.101	0.607	93.2	0.42	8.8	16
32		Prop_Hrs_<60	0.200	-0.168	0.590	86.9	0.36	9.8	19
33		Prop_Hrs_<80	0.083	-0.309	0.508	65.6	0.26	11.1	25
34		Prop_Hrs_<100	0.071	-0.323	0.490	63.2	0.26	11.1	26

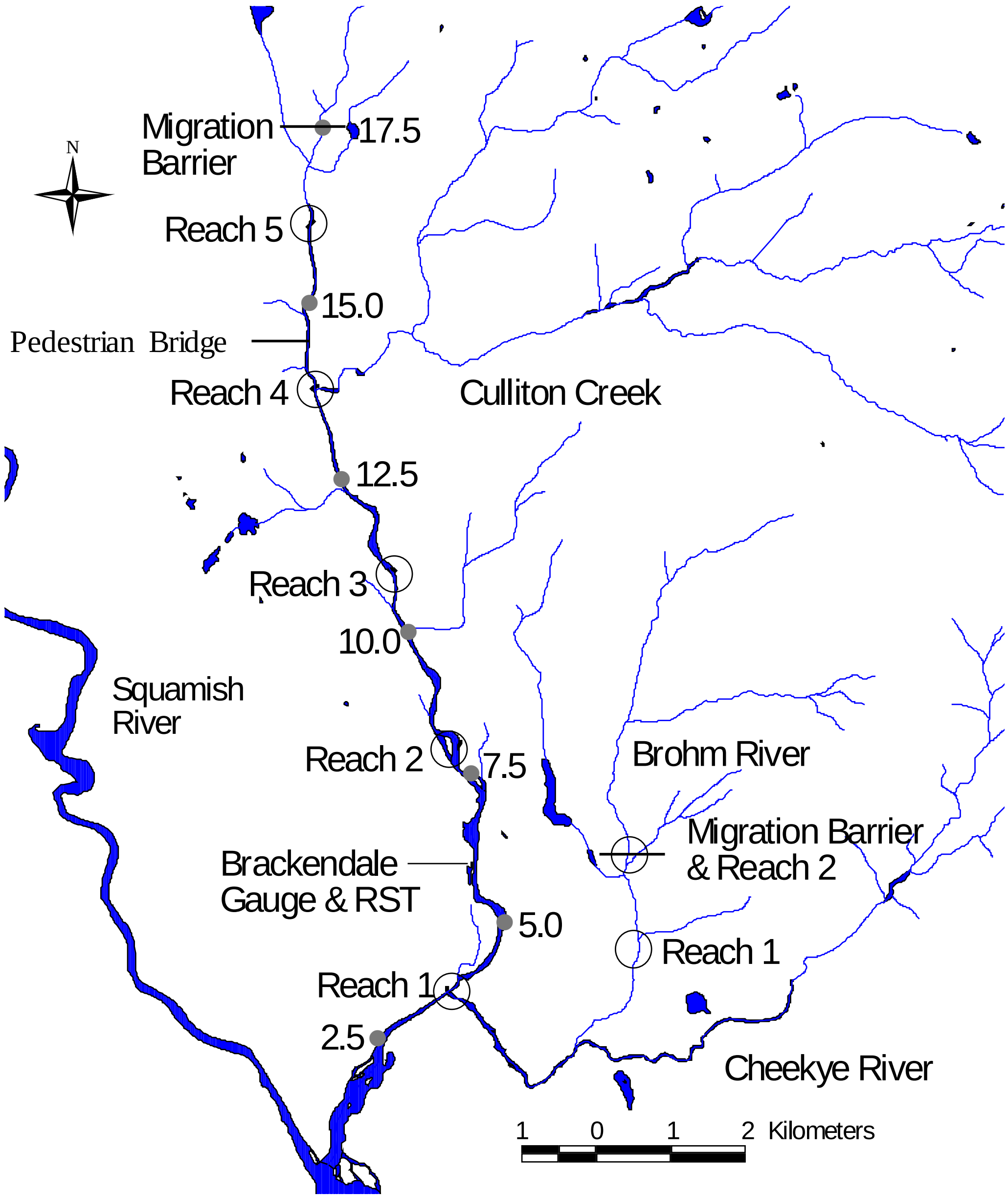


Figure 1.1. Map of the Cheakamus River study area showing the locations of the upstream limit of reach breaks used for habitat and juvenile surveys (open circles), distance (km) from the Squamish River confluence (gray points), migration barriers for anadromous fish in the Cheakamus and Brohm Rivers, 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.

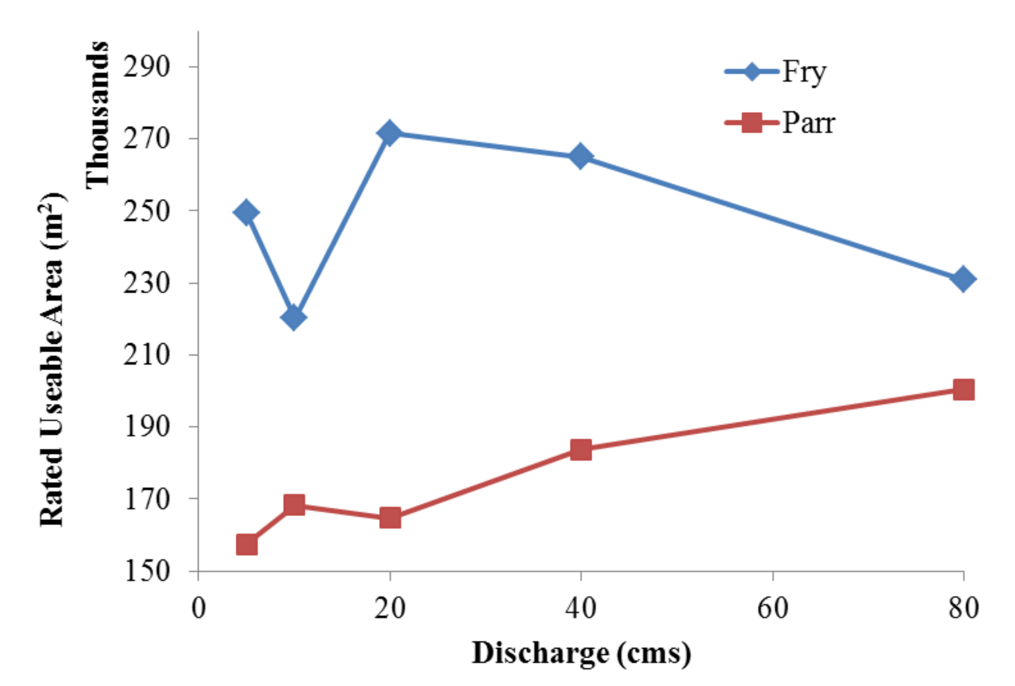


Figure 1.2. Changes in predicted useable juvenile Steelhead habitat in the Cheakamus River (summed across reaches) as a function of discharge. This habitat model was used in the initial WUP assessment (BC Hydro 2007).

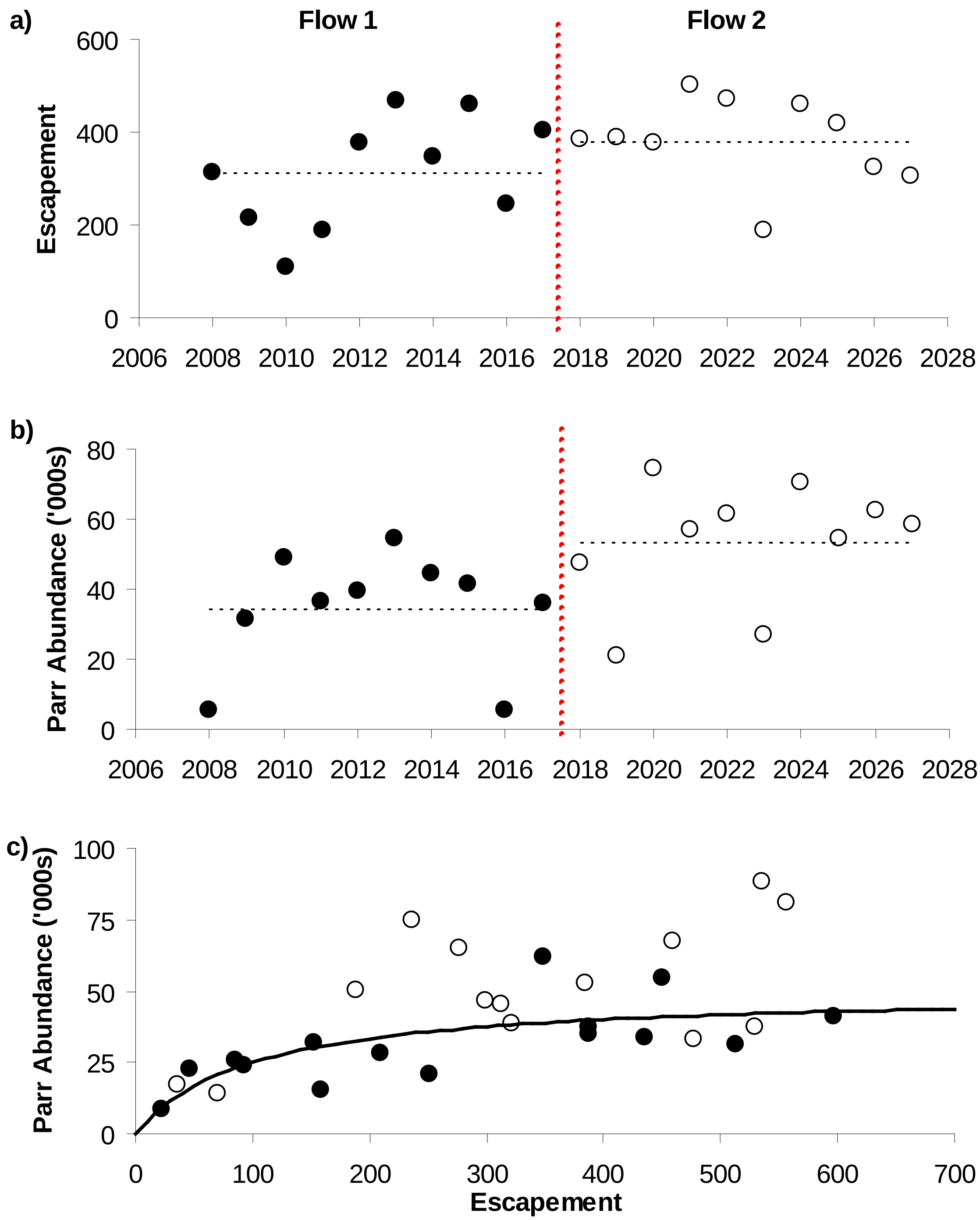


Figure 1.3. Theoretical responses of escapement (a) and parr abundance (b) under two flow regimes, with 10 years of data collected under each regime, and the stock-recruit relationship between these life stages over the two periods (c). Solid and open circles represent data collected under flow regimes 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).

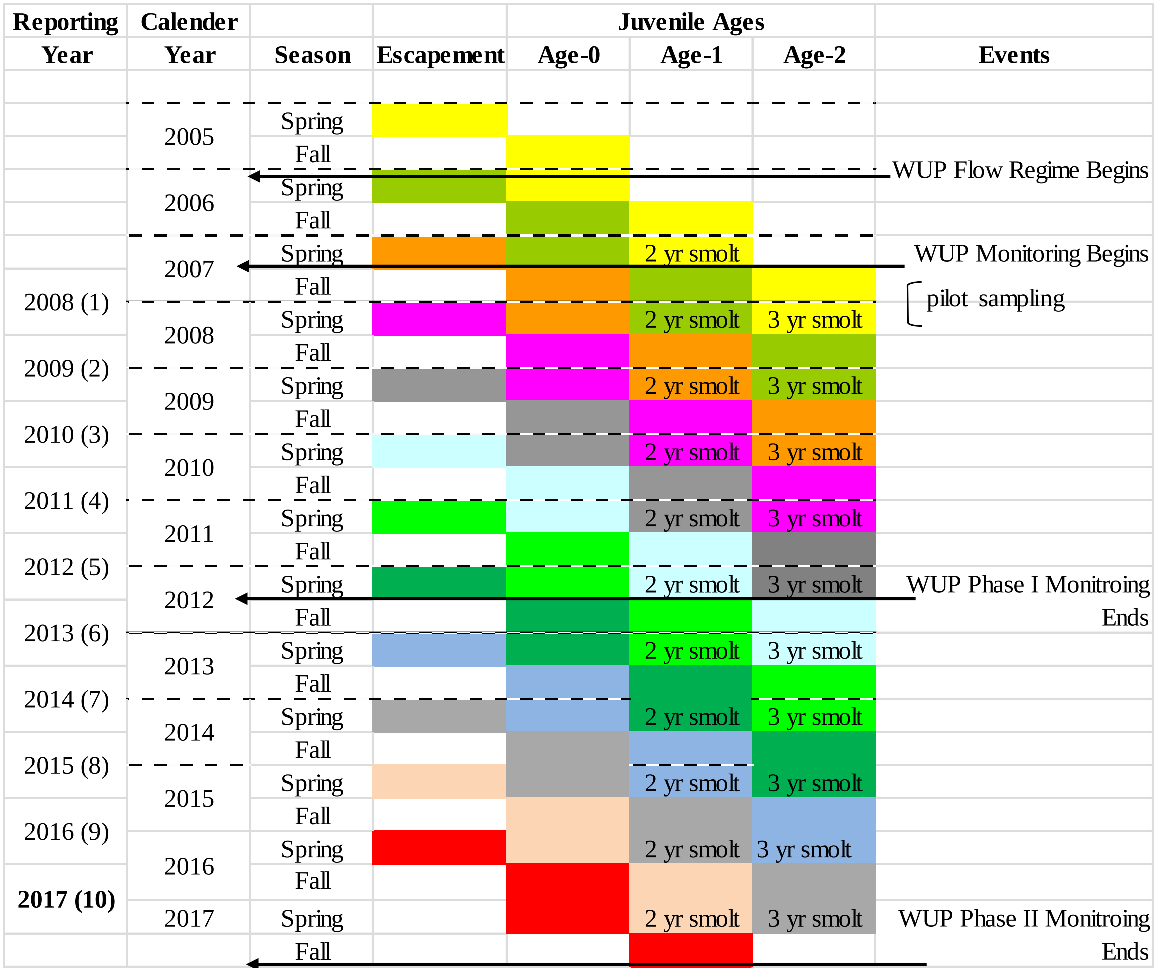


Figure 1.4. Life history table for the freshwater life stages of Steelhead in the Cheakamus River in relation to annual and seasonal monitoring periods, WUP assessments and reporting periods, and implementation of the WUP flow regime. This report covers reporting years 1-10. Each color tracks the cohort 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.

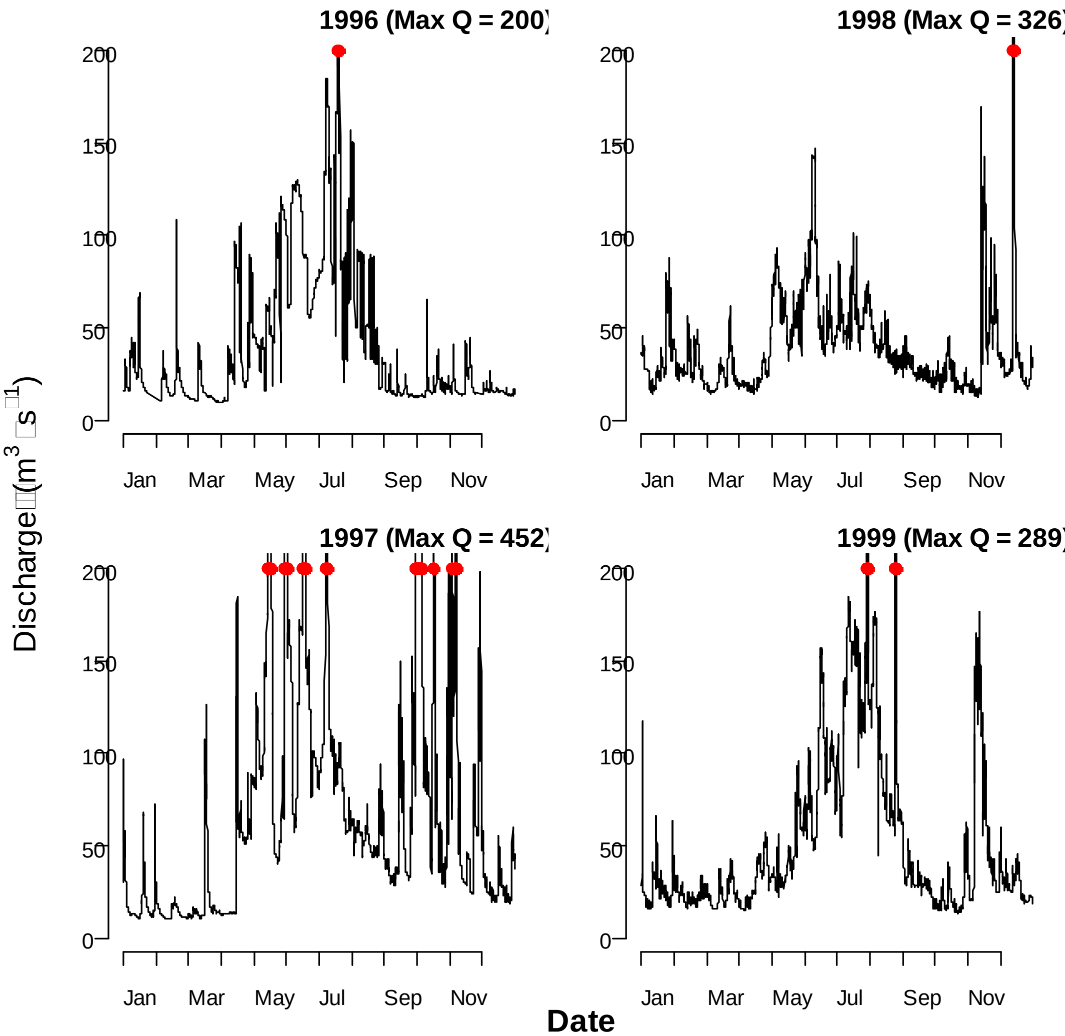


Figure 2.1. Hourly discharge at the WSC Brackendale gauge 1996-2017. Red points denote hours when discharge exceeded the y-axis maxima of 200 $m^3 s^{-1}$.

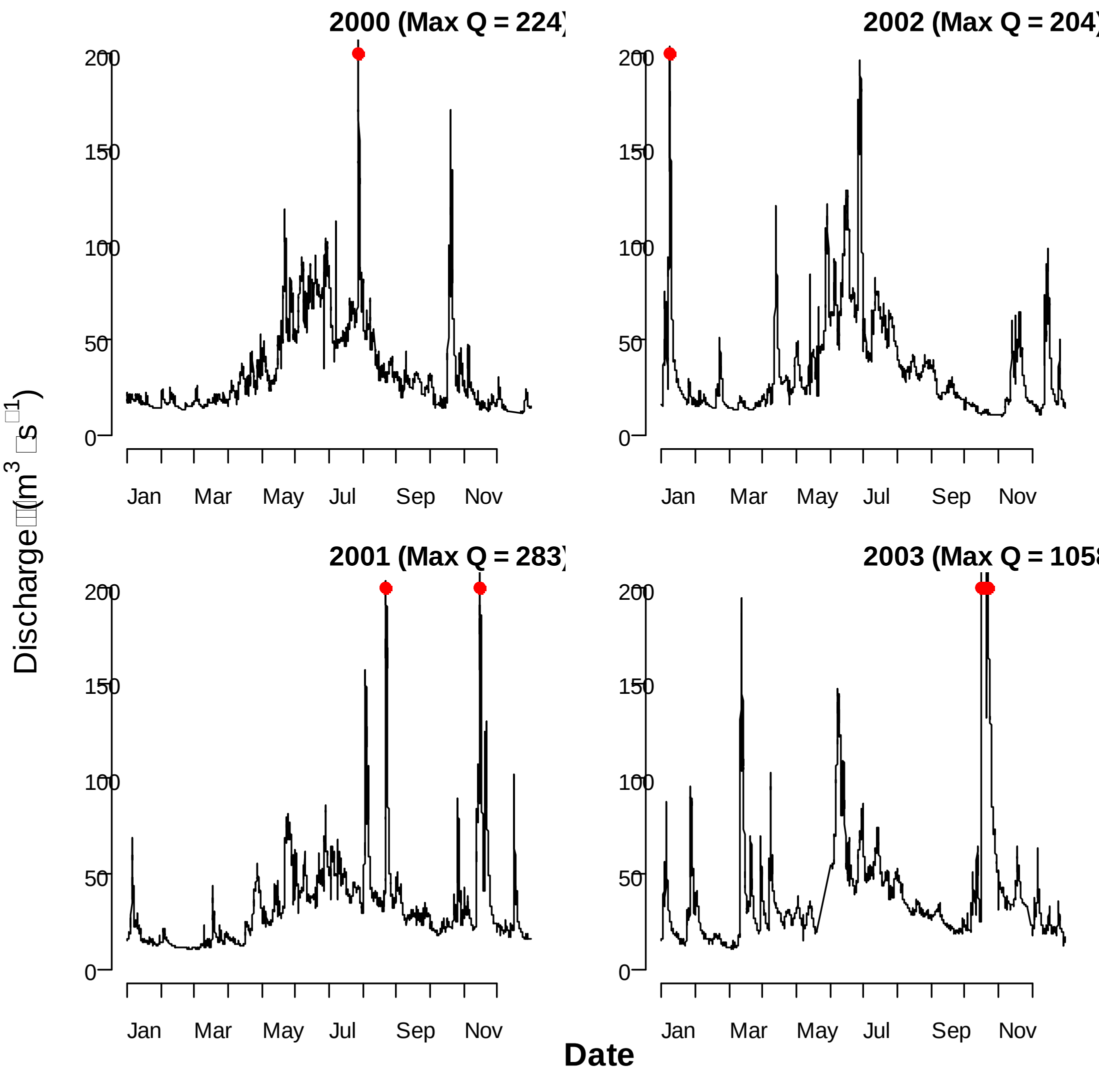


Figure 2.1. Con't.

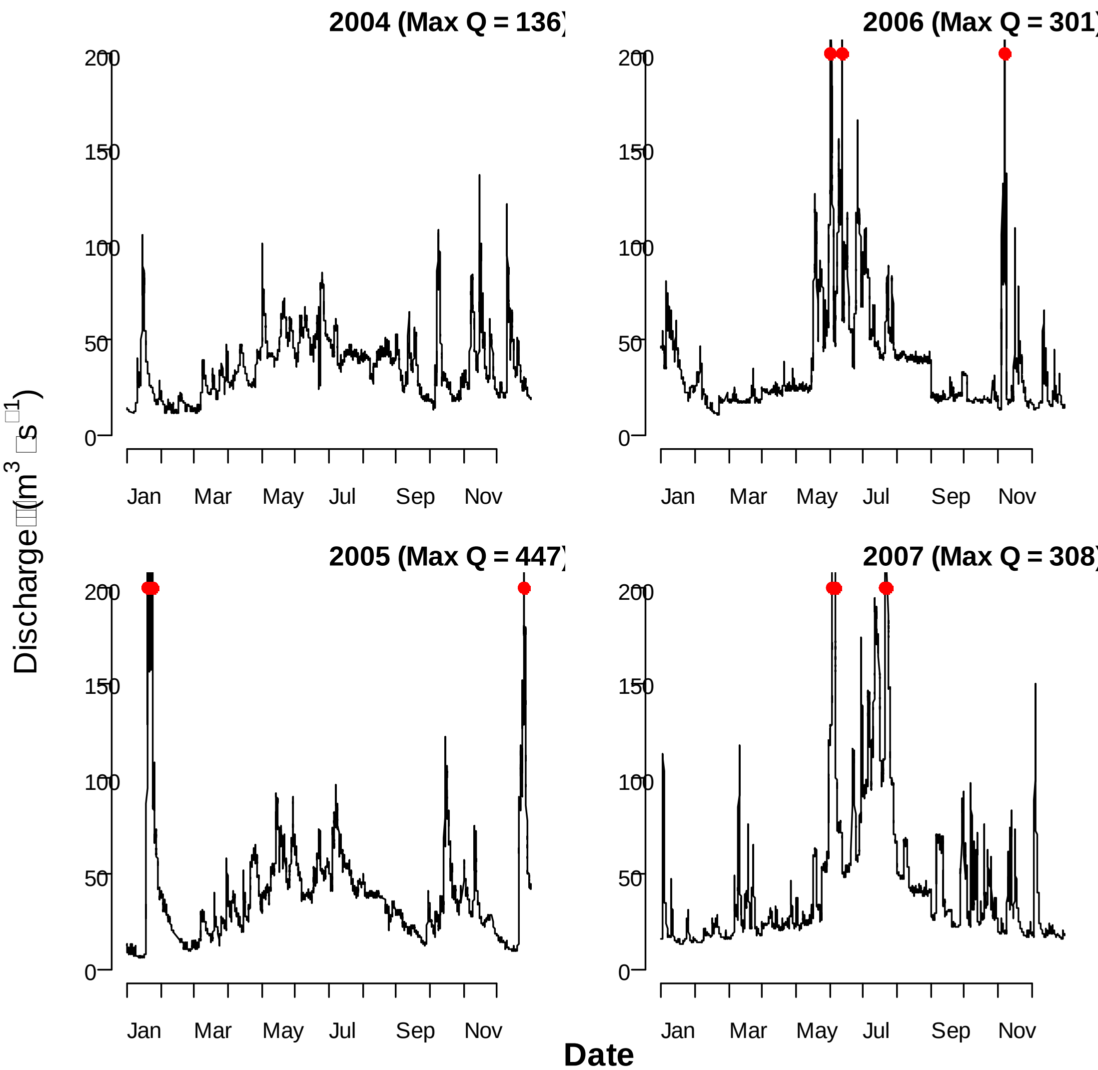
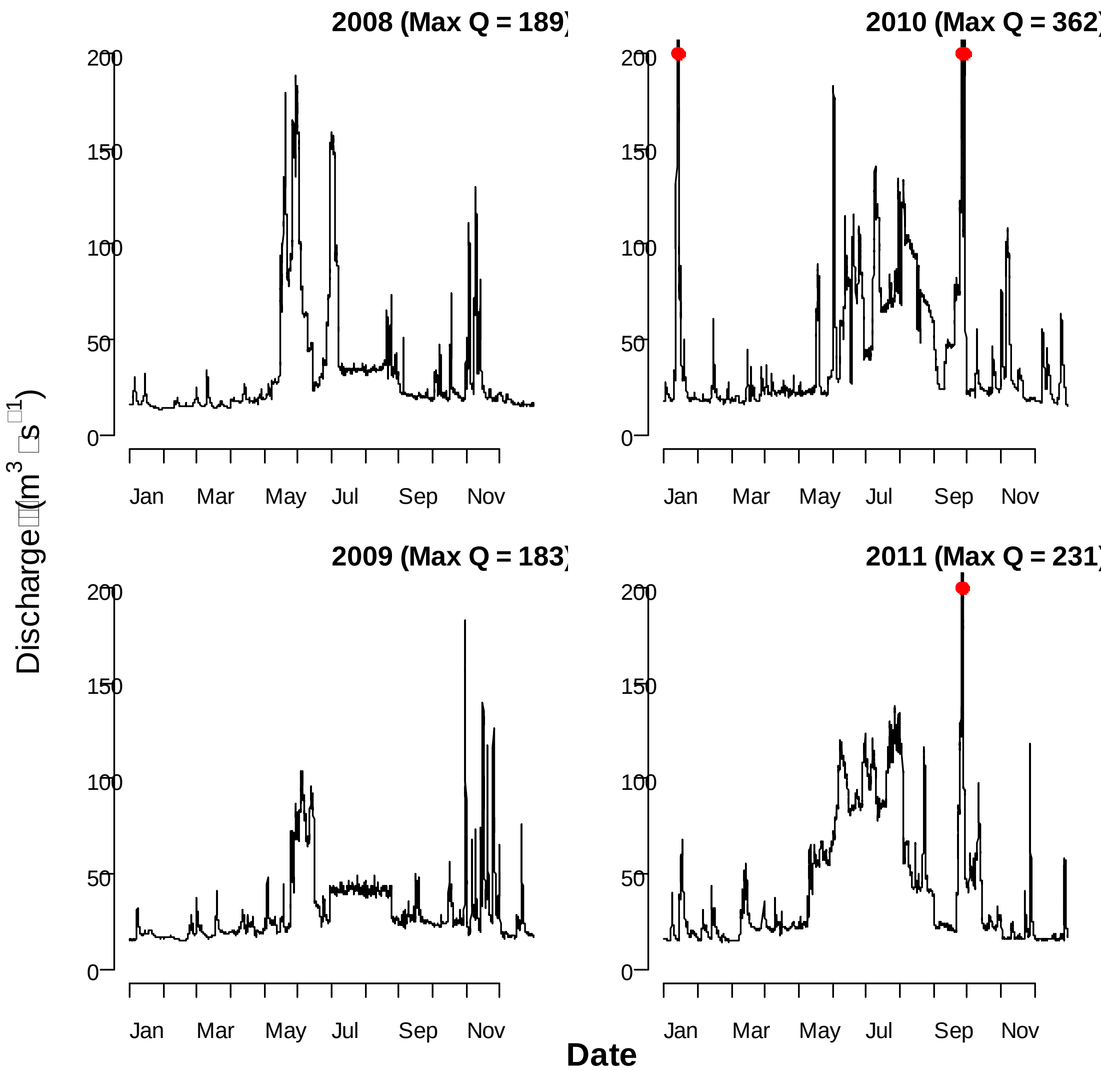


Figure 2.1. Con't.



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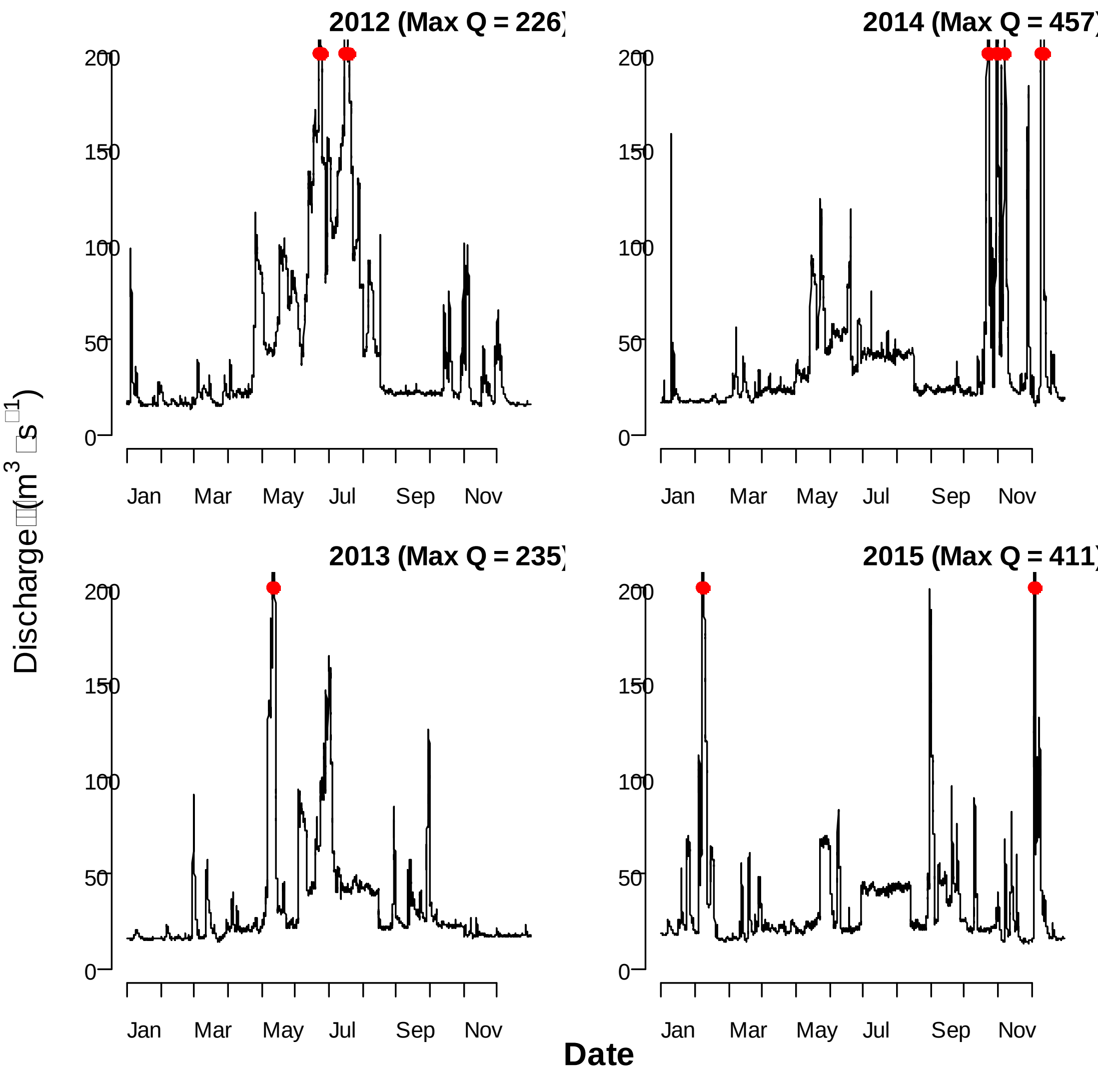

Figure 2.1. Con't.

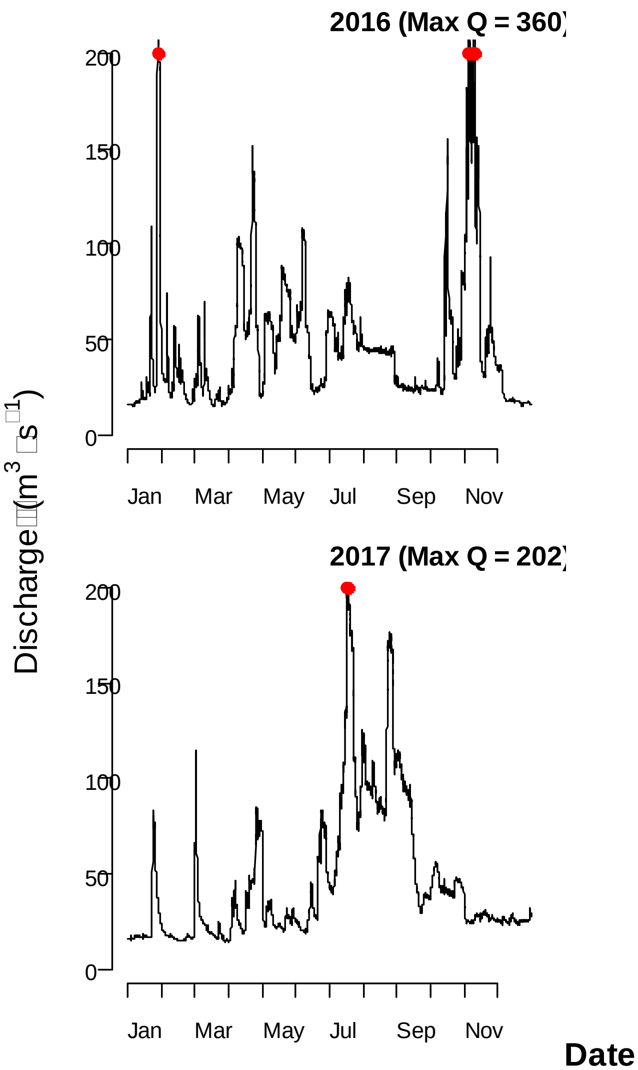

Figure 2.1. Con't.

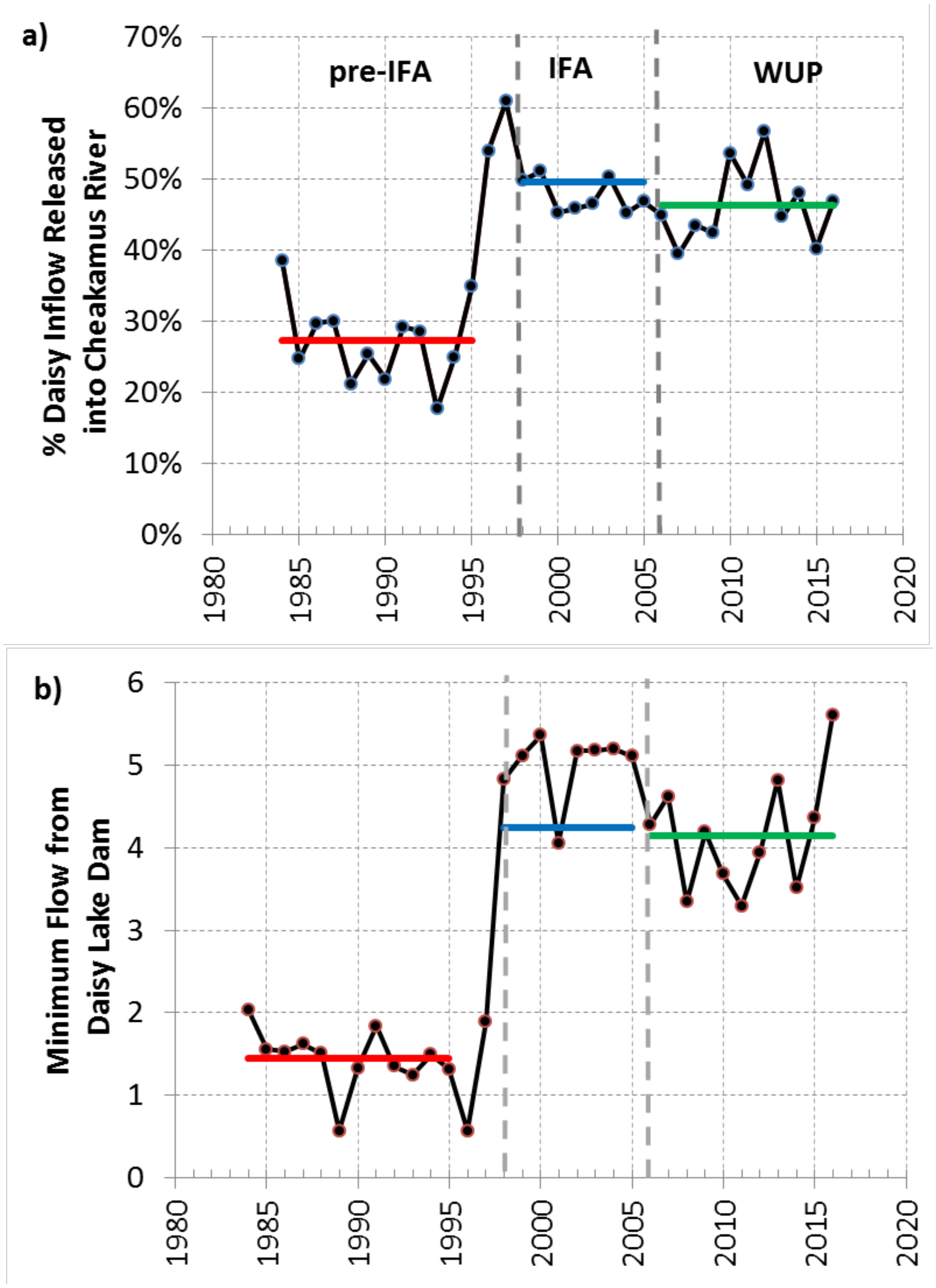

Figure 2.2. Annual percentage of inflow to Daisy Lake Reservoir released into the Cheakamus River from Daisy Lake Dam(a), and minimum flow releases from Daisy Lake Dam. Red, blue and green lines show the average levels prior to the Instream Flow Agreement (IFA), during the IFA period, and during the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

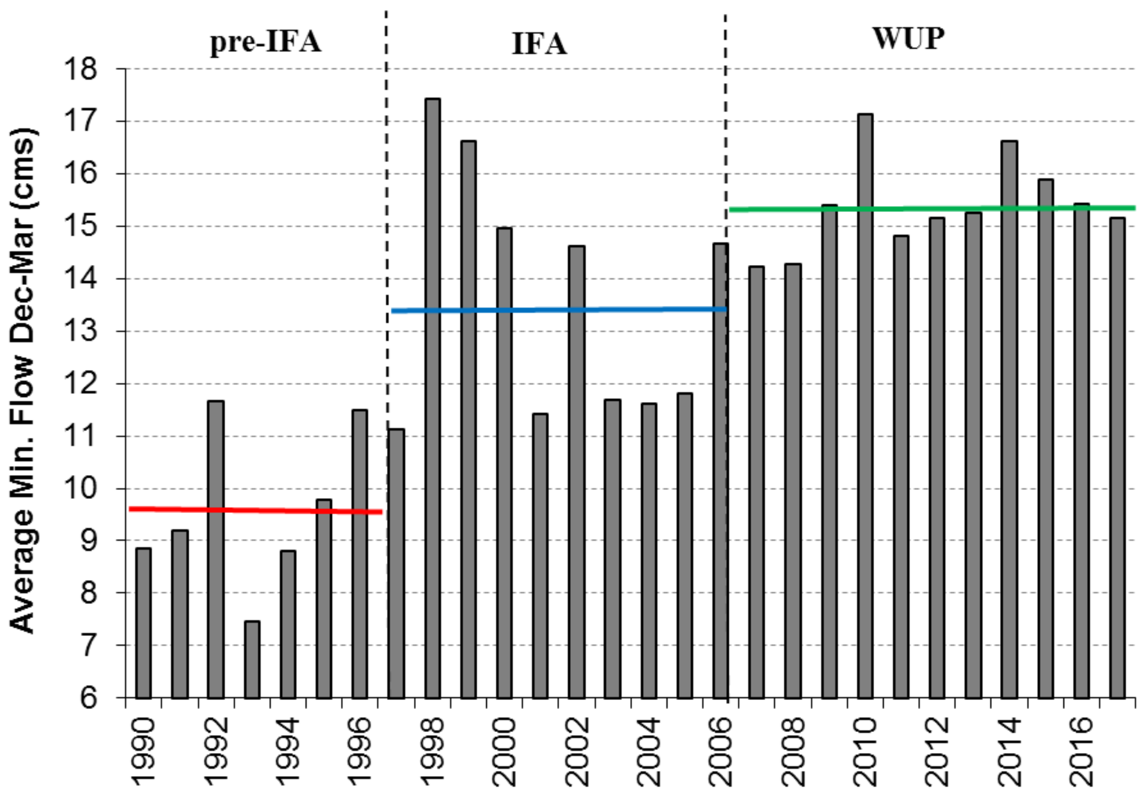

Figure 2.3. The average minimum flows during the winter at the Brackendale gauge on the Cheakamus River, 1990-2017. The average minimum flow between December and March was computed as the average of the minimum flow in December from the previous year (based on average daily flows), and the minimum flows 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.

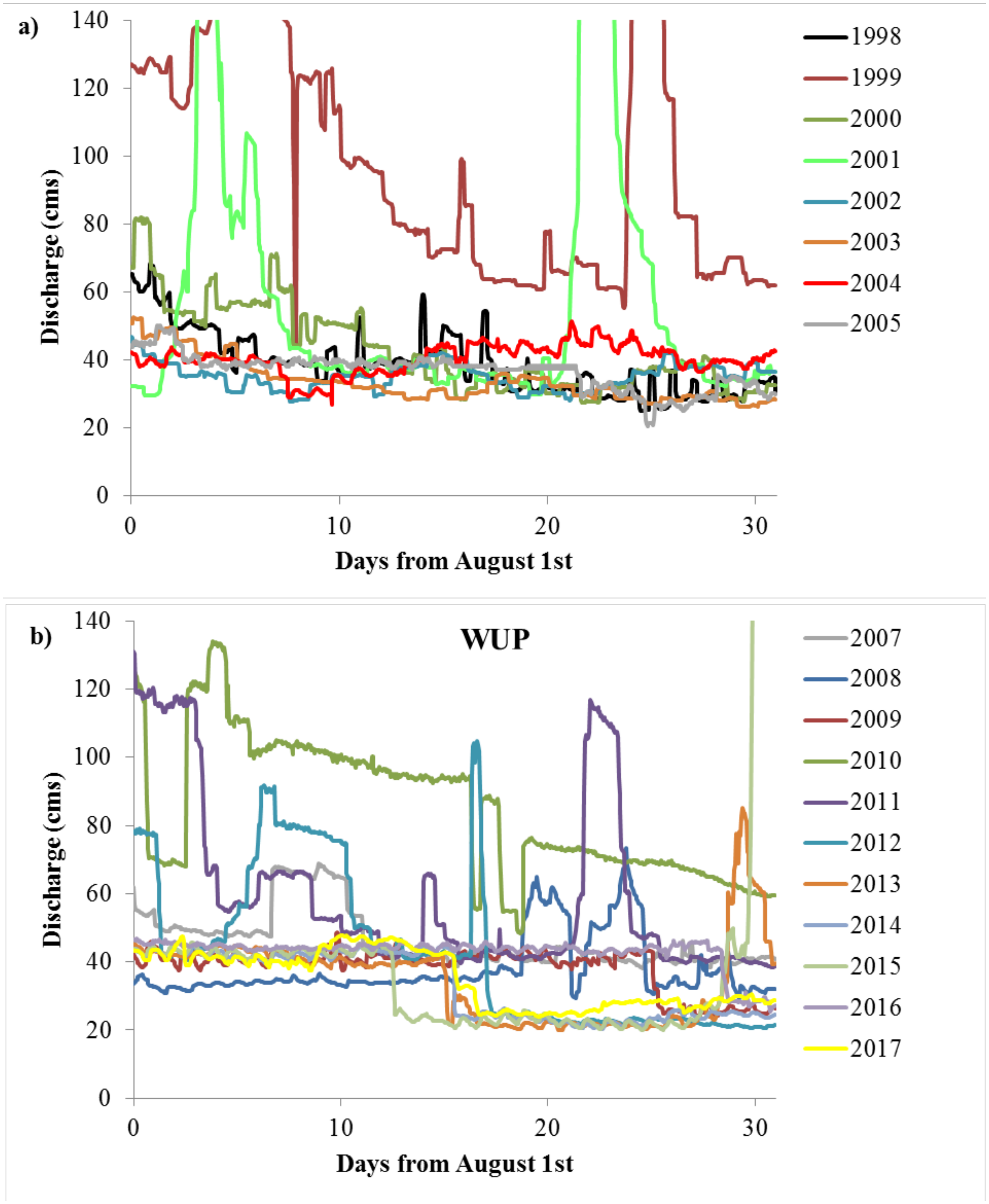

Figure 2.4. Hourly discharge at the Brackendale gauge on the Cheakamus River in August (a) during the IFA (a) and WUP (b) flow regime periods.

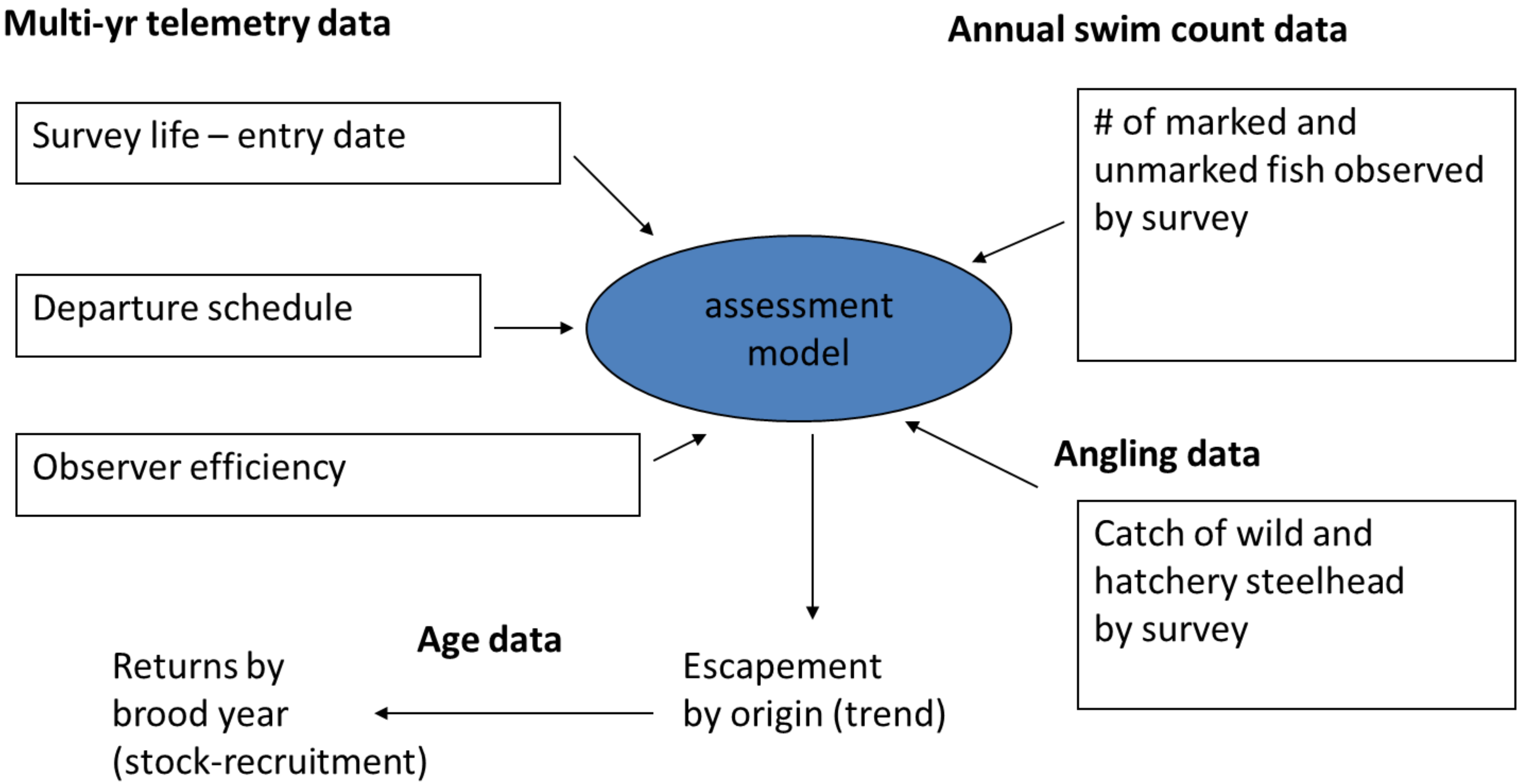

Figure 3.1. Diagram showing how different data sets contribute to the model that estimates Cheakamus River steelhead escapement and the spawner-adult stock-recruitment model.

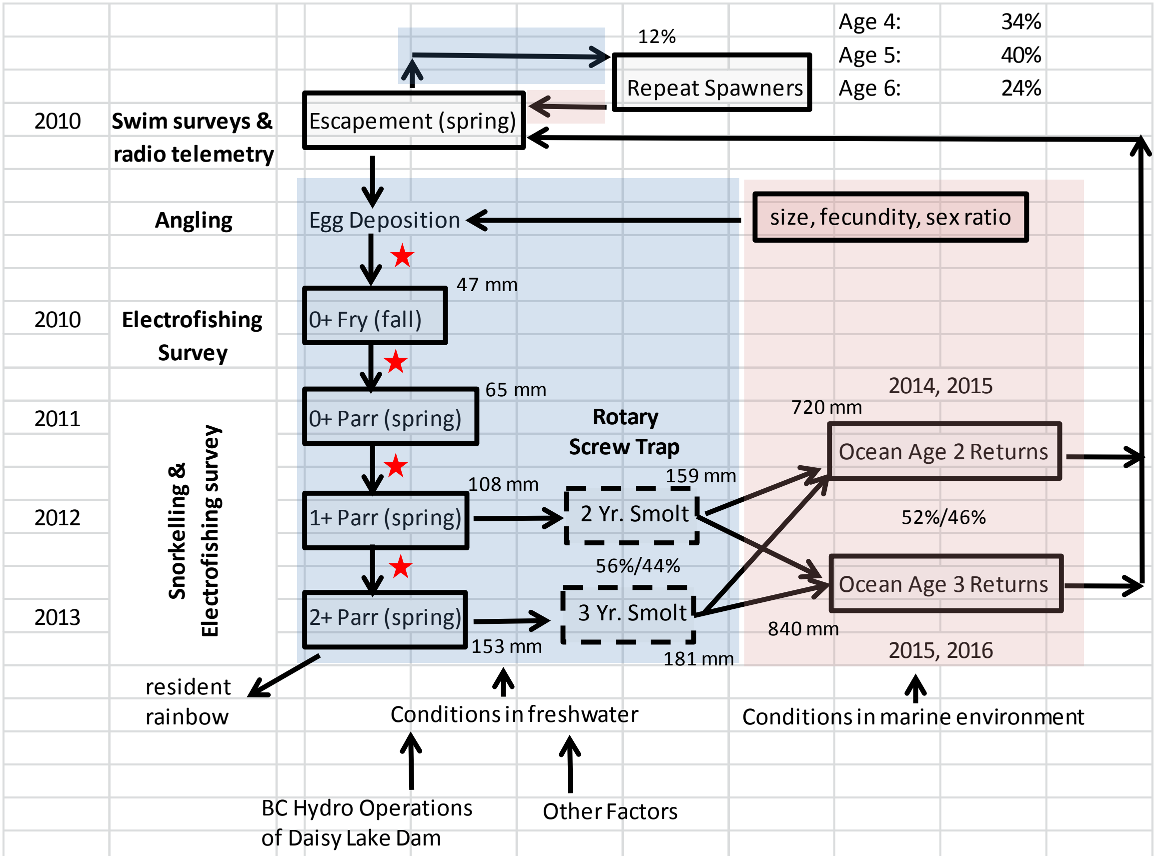

Figure 4.1. Summary of Steelhead life history in the Cheakamus River in relation to WUP monitoring activities. The years on the left of the diagram track the timing of a cohort spawned in 2010 to a 3 yr. smolt exiting the Cheakamus River in 2013 or remaining in the river as a resident trout. The typical 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.

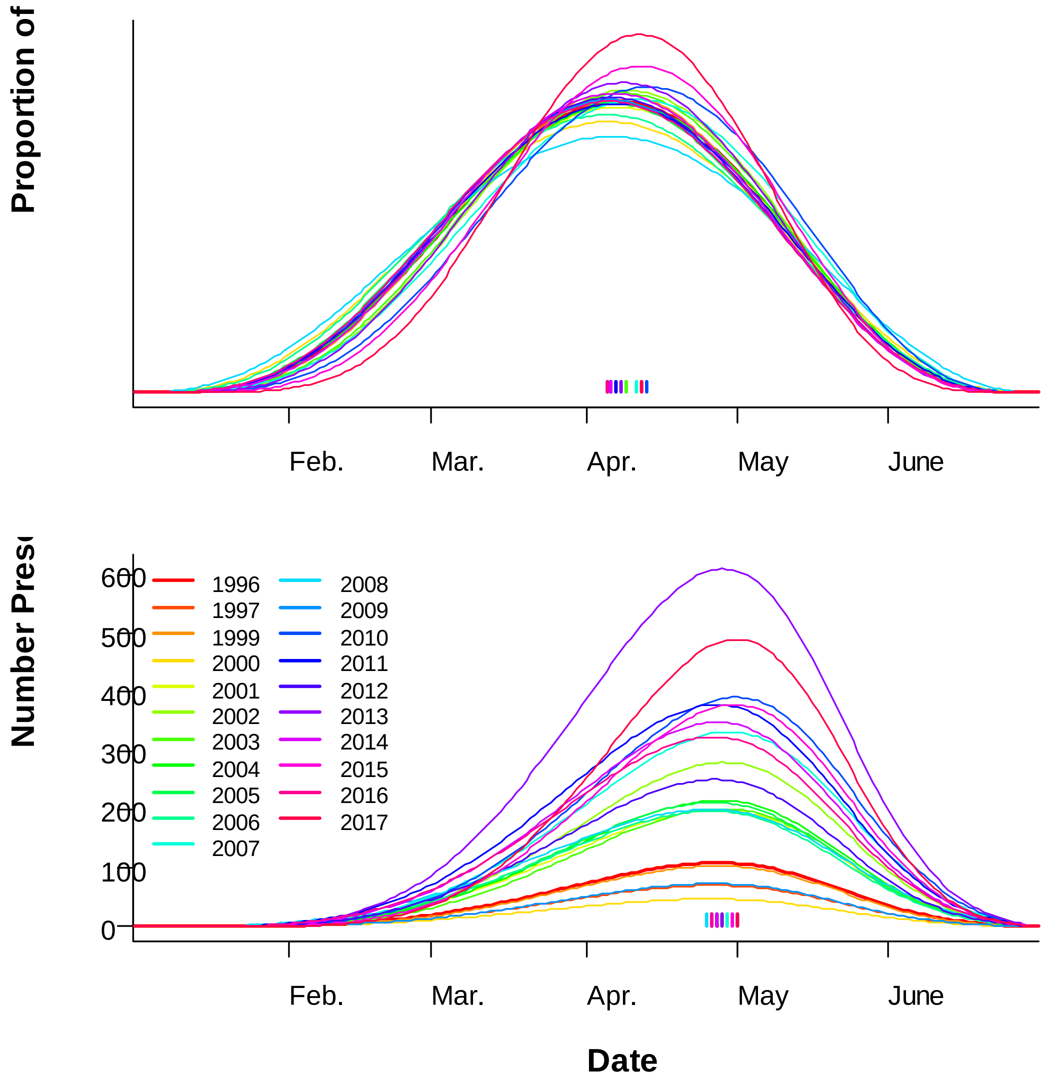

Figure 4.2. Arrival-timing of Steelhead spawners to the Cheakamus River upstream of the Cheekye-Cheakamus confluence. The top plot shows arrival timing as a proportion of the total run arriving by date. The bottom plot shows the number present in the survey area by date, which is the difference between 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.

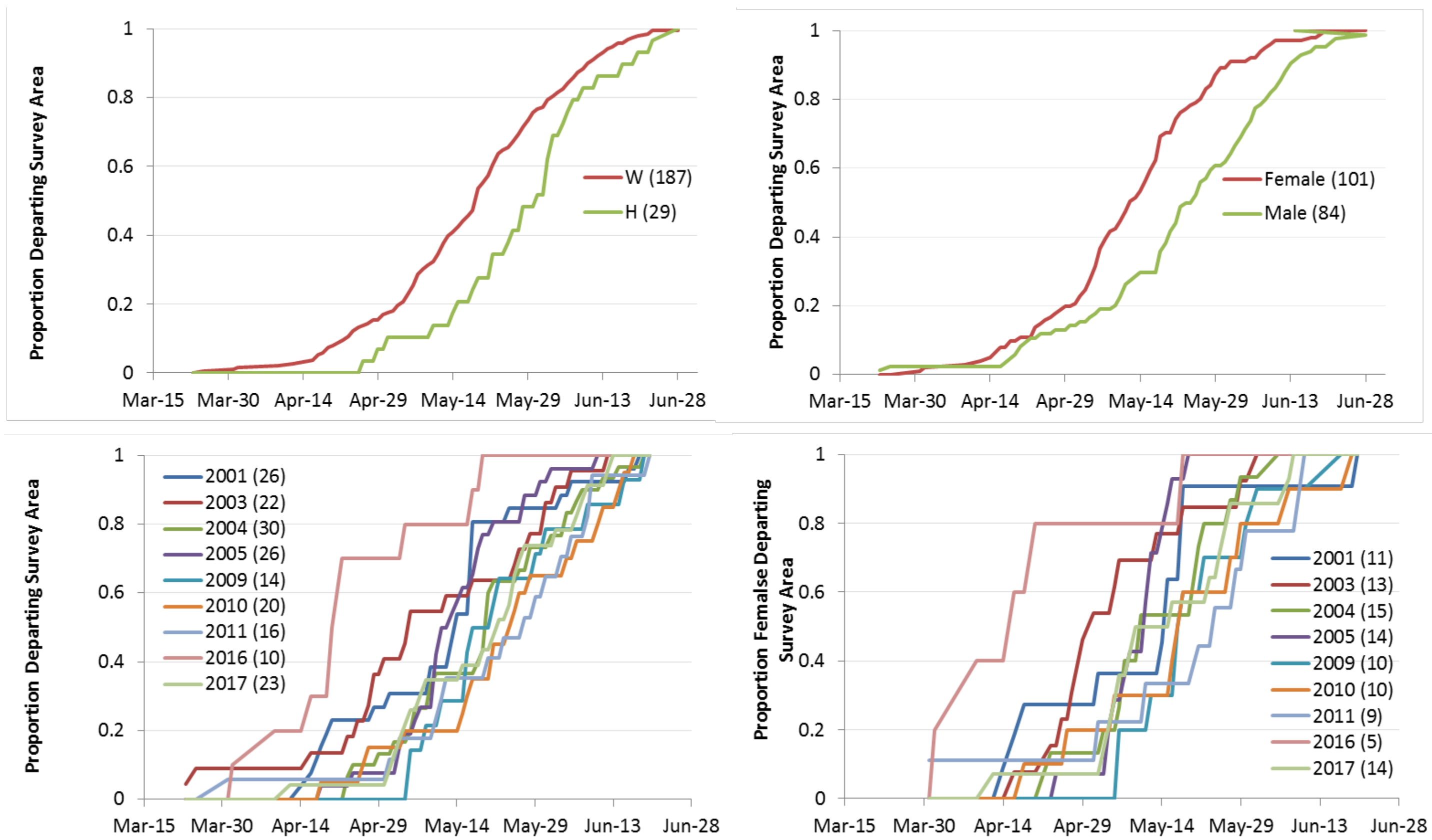

Figure 4.3. Cumulative proportion of radio tagged steelhead departing the Cheakamus River survey area by date based on data from all years radio telemetry was conducted. Numbers in parentheses in the legends denote the sample size (# of tagged fish recorded as departing at the fixed station). Owing to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

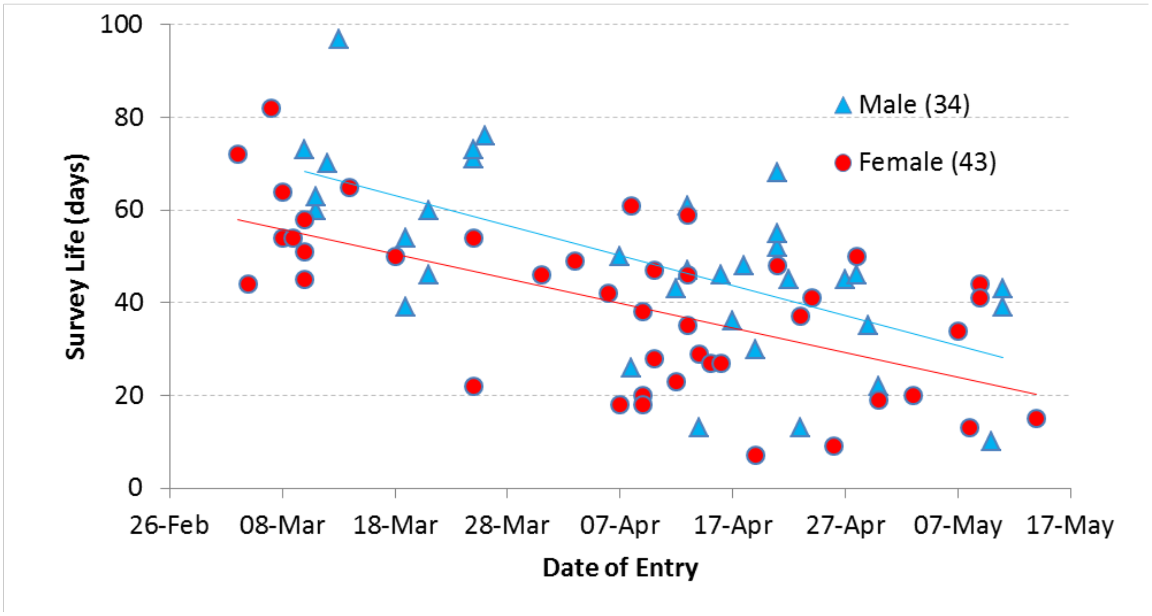

Figure 4.4. Relationship between date of entry and duration of time spent in survey area (upstream of the Cheekye-Cheakamus confluence) for male and female steelhead in the Cheakamus River based on data from all years telemetry was conducted. Numbers in parentheses in the legend denote the sample size.

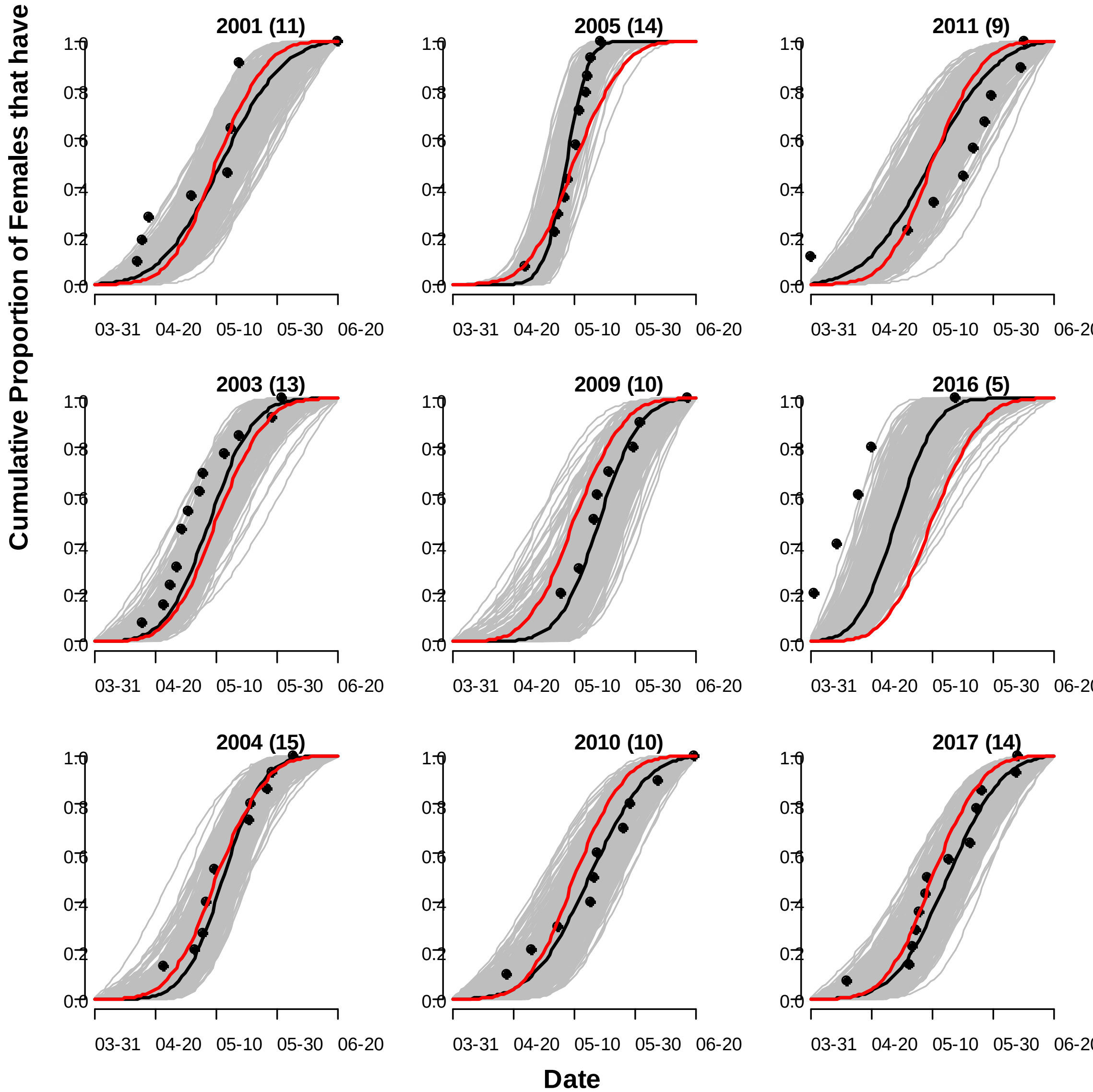

Figure 4.5. Estimated timing of departure from the Cheakamus River survey area for radio tagged female steelhead. Points show the cumulative proportion of females that have departed by date. The black curves show the best-fit depature timing for that year and the red curve shows the average curve across years. Light grey curves show the extent of uncertainty in year-specific predicitons 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).

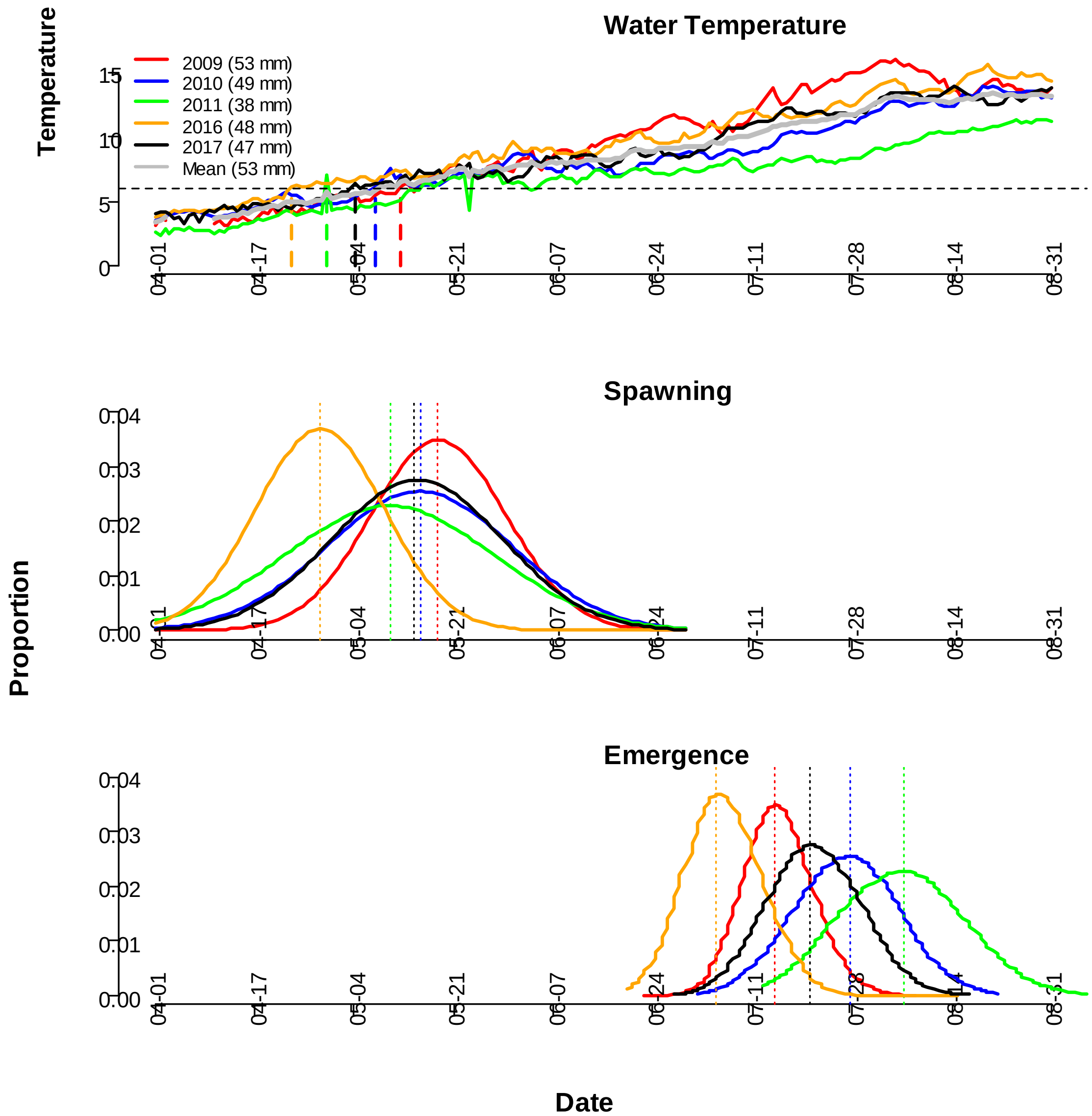

Figure 4.6. Predicted emergence timing of Steelhead in the Cheakamus River based on water temperature and spawn-timing distributions. Water temperatures over the potential incubation period are shown in the top panel. The middle panel shows spawn-timing distributions based on the modelled departure date 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 paenl denotes 6 ^{o}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 showen by the vertical dashed lines in the middle panel.

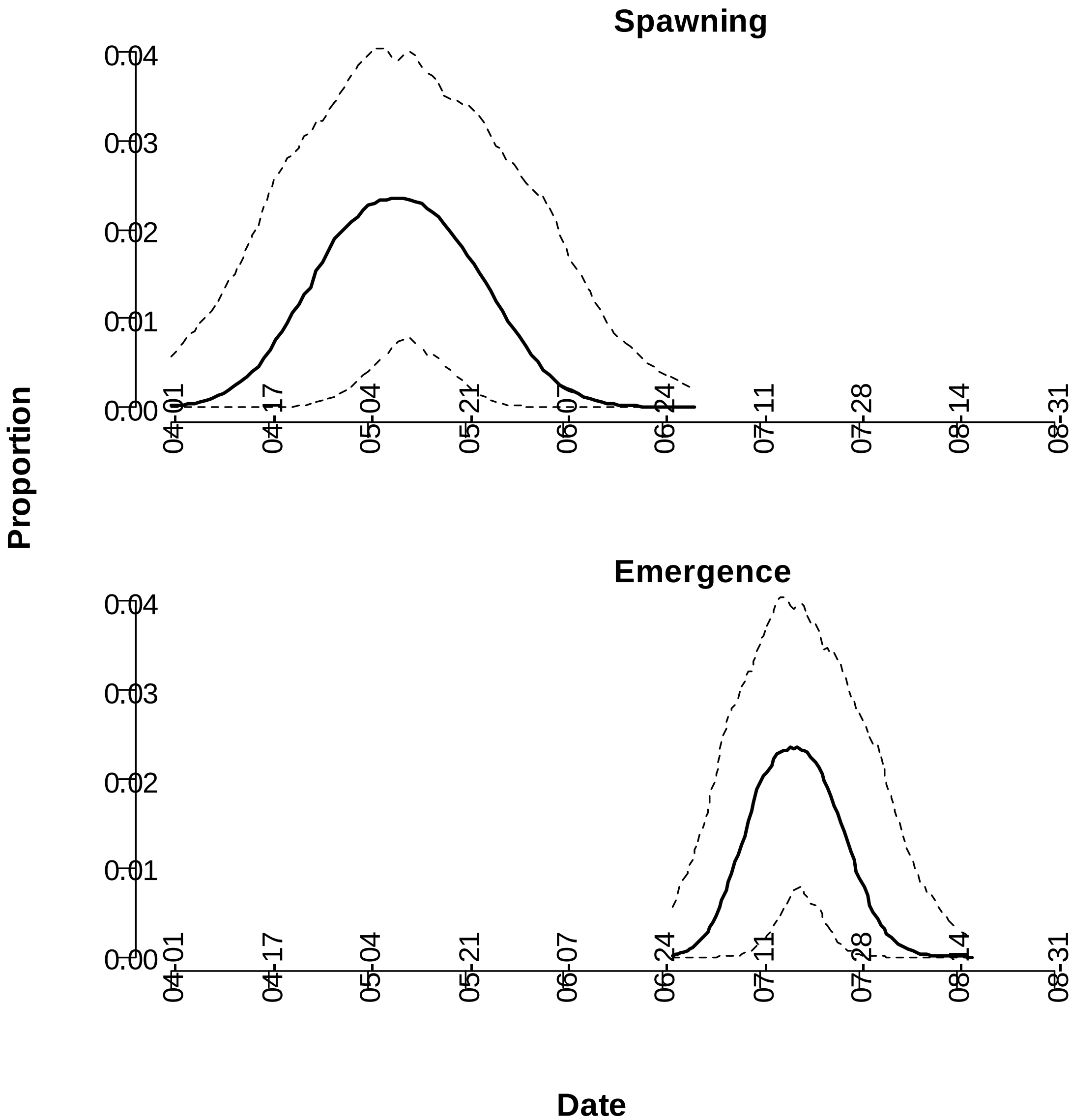

Figure 4.7. Average spawning and emergence timing for Steelhead in the Cheakamus River. Solid and dashed lines show the mean and 80% credible interval. Results are based on the average number of days to emergence by day between 2008 and 2017 (based on water temperatures) and the average spawn timing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

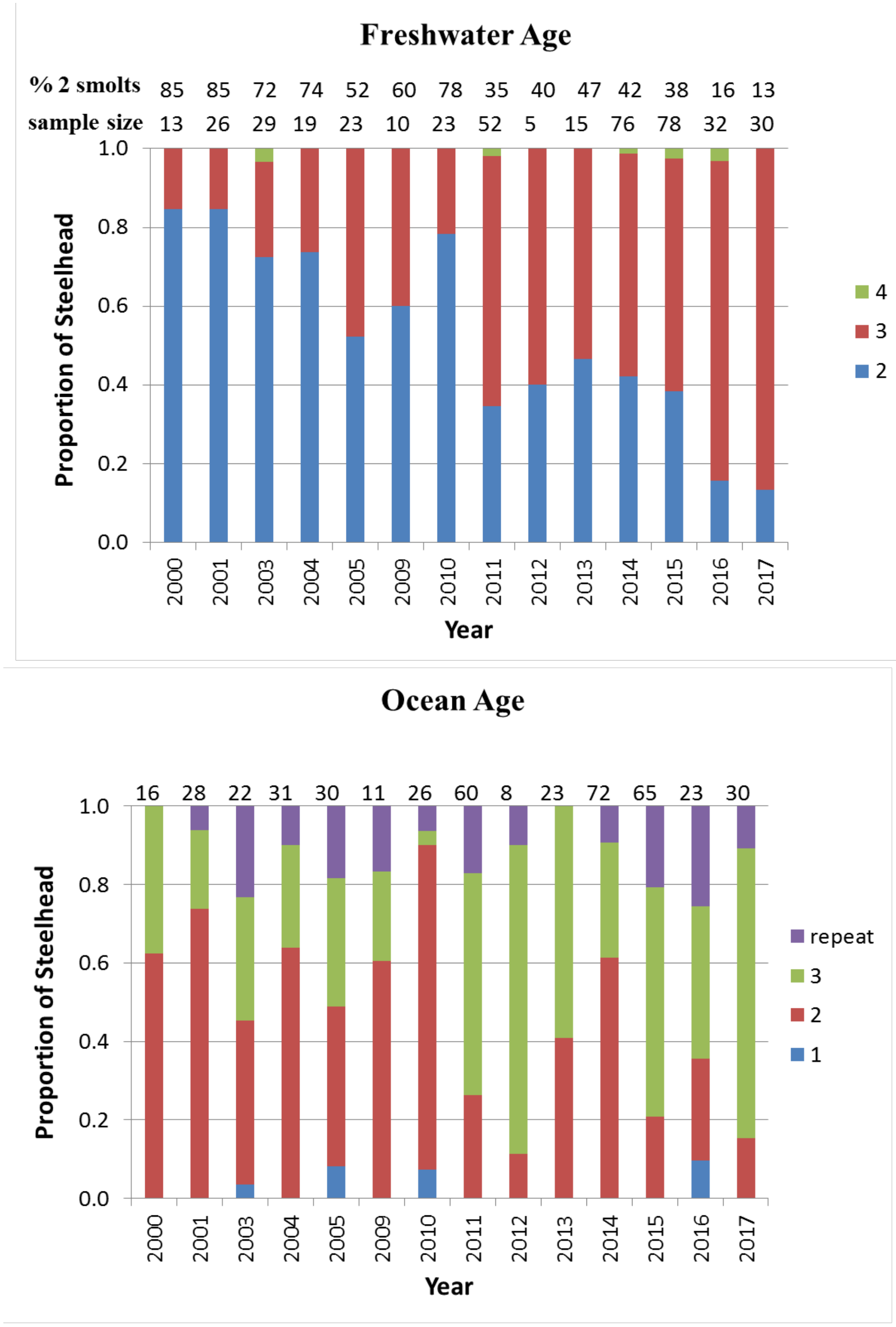

Figure 4.8. Proportion of Steelhead in the Cheakamus River by freshwater and ocean age as determined by scales collected from returning spawners. Freshwater and ocean age (in years) is the number of winters spent in freshwater and the ocean, respectively. Year on the x-axis denotes the year that scales 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). The % 2 smolts numbers at the top of the top panel shows the percentage of age 2 year smolts as determined from scale reads from smolts collected at the RST. Year in this case (on the x-axis) refers to the year of outmigration.

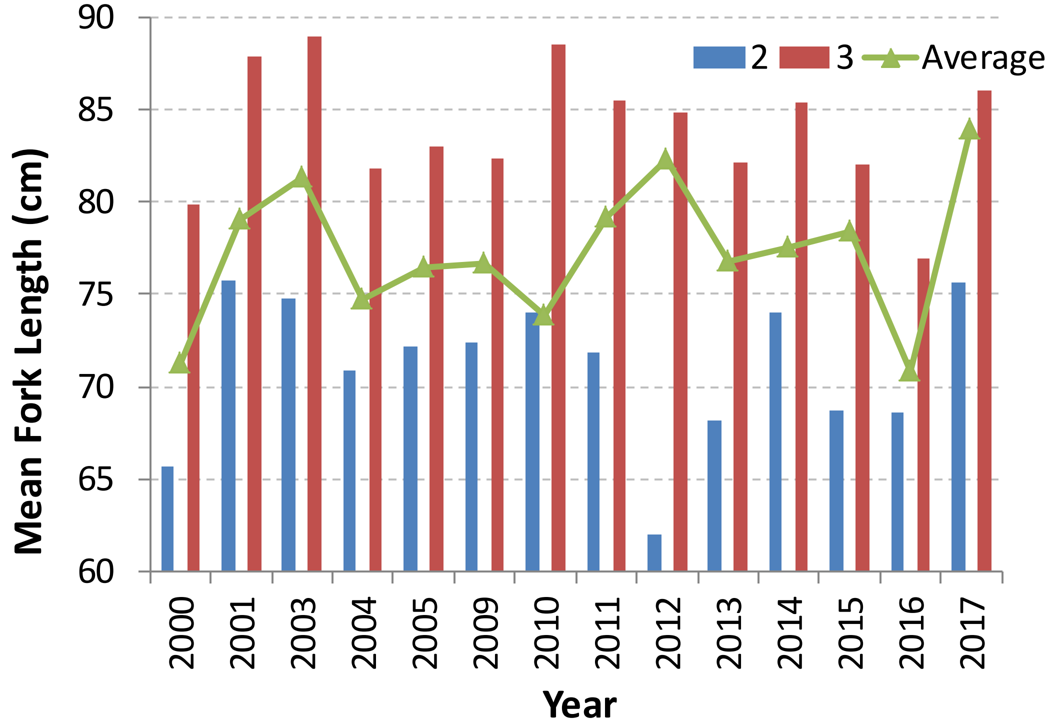

Figure 4.9. Mean size of returning Steelead spawners by ocean age and the average size for all fish captured.

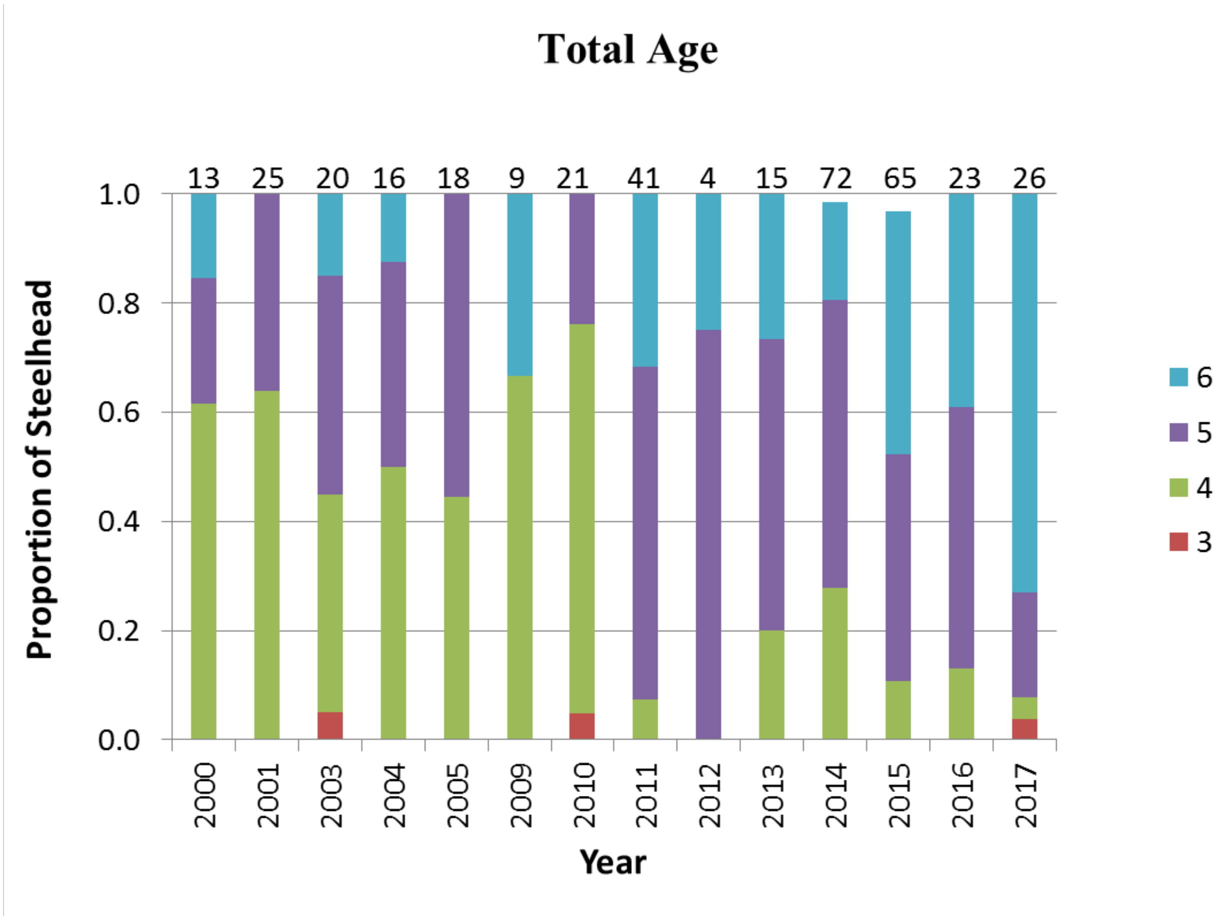

Figure 4.10. Proportion of Steelhead spawners returning to the Cheakamus River by total age. Numbers at the top of each bar denote the number of fish where a total age could be determined (i.e., both freshwater and ocean age could be determined).

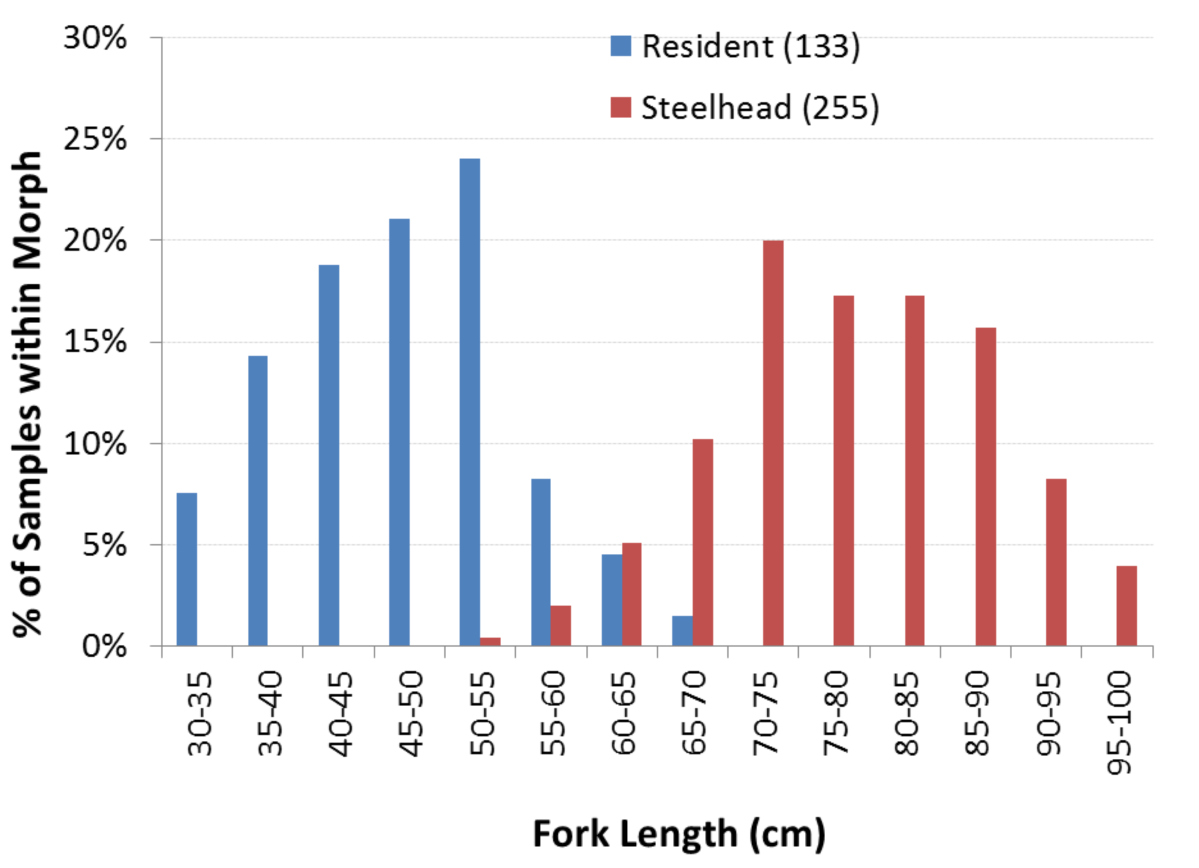

Figure 4.11. Comparison of size distribution of resident rainbow trout and steelhead in the Cheakamus River based on collection of 388 scales between 2009 and 2017.

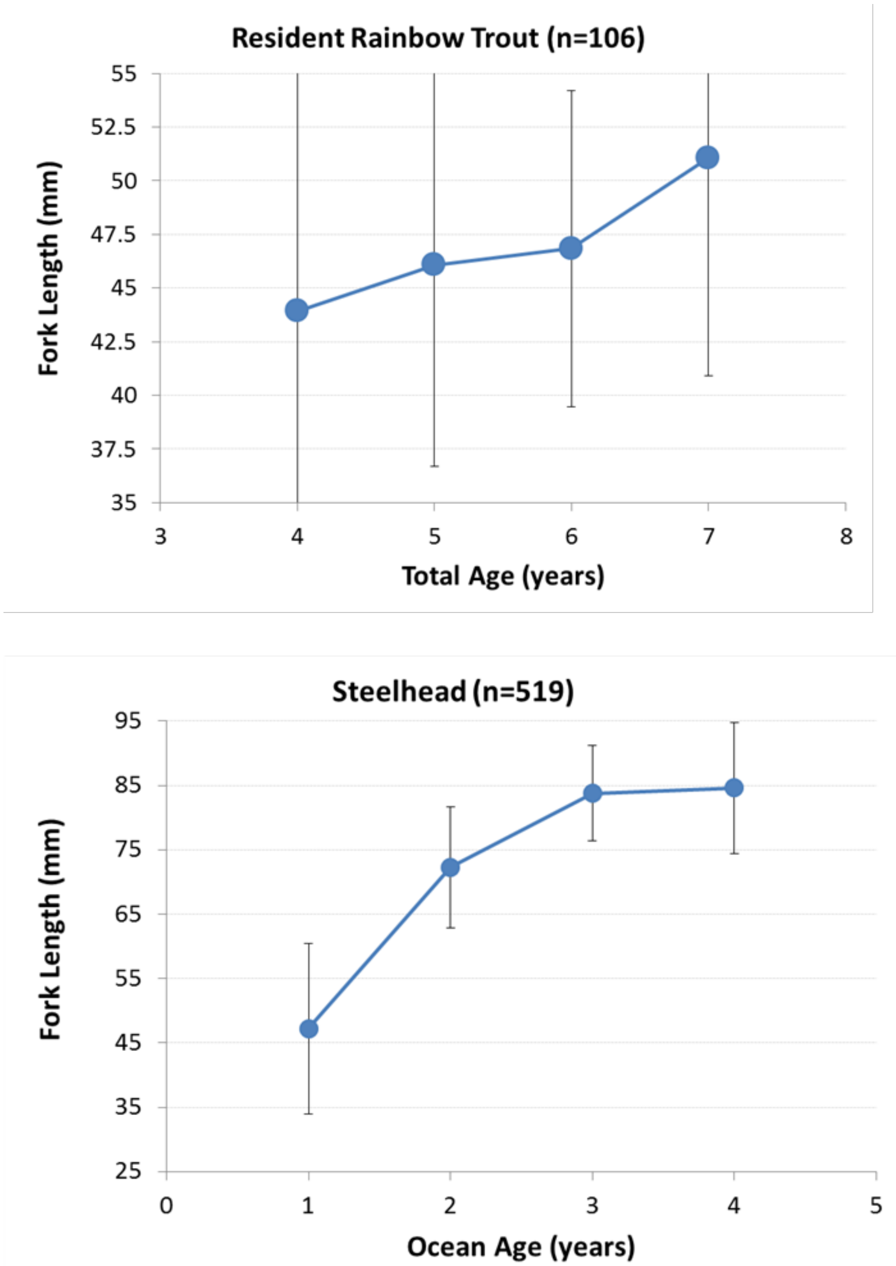

Figure 4.12. Mean fork length of resident rainbow trout (top) and steelhead (bottom) captured in the Cheakamus River between 2000 and 2017. Error bars show 1 standard deviation around the mean. 'n' at the top of each plots denotes the sample size (# of fish where total age could be determined).

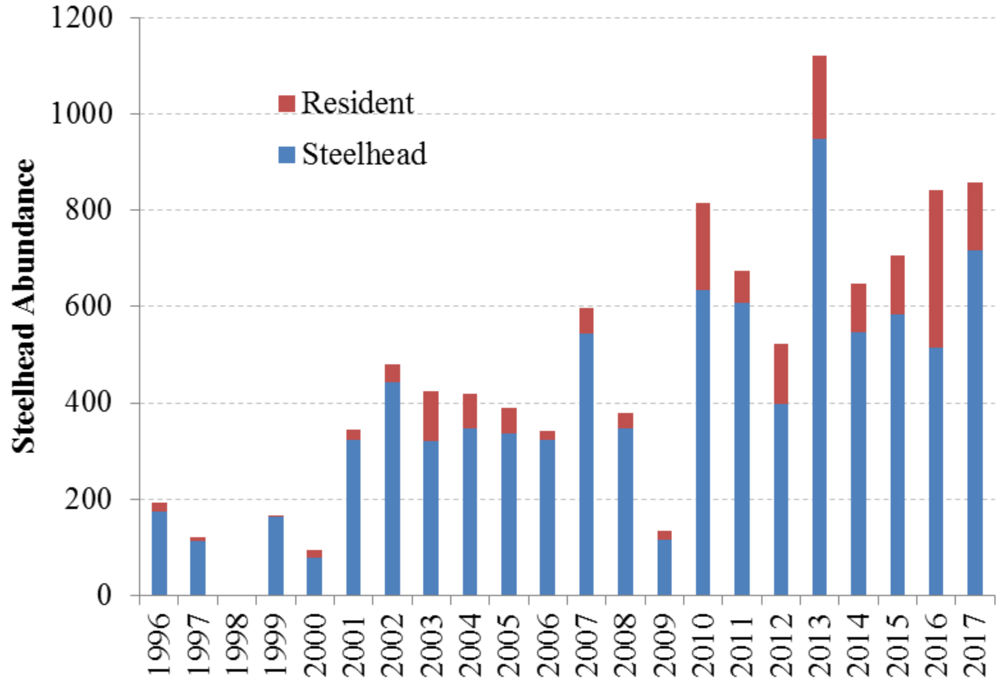

Figure 4.13. Abundance of returning steelhead (escapement) and resident rainbow trout in the Cheakamus River. The total height of the bar is the combined abundance of Steelhead and resident trout.

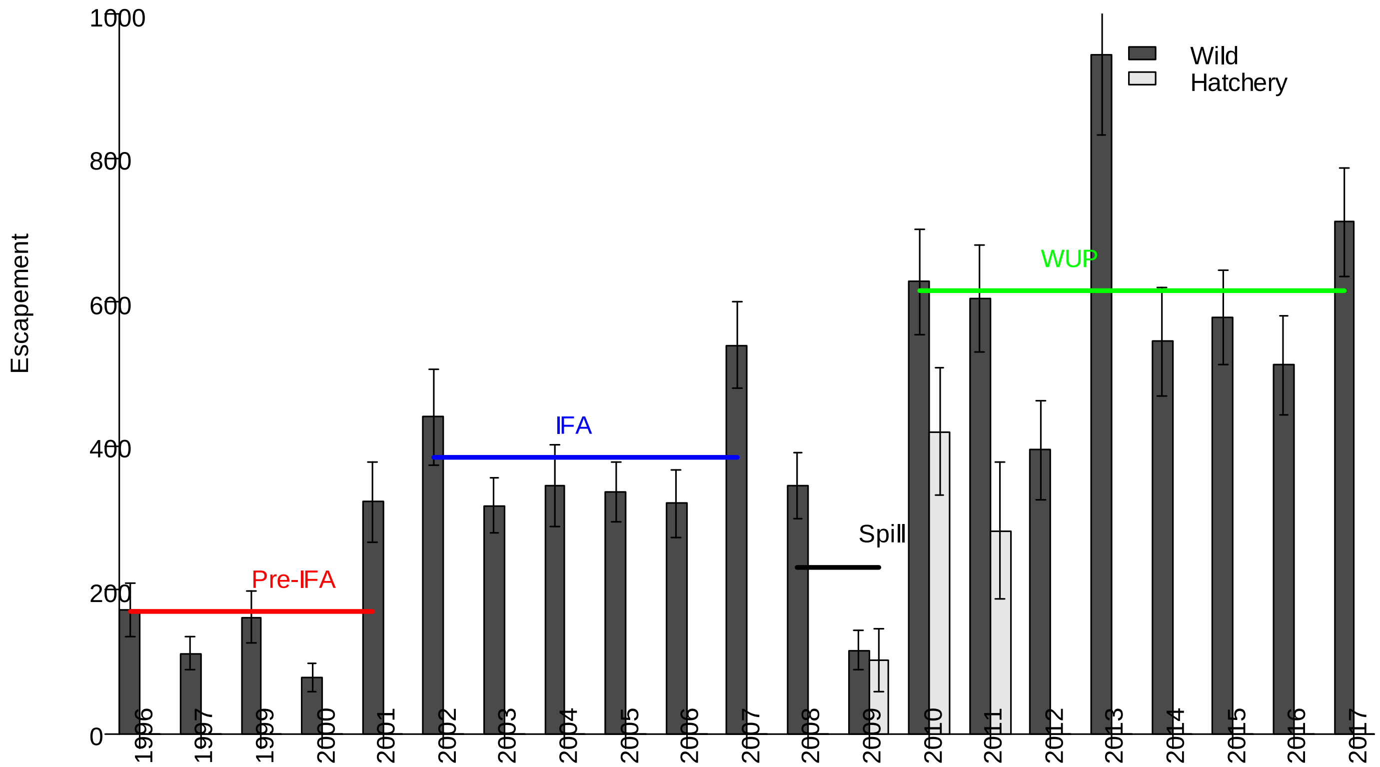

Figure 5.1. The Steelhead escapement trend in the Cheakamus River, 1996-2017. The height of the bars and error bars show the most likely escapement estimates and 80% credible intervals, respectively. The colored horizontal lines show the average escapement for years where the returns had reared as juveniles 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 sodal spill.

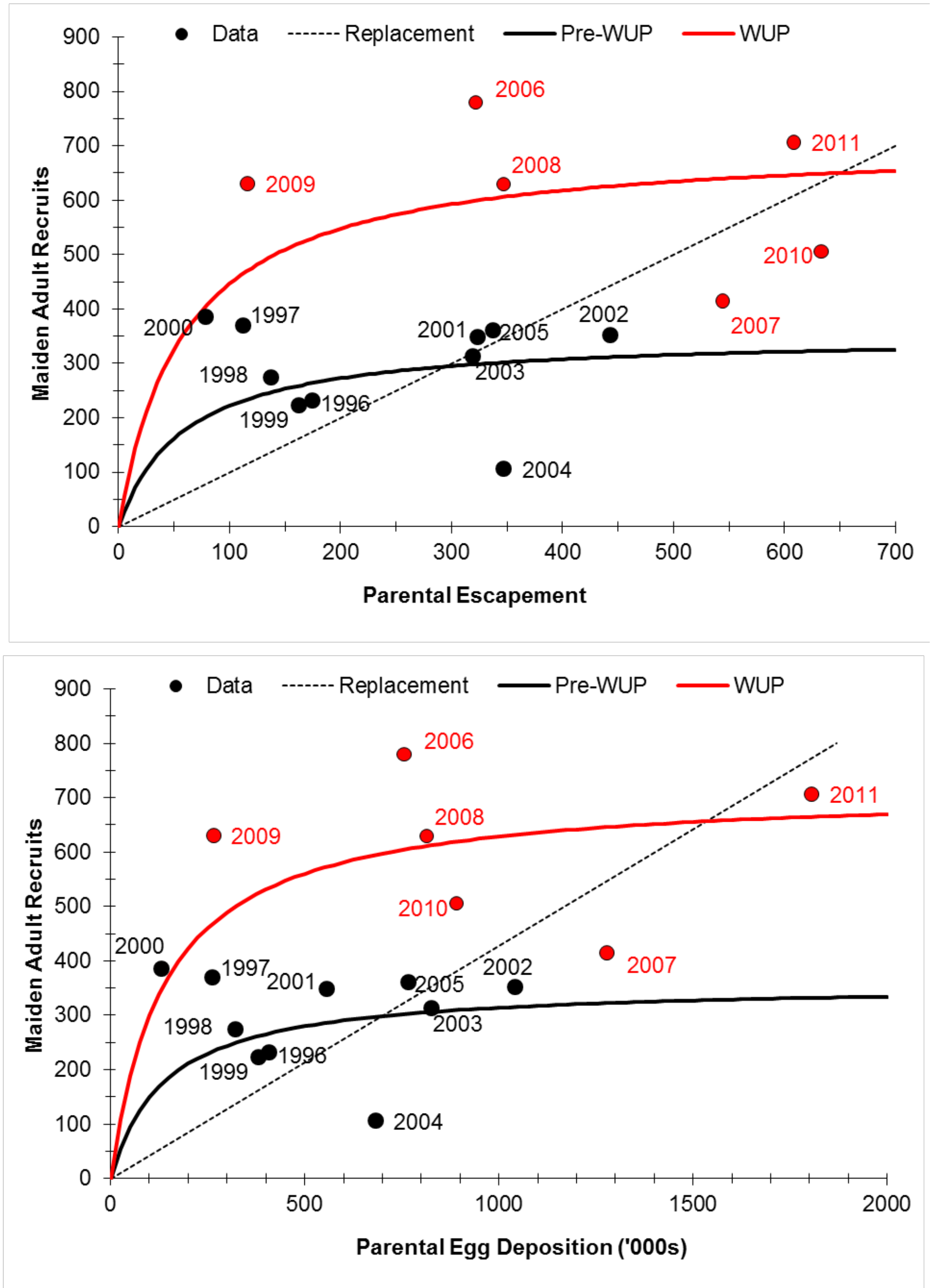

Figure 5.2. The relationship between the number of Steelhead spawners (top) and egg deposition (bottom) and the resulting maiden adult returns (total returns less repeat spawners) to the Cheakamus River. Recruitments based on fish that reared in the river under WUP flows are highlighted in red. The year 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 constrained Beverton-Holt stock-recruit relationships under pre-WUP and WUP flows.

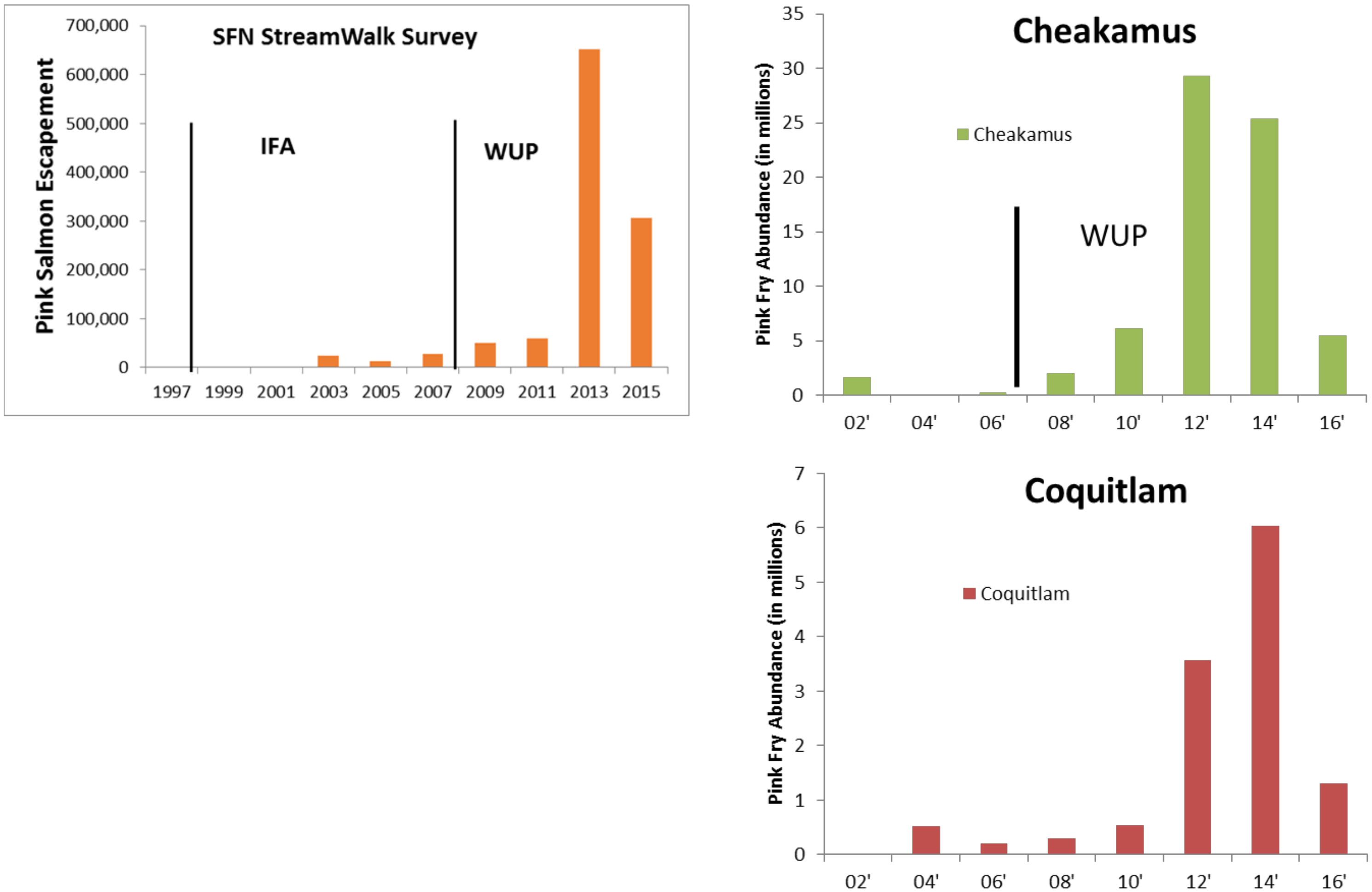

Figure 5.3. Trends of pink salmon returns. The top-left panel shows the index of pink salmon returns in the Squamish watershed as a whole from stream walks conducted by the Squamish First Nation (SFN). Plots on the right show run size of pink salmon fry which occur in the calendar year following the return year.

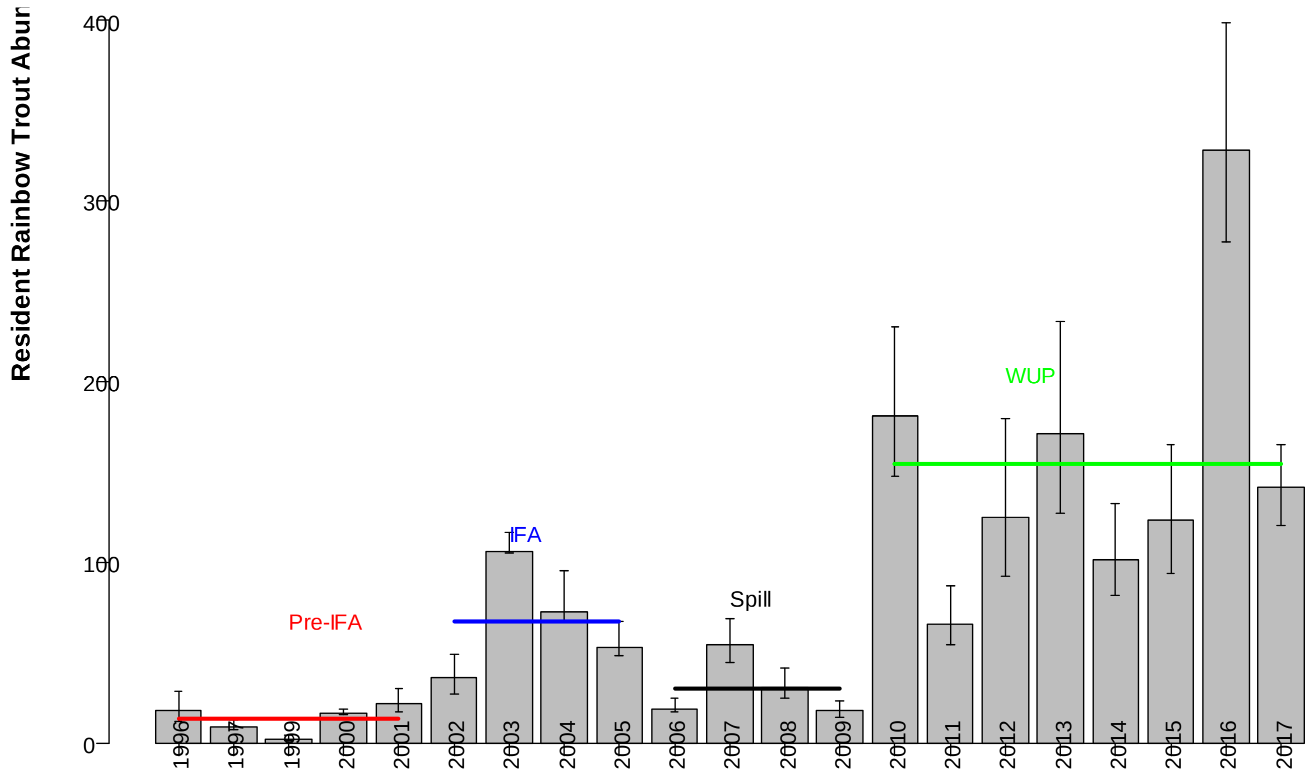

Figure 5.4. Resident rainbow trout abundance estimates in the Cheakamus River upstream of the Cheekye River confluence. The height of bars and error bars represent the mean and 95% credible intervals for annual estimates. Assuming a minimum age of 4 yrs, old, the average abundance for resident trout 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.

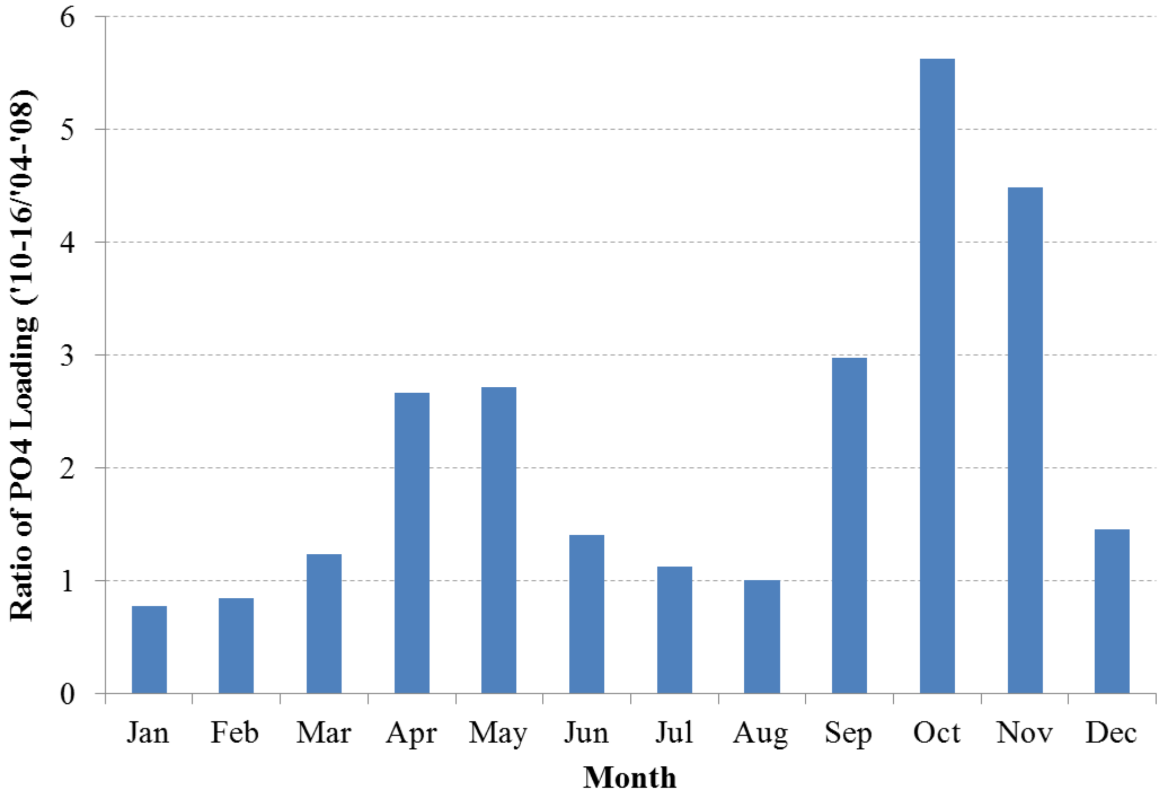

Figure 5.5. Change in phospate loading from the Whistler sewage treatment plant. The height of the bars is the ratio of the average phosphate loading (kg/day) by month from 2010-2016 to the average from 2004-2008.

a) Cheakamus River

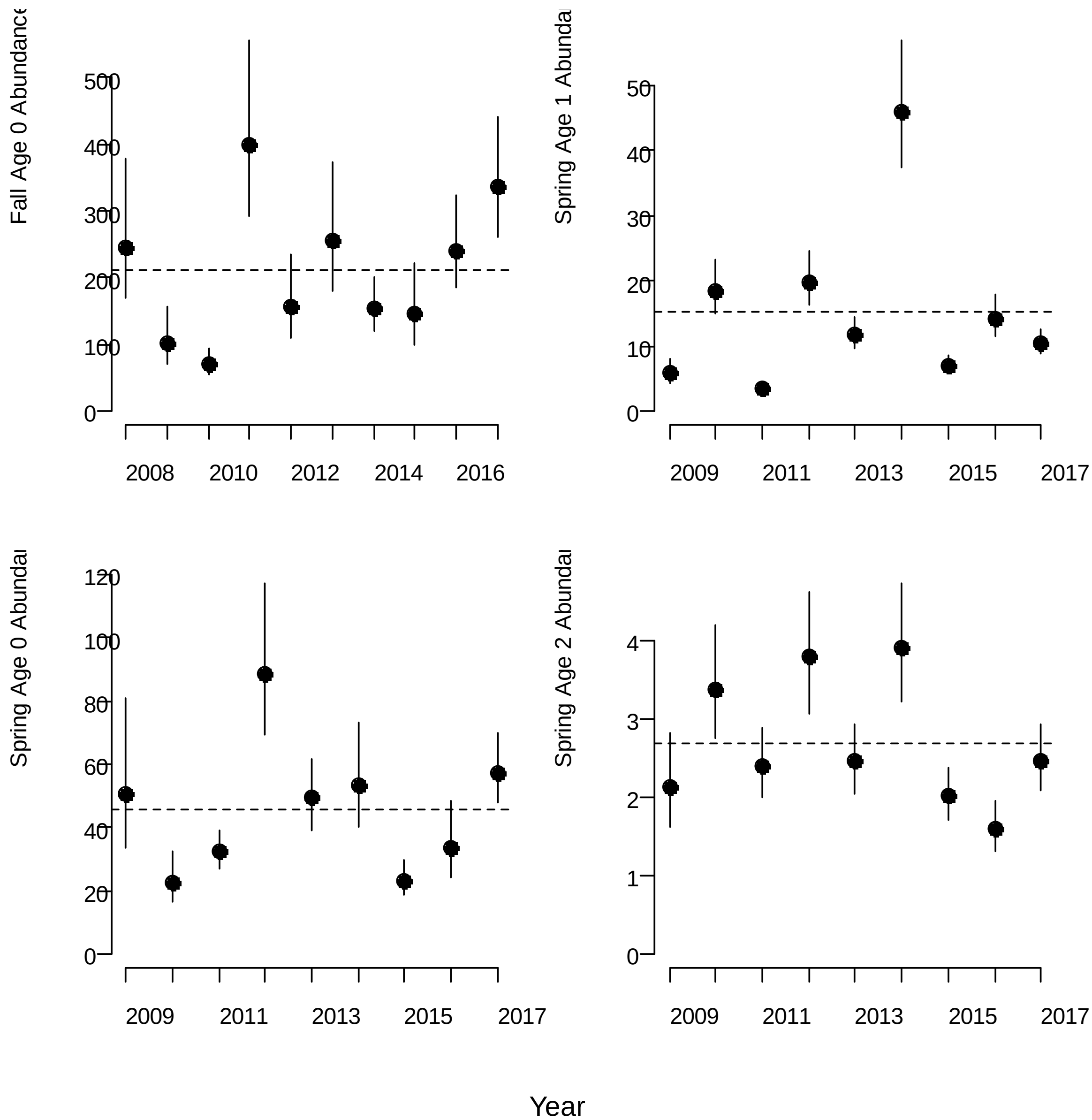

Figure 6.1. Juvenile steelhead abundance estimates in Cheakamus (a) and Brohm (b) Rivers. The height of bars and error bars represent median values and the 95% credible intervals, respectively. The dashed horizontal lines show the across-year averages.

b) Brohm River

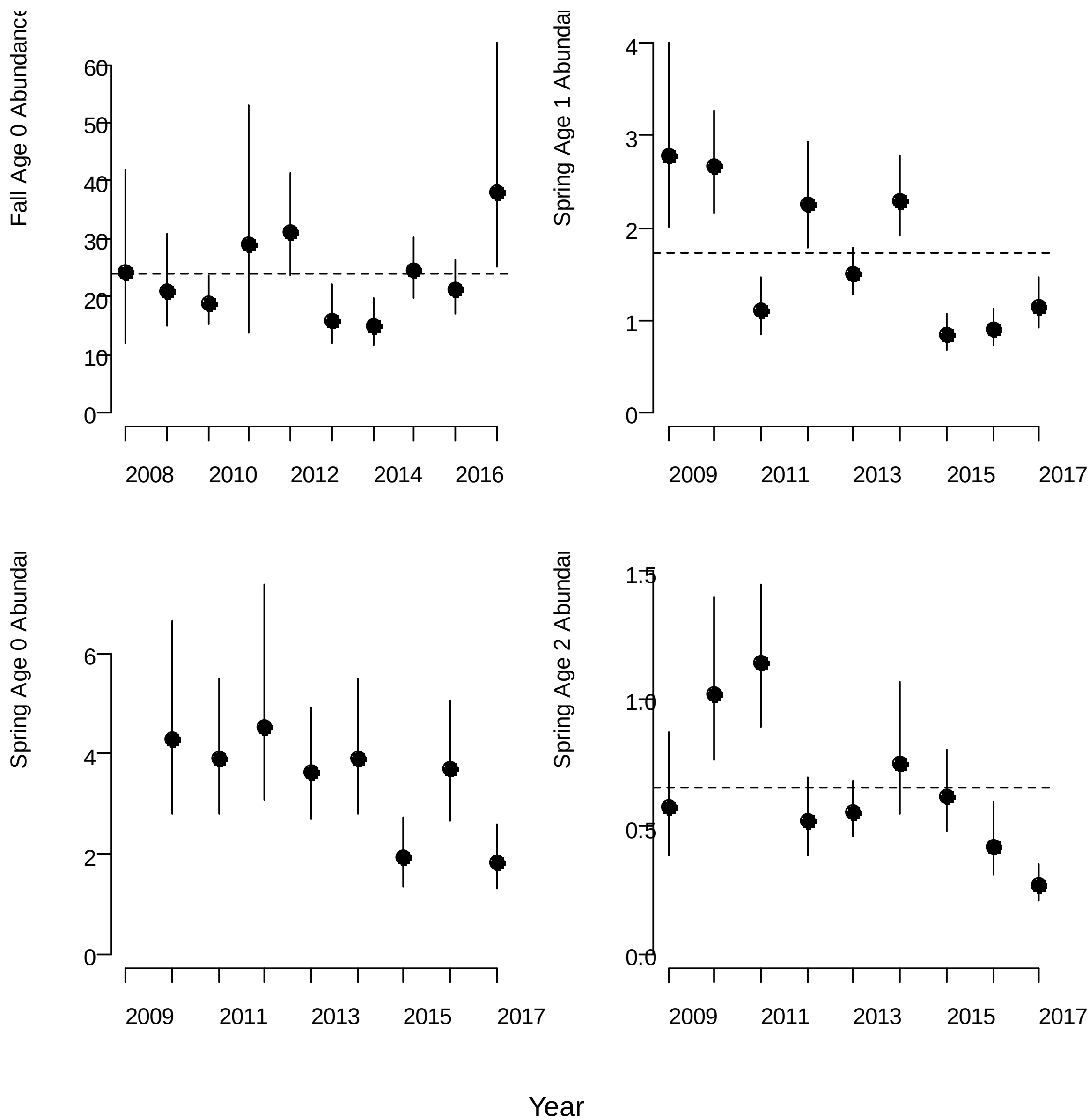

Figure 6.1. Con’t.

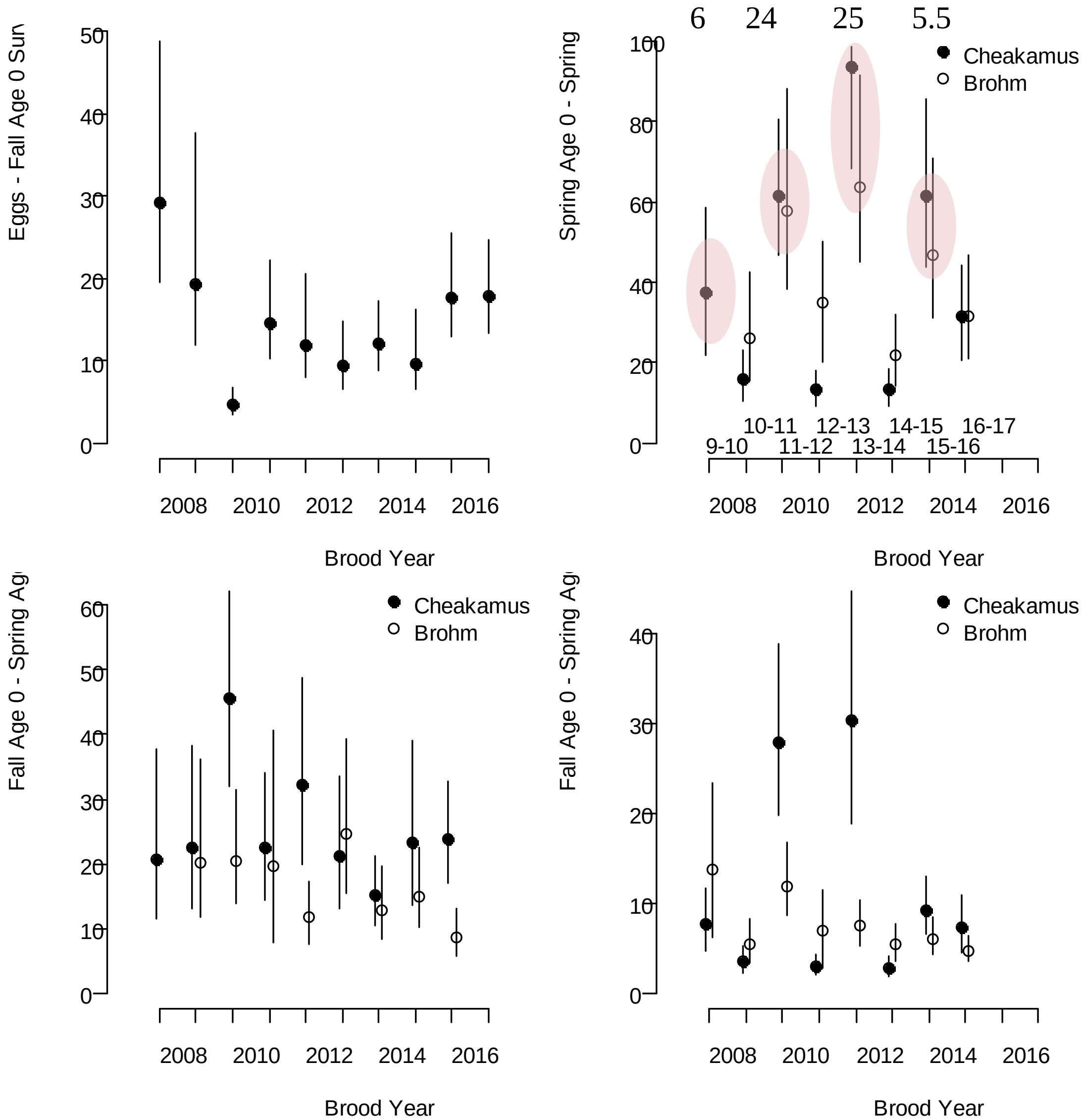

Figure 6.2. Survival for different Steelhead life stages in the Cheakamus and Brohm Rivers by brood year. Points and vertical lines denote means and 95% credible intervals, respectively. Numbers at the top of the top-right panel are estimates of the number of outmigrating pink salmon fry (in millions)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

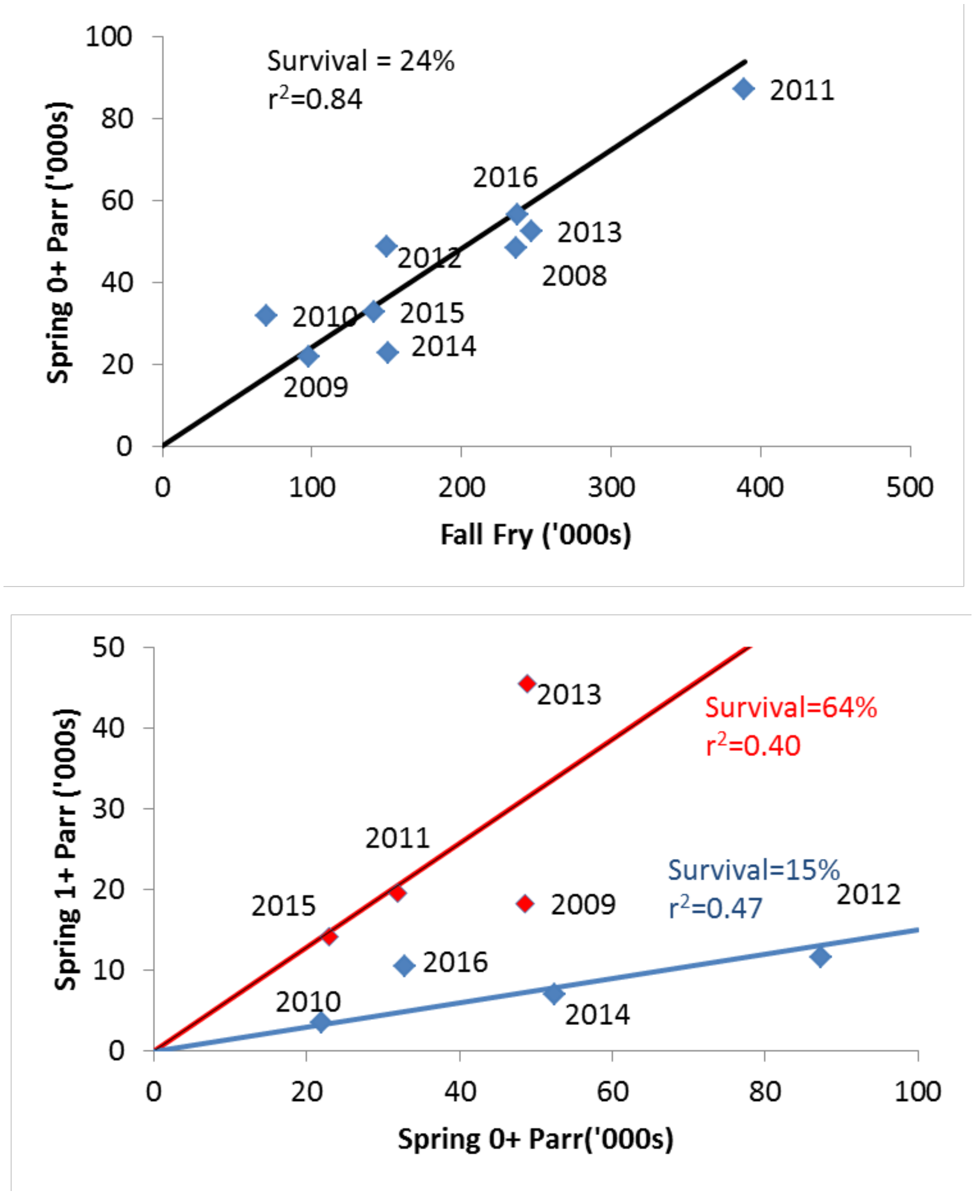

Figure 6.3. Relationship between Steelhead fall fry abundance and 0+ parr abundance the following spring (top), and between 0+ parr abundance in the spring and 1+ abundance the following spring (bottom). Separate relationships in even and odd years are used to highlight differences in survival in odd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

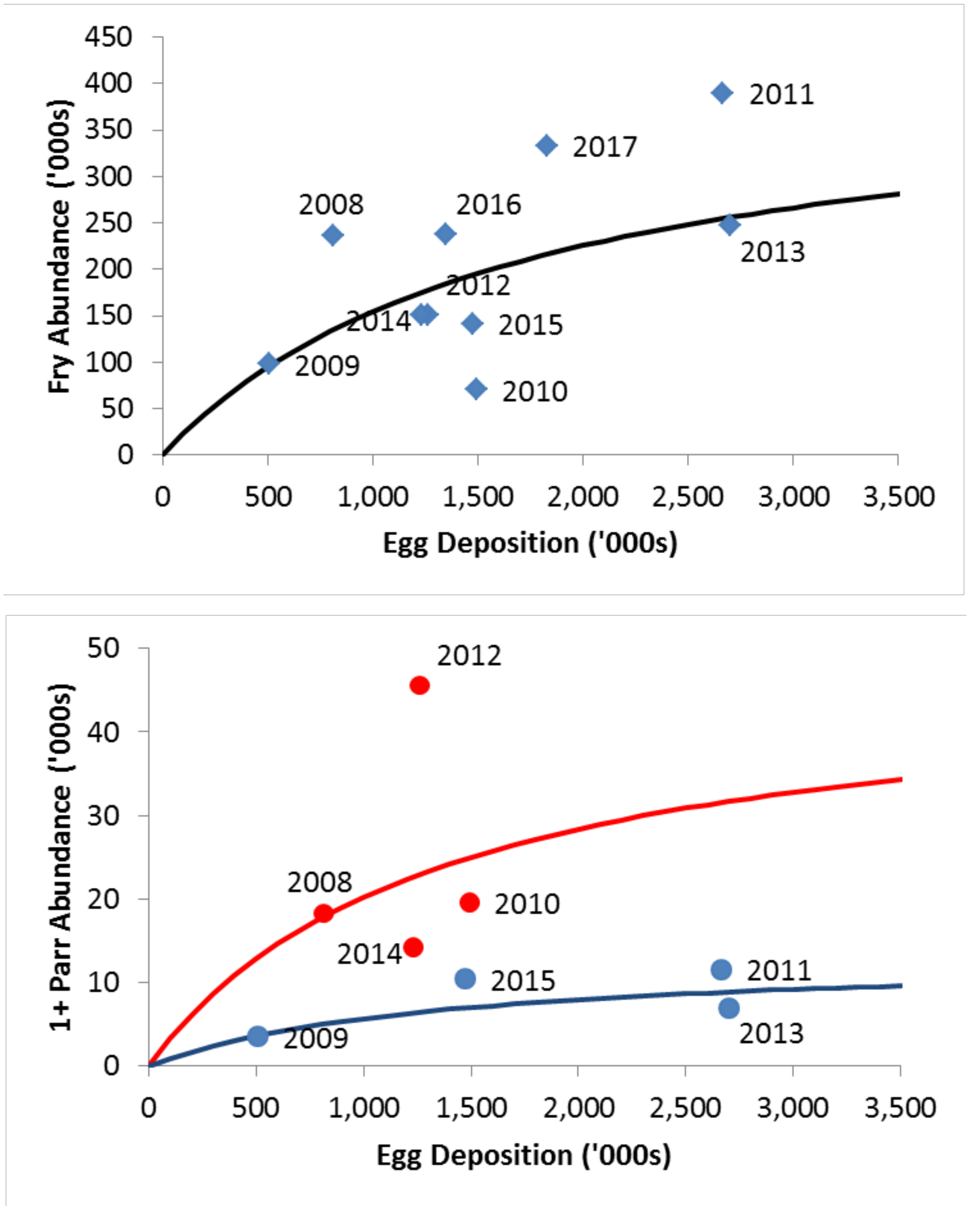

Figure 6.4. Relationship between Steelhead egg deposition in the Cheakamus River and production of fry in the fall (~ 6 months after spawning, top) and 1+ parr in the spring (~ 2 years after spawning, bottom). Text beside each point denotes the brood year. Lines show the best-fit Beverton-Holt stock-recruitment relationships. The red and blue lines and points in the bottom panel identify separate relationships for even and odd brood year. The difference between these egg-parr relationships shows the positive effect of high pink salmon returns on annual parr survival rates, which increases production for even year cohorts.

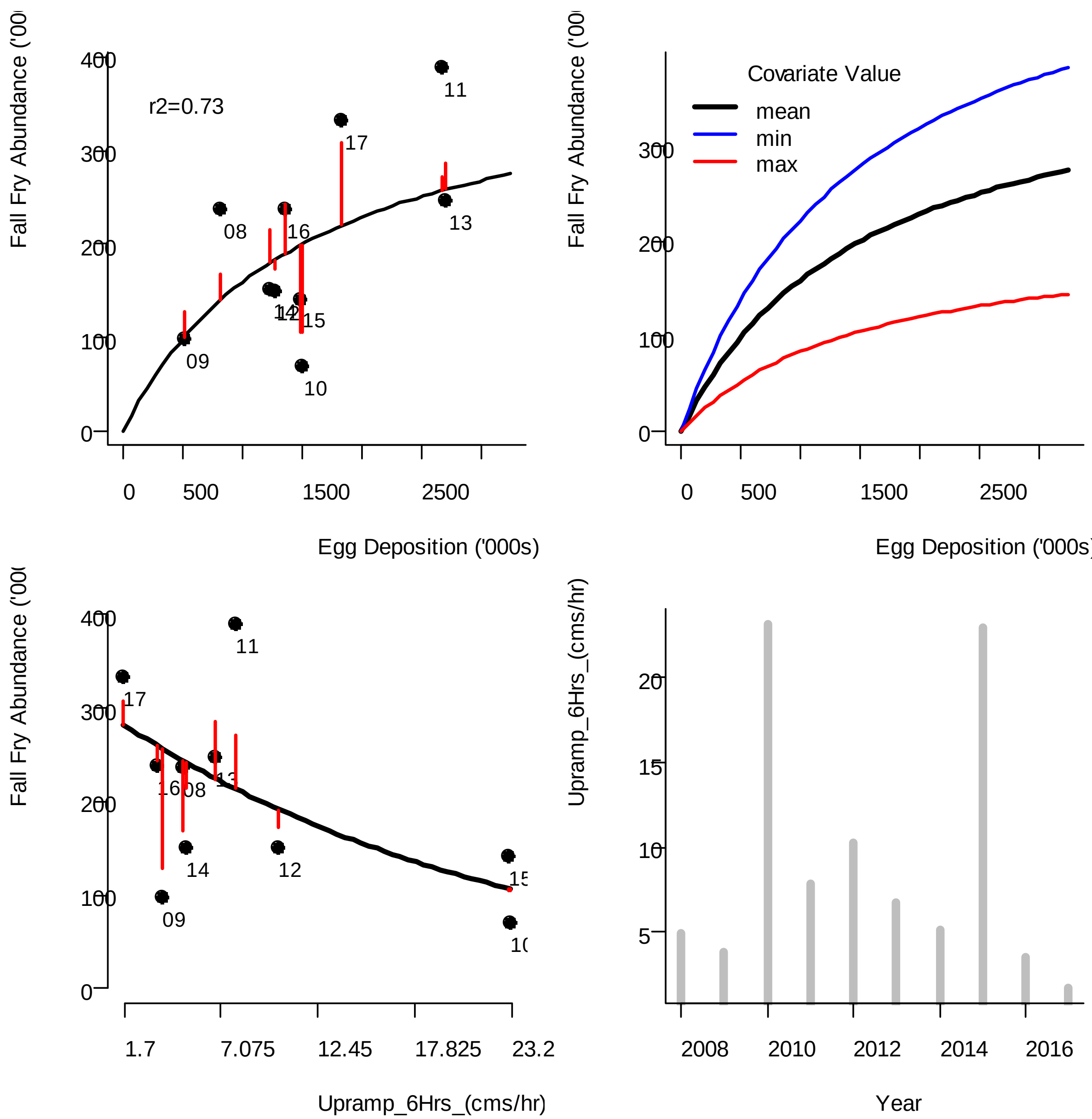

Figure 6.5. Fit of a Beverton-Holt model to Steelhead egg deposition – fall fry abundance in the Cheakamus River with a flow covariate effect. This model shifts the stock-recruitment curve each year based on the maximum increase in discharge over 6. The upper-left panel shows the data (points) with text 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 bottom-left panel shows the relationship between the covariate and recruitment at the average egg deposition over years (black line) and predicted deviations at each annual level of egg deposition (vertical red lines). Also shown is the effect of the covariate on the recruitment curve based on the minimum, mean, and maximum covariate values across years (top right) and the annual covariate values (bottom right).

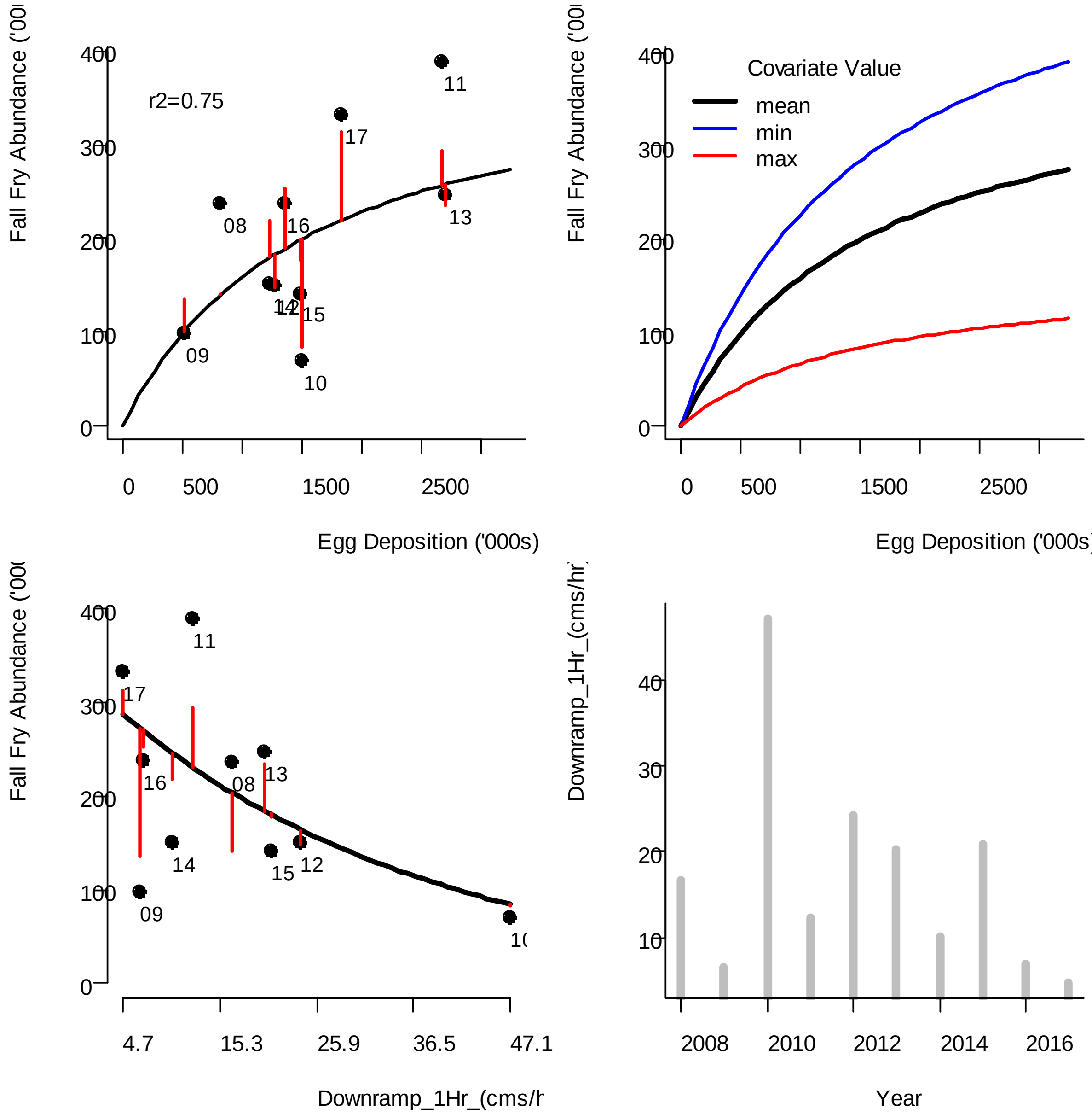

Figure 6.6. Fit of a Beverton-Holt model to Steelhead egg deposition – fall fry abundance in the Cheakamus River. This model shifts the stock-recruitment curve each year based on the maximum decrease in discharge over 1 hour between July 1 and the first date of fry sampling in each year which occurs in early September. See caption for Figure 6.5 for additional details.

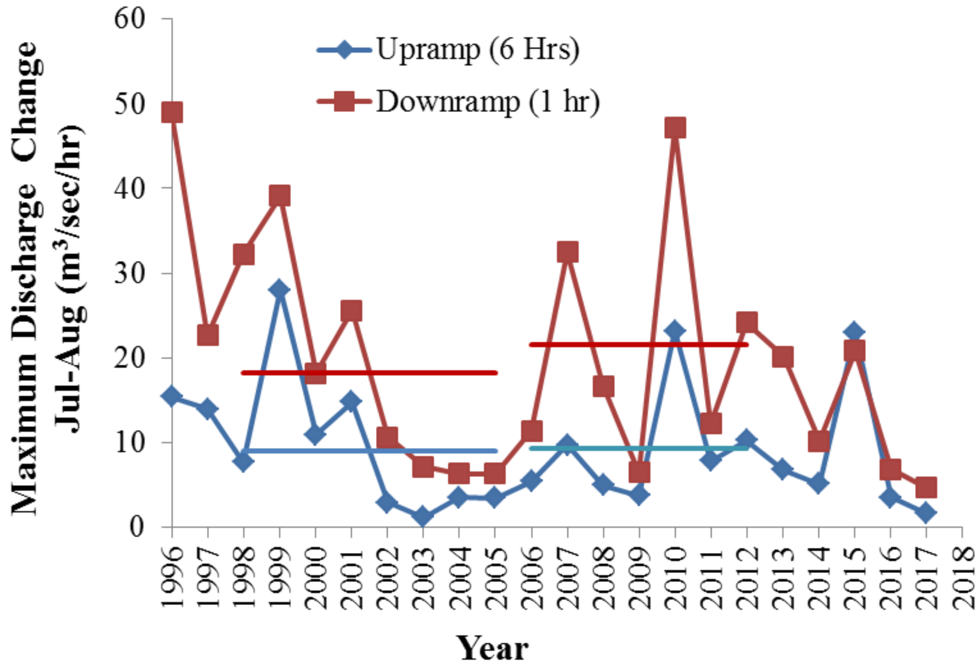

Figure 6.7. Maximum change in discharge between July 1 and the first date of fry sampling in each year which occurs in early September (average end date used for all years prior to 2008). Horizontal lines show the average values during Instream Flow Agreement (IFA) and Water Use Planning (WUP) periods. The average for the WUP period was limited to years where brood year freshwater production could be assessed from escapement returning 5 years later (see text for additional discussion).

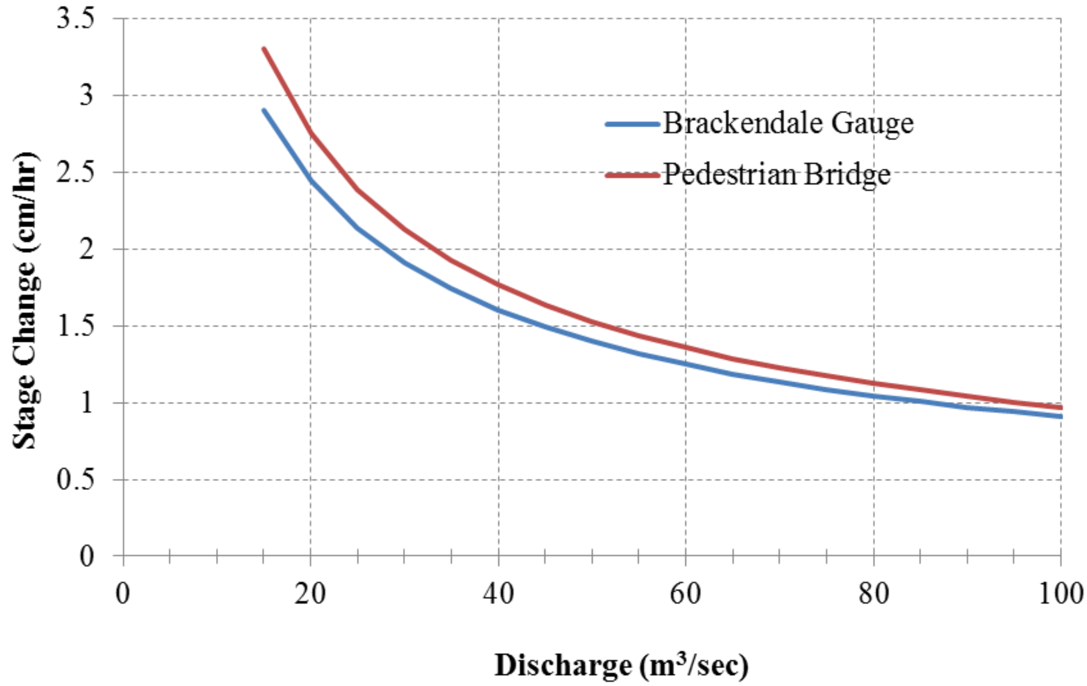

Figure 6.8. Predicted rate of stage change at two locations in the Cheakamus River (see Fig. 1.1) at a range of discharges given a ramping rate at Daisy lake Dam of 1.5 $m^3 \cdot s^{-1} \cdot hr^{-1}$.

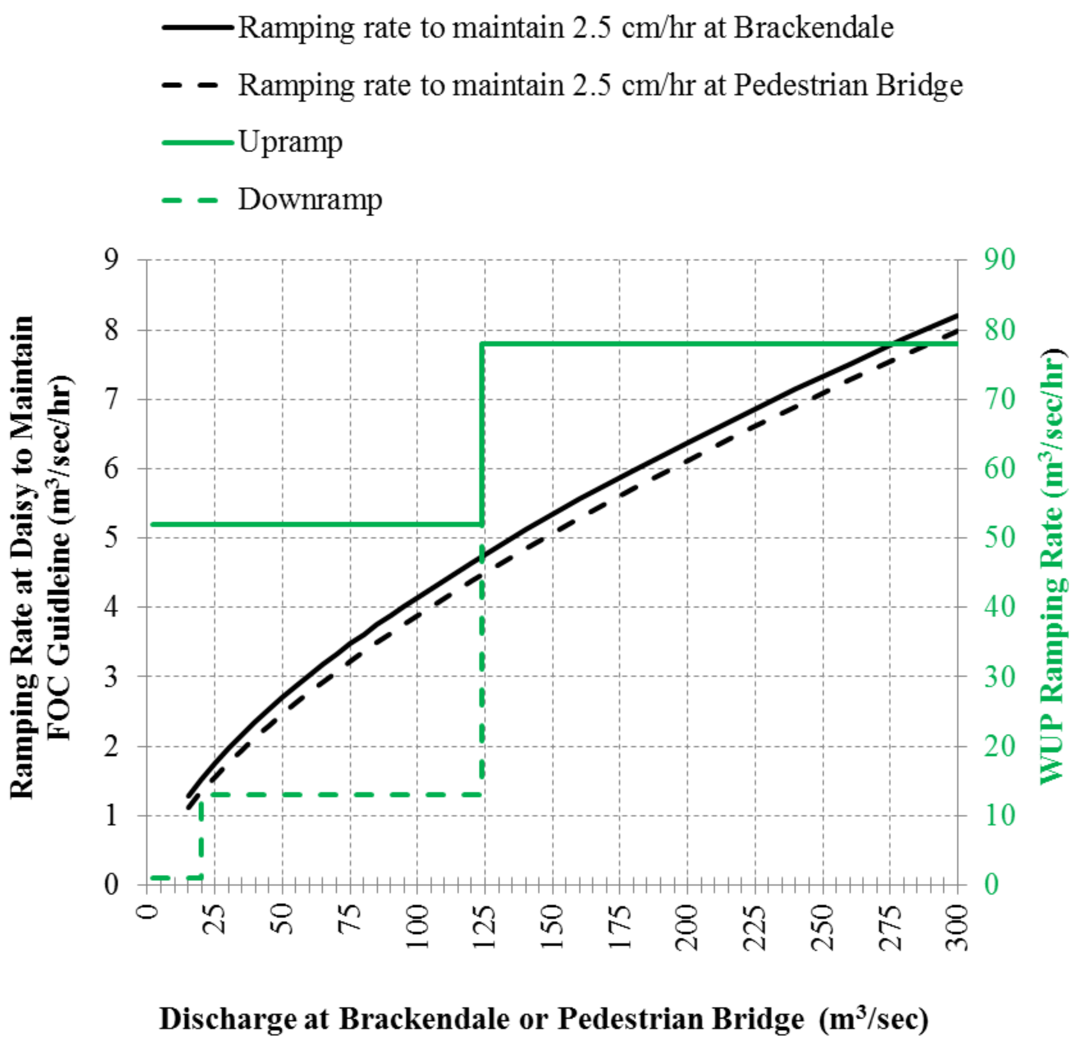

Figure 6.9. Ramping rates at Daisy Lake Dam to maintain a stage change at the Brackendale gauge and Pedestrian Bridge of 2.5 $cm \cdot hr^{-1}$ for a range of discharges at the Brackendale gauge (solid black line) or the Pedestrian Bridge (dashed black line). Ramping rate rule curves are: RR=0.239·Q^0.62 (Brackendale) and RR=0.1904·Q^0.655 (Pedestrian), where RR is ramping rate at Daisy Lake Dam ($m^3 \cdot s^{-1}$) and Q is discharge ($m^3 \cdot s^{-1}$) at Brackendale or the Pedestrian Bridge, respectively. The right-hand axis shows the ramping rates in the current WUP order assuming that 50% of the discharge at the Brackendale gauge or the Pedestrian Bridge is from local inflows and 50% is from releases from Daisy Lake Dam (which determine Daisy Lake Dam ramping rates, see Table 2.1b). Note the scale of the right-hand axis is 10-fold greater than the left-hand axis.