Cheakamus River Adaptive Stranding Protocol Monitoring Results

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Final Report

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1 Introduction

1.1 Report Objectives

This report summarizes results from the fifteen monitored rampdowns (September 2018 to September 2019) following the initiation of the Cheakamus Adaptive Stranding Protocol (CASP 2018) as well as incorporating the results from the August 20, 2018 rampdown (Korman *et al.* 2018). We then use these findings to assess whether there is support for the hypothesized relationship between stranding risk and several risk factors, and the effectiveness of mitigation to lower risk. We also assess the effectiveness of monitoring methods to meet the objectives of the Cheakamus River Adaptive Stranding Protocol (CASP 2018). We provide a summary of key findings to inform decisions related to stranding mitigation. For this, risk is quantified using the approaches described in the CASP (2018). These are based on relative risk and not population level risk. In response to comments by stakeholder and agency reviews, we do not categorize risk as low, medium or high in this report, which imply population level effects. We instead report the numeric risk levels or use the terms 'lower' and 'higher' for comparisons.

Instream Fisheries Research (IFR) carried out all rampdown monitoring and reporting as per the approach in the CASP (2018). Monitoring and reporting was funded through a contract with BC Hydro Environment, which meant that assessment work was limited to only high priority activities.

This report does not include review of CASP objectives or its approach. However, we do discuss the effectiveness of the present monitoring methods to assess relative stranding risk given the current approach as described in the CASP (2018). Anticipating changes to both the CASP approach, including how stranding risk is quantified and monitoring methods, we have not included recommended changes to the existing methods as they would be irrelevant if the approach changes.

1.2 Cheakamus River Hydroelectric Facility Water Use Plan

The Cheakamus River hydroelectric facility completed in 1957 is comprised of Daisy Lake Reservoir and Dam, an 11km tunnel through Cloudburst Mountain connecting the reservoir, via

Shadow Lake, to the penstocks and Cheakamus Generating Station on the Squamish River. Thus, inflows to Daisy Lake Reservoir are either diverted to the Cheakamus Generating Station, and bypass the lower 25km of the Cheakamus River, or released from the dam back into the Cheakamus River. The Water Use Plan (WUP) implemented in 2006 forms the basis for dam operations with the objective to maximize the productivity of wild fish populations while balancing social and economic values. The WUP mandates minimum flows must be maintained at two locations along the Cheakamus River. Minimum discharge from Daisy Lake Dam is set at:

- 3 m^3/s November 1 to December 31;
- 5 m^3/s January 1 to March 31;
- 7 m^3/s April 1 to October 31;

and the minimum flows at the Brackendale gauge (WSC 08GA043) are set at:

- 15 m^3/s November 1 to March 31;
- 20 m^3/s April 1 to June 30;
- 38 m^3/s July 1 to August 15;
- 20 m^3/s August 16 to August 31, unless directed by Comptroller to maintain flows of 38 m^3/s for recreation;
- 20 m^3/s September 1 to October 31.

Operations at the Daisy Lake Dam require changing the flow to the Cheakamus River to balance between minimum flows to the Cheakamus River, inflows to Daisy Lake Reservoir, reservoir storage capacity and diversion to the Cheakamus Generating Station on the Squamish River. This results in two scheduled rampdowns in response to the reduction in required minimum flow that occur between August 16 and September 1 (38-20 m^3/s) and November 1 (20-15 m^3/s). We refer to these events as 'scheduled' rampdowns. There are also a number of 'unscheduled' rampdowns. Under the current WUP, unscheduled rampdowns typically occur when inflows to Daisy Lake exceed its storage capacity and discharge capacity of the Cheakamus generating station on the Squamish River (65 m^3/s). Spilling above the minimum is most common during the spring freshet, due to snow melt, in the fall, when winter rain events, and when dam or powerhouse maintenance decreases storage or diversion.

The WUP and Water Use License specify the maximum ramp rates for increase and decrease based on the flow out of Daisy Lake Dam:

- If total discharge from Daisy Lake Dam is less than $10m^3/s$
    - the maximum rate of increase is $13m^3/s$ per 15 minutes
    - the maximum rate of decrease is 1.0 m^3/s per 60 minutes
- If total discharge from Daisy Lake Dam is 10-62 m^3/s
    - the maximum rate of increase is 13 m^3/s per 15 minutes
    - the maximum rate of decrease is 13 m^3/s per 60 minutes
- If total discharge from Daisy Lake Dam is greater than $62m^3/s$
    - the maximum rate of increase is 13 m^3/s per 10 minutes
    - the maximum rate of decrease is 13 m^3/s per 10 minutes

Table 1 lists the stage-based rampdown rates that correspond to the discharged-based rates for a range of flow reductions at the Brackendale Water Survey of Canada station on the Cheakamus River (WSC 08GA043). At a constant discharge based rampdown rate (m^3/s), rampdown rates based on stage change (cm/h) increase as flows decrease. Stage-based rates will vary a small amount between periods with different rating curves since the relationship between discharge and stage are based on rating curves that are periodically updated.

1.3 Effect of ramping on fish stranding

A primary difference between regulated and unregulated rivers is the potential for increased frequency, rate and magnitude of flow change compared with a natural hydrograph. As a consequence, fish in regulated rivers are subjected to flow changes during ramping events that exceed the rate of change of the river prior to regulation. Flow regulation can also lead to potentially consequential divergences from a natural hydrograph: long periods of stable flows followed by a rapid decrease or replacing a short high intensity peak flow event with a lower magnitude increase and decrease but with a longer period of stable high flows.

Fish stranding occurs when the rate of a flow reduction (rampdown) exceeds the ability of fish to respond to the change restricting them to poor habitat (Nagrodski *et al*. 2012). In the context of the Cheakamus River, poor habitat includes dewatered bars, where mortality occurs almost immediately, and isolated pools, where the outcome is far more variable. The likelihood of stranding depends on a number of factors that are understood to varying degrees. They include variables in which fish differ: species, life-stage, fish size and habitat use; ways that the river environment differ: season, water temperature, time of day, river morphology and substrate; (Jones and Stokes, 2003, Nagrodski *et al*. 2012); as well as flow characteristics prior to and during a rampdown: rate of flow reduction (ramp rate), magnitude of reduction, duration prior to rampdown the habitat was wetted (wetted history) (Bradford *et al.* 1995, Bradford, 1997, Halleraker *et al.* 2003).

1.4 Development of Cheakamus River Adaptive Stranding Protocol

The WUP included monitoring programs to evaluate the effect of rampdowns on juvenile and adult fish below the Cheakamus Generating Station in the tailrace (MON-4), and in the 2.5km below Daisy Lake Dam (MON-5), as well as a desktop risk assessment of potential fish stranding in the Squamish River below Cheakamus Generation Station (MON-3). No monitoring program was included in the WUP (BC Hydro, 2005) to directly address the impact of rampdowns on fish in the 17km anadromous section of the Cheakamus River, which supports more diverse and larger fish populations than either of the areas that were included in monitoring, Rampdown rates were a concern for the Consultative Committee Report (CC Report 2002). The Consultative Committee recommended that transitions between seasonally varying flows be gradual enough to prevent stranding, however, there is no record what rates would meet this objective. WUP ramp rates were based on a different approach than DFO ramping guidelines. WUP up and down ramp rates were based on discharge level and remain the same throughout the year. DFO guidelines, based on stage change (vertical drop in river level), vary by time of day and season / life-stage (Table 2). Generally, the lowest rampdown rates are similar for both WUP (Table 1) and DFO guidelines, assuming night-only ramping, which was largely the case for the 15 ramps monitored since August 2018. In contrast, the maximum WUP rates can be up to five-fold higher than the maximum DFO guidelines depending on how the WUP rates are applied (the highest WUP rates are based on change per 15 minutes not per hour).

Findings from the MON-3 synthesis report (Korman and Schick 2018) raised the possibility of population level impacts on juvenile Steelhead Trout from rampdowns. This, combined with documented fish stranding by anglers during the summer of 2018 and continued concern by members of the Consultative Committee highlighted the broad concern about the present rampdown rates. BC Hydro responded to this concern by initiating stranding monitoring for the scheduled mid-August 2018 reduction in minimum flows from 38-20m^3/s. Results from this rampdown provided further support that rampdowns in the anadromous section of the Cheakamus could be having substantial population level impacts (Korman *et al.* 2018).

As a result, BC Hydro developed the Draft Cheakamus River Adaptive Stranding Protocol (CASP 2018). The intention of the CASP is to address fish stranding in the anadromous portion of the Cheakamus River under an adaptive management framework, as well as address information gaps identified in the WUP monitoring program.

The objectives of the CASP (2018) are:

1. *Develop a fish stranding risk field assessment approach to inform stranding risk associated with key variables including magnitude/rate of flow reduction, season, and river discharge level;*
2. *Test hypotheses with respect to how key variables influence relative fish stranding risk on the lower Cheakamus River;*
3. *Define clear operations decision criteria that will be used to evaluate when mitigation measures are required; and*
4. *Identify potential mitigation measures and mitigation option selection criteria.*

The two primary study hypotheses of the CASP (2018) are :

1. *Fish stranding during flow rampdowns are a function of:*
    a. *Season – stranding risk is highest when sensitive life-stages are present (newly emerged and early rearing fish) and/or use of habitats at highest risk of stranding is high.*
    b. *River discharge – stranding risk increases as rampdowns occur at lower discharge level.*
    c. *Rampdown magnitude – stranding risk increases as the magnitude increases.*
    d. *Rampdown rate – stranding risk increases as the rate of change of either discharge or stage increases.*

Several other risk factors are also listed in the CASP (2018) for consideration in the assessment. While there is some overlap with the primary hypotheses, these include: fish species, life-stage, and density present, time of day, wetted history, site specific channel morphology and substrate.

2. *Targeted mitigation measures focused on factors influencing high-risk flow rampdowns that will reduce relative risk of fish stranding on the lower Cheakamus River.*

CASP (2018) describes the approach of the protocol in detail. We include only a summary here for reference and to note where the approach has changed during the monitoring period.

1. *Assign flow ramp down risk categories based on hypothesised fish stranding risk factors*

   Table 3 lists the hypothesised risk levels in relation to discharge range and season. <u>Note</u>: that these hypothesized risk levels imply population level effects.

2. *Assess relative stranding risk across the range of ramp down types*

   CASP measures relative stranding risk in two ways based on how fish are stranded. For bar stranding, it is the density of stranded fish in high risk habitats. This is reported as the number of fish stranded per square meter of dewatered shoreline and grouped into three density ranges (Table 4). However, the three risk levels are no longer referred to as low, moderate and high. For pool stranding, risk is no longer based on the number of pools classified as low, moderate or high risk (Table 5), instead we report the total number of fish observed and by depth class.

   <u>Note:</u> These changes were initiated to remove any implied interpretation that the reported risk levels were indicative of population level risks.

   The initial approach for including a range of rampdown types was to first establish baseline risk levels using WUP ramp rates and then create contrast in ramp types by mitigating some rampdowns. This approach was changed following the May 17, 2019 rampdown with all rampdowns being mitigated. The mitigation goal being maintaining bar stranding risk at what were considered at that time as 'low' levels. In addition, all rampdowns after this date would be monitored.

3. *Identify appropriate mitigation measures reflective of ramp down type*

4. *Develop decision criteria for when to implement mitigation measures*

   This protocol was also shifted after May 17, 2019 from a criteria based on not mitigating unless needed to mitigating all ramps unless mitigation i.e. reduced ramp rates were expected to have no impact on risk.

5. *Monitor effectiveness of mitigation measures in reducing relative stranding risk*

   The approach to assess mitigation effectiveness also shifted May 17, 2019. Initially, mitigation was considered effective if it reduced risk compared with the WUP baseline. After May 17th, effective mitigation would either maintain or further reduce risk.

6. *Adapt mitigation measures and decision criteria annually based on monitoring results*.

An implied assumption in the CASP (2018) is that if stranding can be reduced to an acceptable level in the highest risk habitats then it will also be adequately reduced for habitats at lower risk of stranding fish.

2 Methods

2.1 Study design

The objective of monitoring is to estimate the relative stranding risk of juvenile fish in high risk habitats during rampdowns and then use that information to; a) test the hypothesized relationship between stranding risk and season, rampdown range, magnitude and rates, and b) evaluate the relative benefit of mitigation strategies for reducing stranding risk. Stranding risk is quantified uniquely for the two high risk habitats included in monitoring: low angle bars; and pools/side-channels isolated from the mainstem during a rampdown. For bar stranding, this is number of fish stranded per square meter of dewatered bar (fish/m^2) referred to as stranding density. Bar sampling sites were selected to represent habitats most likely to strand fish (e.g., low slope and cobble as the dominant substrate) and locations where we expected juvenile fish abundance to be the highest. Professional judgment as well as data collected over the years of CMS WUP monitoring studies informed site selection. Since measuring population level effects is not the primary objective of the CASP (2018), there was no intention that monitoring would quantify average or river-wide stranding. The stranding risk for isolated pools and side-channels is represented as the number of fish observed in all isolated areas within the study area.

The range, magnitude and rate of rampdowns were in many cases adjusted to provide contrast in specific flow variables while holding others constant but also had to meet operational objectives as well. Thus, variations in rampdown conditions were the product of operational decisions in combination with prescribed flow changes intended to support study objectives.

Monitoring focused on five target species: Steelhead trout (*Oncorhynchus mykiss*) and Chinook (*Oncorhynchus tshawytscha*), Coho (*Oncorhynchus. kisutch*), Chum (*Oncorhynchus keta*) and Pink (*Oncorhynchus gorbuscha*) salmon with a focus on juvenile life-stages.

2.2 Stranding survey methods

Stranding monitoring for each rampdown typically included crew activities the day prior, during and after each rampdown. One day prior to the rampdown, a crew measured and marked the boundaries of the bar survey sites. This also included marking the pre-ramp wetted edge of the mainstem channel within the site. For rampdowns in August of 2018 and 2019, electrofishing the mainstem shoreline at each bar stranding site the day prior to the rampdowns was used to estimate pre-rampdown abundance.

Bar and pool surveys occurred the day of the rampdown. The start time for surveys were aimed to coincide with the time flows stabilized after the rampdown at the upper most survey sites. The lag between the last gate movement and when flows stabilize at the upper-most sites was estimated at 1.5 hour at flows above 60m^3/s to 2 hours below this level. Surveys were competed in an upstream to downstream direction to minimize predation during the lag time between when the post-ramp target flow occurred at each site and when sampling occurred. In reality, this lag increased from less than an hour for the upper most sites to 3-6 hours for downstream sites. Considering this, we assumed that the impact of predation was greatest at the most downstream sites. No surveys to quantify the effect of predation or the assumption that we adequately controlled for predation were carried out. Mainstem abundance sampling for the majority of rampdowns took place one to several days after the rampdown. In cases when there were rampdowns on consecutive days, mainstem abundance sampling occurred after the completion of all rampdowns with the exception of the series of rampdown August 2019 when it occurred prior to each ramp.

2.2.1 Bar survey methods

Surveys of bar habitat quantified the density of stranded fish at each of six, and in one case seven high risk stranding sites in reaches 2-5 of the anadromous section of the Cheakamus River (Figure 1). We selected sites to represent the highest risk bar stranding sites at the discharge level for each rampdown. These include sites with one or more of the following characteristics: flat or very low angle profile, undulations or depressions, and cobble as dominant substrate. We also attempted to distribute sites evenly across reaches 2-5 (Fergie's Bridge to Road's End). Reach 1(Fergie's Bridge to Squamish confluence) was excluded from bar surveys due in part to the difficulty of accessing this area by foot but also due to the increased difficulty of detecting differences in stranding impacts at the relatively low juvenile Steelhead abundance in reach 1 (Appendix 1).

At least 25 one square meter quadrates were randomly distributed throughout each bar site using a table of random x and y coordinates. This approach ensured that there was an equal probability that a quadrate would be placed at any location within the site regardless of site width, which varied with ramp magnitude. Site length was always 30m but the width depended on the distance between the post- and pre-rampdown wetted edge. The pre-rampdown wetted edge was marked by painting small rocks the day prior to the rampdown. Crews searched each quadrate by removing all gravel and cobble down to where sand filled the interstitial spaces so newly emerged fish (<25mm FL) were reliably found . The assumption was that 100% of fish within a quadrate were found. All fish were identified to species and forklength measured.

Bar surveys provide an estimate of the total number of stranded fish in the site by dividing the number found by the proportion of the site that was sampled. For example, if five fish were found and the sampled area represented 20% of the site area then the estimate of all fish stranded in the sites would be:

Fish stranded per site = (fish found) / (site area sampled / total site area) = 5 / 0.2 = 25.

The number stranded at all sites was estimated as:

Fish stranded at all sites = ∑ fish found / (∑ site area sampled / ∑ total site area).

Stranding density was calculated as:

Stranding density = ∑ fish stranded at all sites / ∑ total site area

Fish stranded per square meter is the primary metric of bar stranding risk in the CASP (2018). For select rampdowns we also estimated the percent of fish stranded in relating to their estimated abundance in the mainstem based on electrofishing 1-3 days after the rampdown at or near each site. We estimated this as:

Proportion stranded = ∑ fish stranded at all sites / ∑ mainstem abundance at all sites.

Estimating the total number of fish stranded at only high risk bar habitat or for the entire study area was not a CASP (2018) objective, and thus, the sampling methods was not designed for this purpose in mind. For this to occur would require random site selection as well as habitat assessments to quantify the amount of habitat exposed for a rampdown. Since neither of these steps occurred, we provide no estimates of total fish stranded in the study area.

2.2.2 Pool survey methods

The original approach to assessing stranding risk in isolated pools and side-channels was based on the number of pools with a given risk category in the study area (CASP 2018). This has been revised to asses risk based only on the total number of fish observed in isolated pools and side-channels. Fish isolated from the mainstem in pools were assumed at increased risk of mortality either by predation or causes associated with partial or complete dewatering of pools. Pools were considered isolated if there was no connection for juvenile fish to access the mainstem.

Surveys include visually estimating the number of juvenile fish observed in all isolated pools in reaches 2-5 for all rampdowns and for some surveys included reach 1as well. Surveys in reach 1 were discontinued mid-May to August 2019 due largely to the low observed abundance during surveys November to early May that included this area. Surveys of reach 1were also not completed during the August 20, 2018 or October 22, 2018 rampdowns.

Because of the difficulty counting larger numbers of fish (>100) we estimated the minimum and maximum order-of-magnitude abundance in each pool: 0, 1-10, 10-100, 100-1,000 and 1,000-10,000. Due to daytime concealment of juvenile salnimids, which increases with age class and decreases with temperature (Bradford and Higgins 2000) the number observed was expected to be less than the number present. The order-of-magnitude abundance categories anticipated to some degree account for the proportion of fish not seen. Study components to quantify the proportion of fish seen was added to the study in February 2019. Based on initial results, it is reasonable to consider the order-of-magnitude categories as consistent indicators of abundance for comparisons during May-September when river temperatures are generally above 10 °C but are likely not comparable to visual estimates during October-April when a larger proportion of fish are concealed from view.

2.2.3 Mainstem abundance survey methods

To reflect the relative number of fish susceptible to stranding during the rampdown, crews of two electrofished the shoreline at each of the bar sites one to two days after the rampdown. We used a single-pass open-site sampling approach following the methods used for the MON-3 juvenile Steelhead estimates (Korman and Schick 2017). Using this approach, the catch from the single-pass electrofishing is expanded by an estimate of capture efficiency to estimate abundance at the site. We used capture efficiency estimates from MON-3 as well as from a small number of mark-recapture trials (see section 2.3.1).

For the majority of rampdowns, we estimated the pre-ramp mainstem fish abundance as the abundance after the ramp plus the total estimated number of fish stranded at bar survey sites. The decision to electrofish after a rampdown rather than before was based on the assumption that for rampdowns following storm driven high flow events, common in the fall and winter, electrofishing would be less reliable at the higher pre-rampdown flows than afterwards. For the August 2018 and 2019 rampdowns we electrofished prior to each ramp. With stable pre-ramp flows, this approach has the advantage of being a more direct estimate of pre-ramp abundance. In August 2018, we electrofished both before and after to assess whether we could detect any difference in abundance before the ramp vs after. We were not able to detect a difference and were thus satisfied that electrofishing before or after would most likely give us a good sense of localize abundance at the highest risk sites.

With preliminary capture efficiency data collected as part of the CASP in combination with data from MON-3, we can report site density estimates for all target species, though the precision of preliminary data is likely low other than for Steelhead and Coho greater than 35mm forklength. MON-3 provided reliable capture efficiency information for Steelhead and Coho greater than 35mm forklength at temperatures above 5°C.

2.3 Calibration methods

In February 2019 an additional component was added to estimate the observer efficiency for pool surveys and the capture efficiency of backpack electrofishing in pools and mainstem habitats. Eventually we will have specific rates for each target species, habitat type, and over a range of water temperatures. Estimating observer efficiency and capture efficiency are necessary to evaluate stranding risk levels when the number of individuals across sites could be substantially different than the fall age-0 Steelhead fry abundance used to benchmark stranding rates in Tables 2-4. Abundance differs substantially between species as well as over the course of emergence, outmigration and under conditions of high mortality.

Information about the detection probability of electrofishing and visual surveys is necessary to convert captures (electrofishing) or observations (visual surveys) into abundance estimates at our sample sites and in isolated pools. Table 5 lists the general timing of mark-recapture sampling to fill in data gaps. The timing is intended to capture water temperatures above and below 5ºC as well as periods of moderate to high abundance of each target species / age-class. Timing mark-recapture to coincide with higher abundance increases the precision of detection probability estimates since it allows more fish to be marked per day.

2.3.1 Mainstem mark-recapture method

Each mainstem mark-recapture experiment spans a two-day period following the procedures developed by (Korman et al. 2009). On day 1, a crew uses electrofishing to capture and mark ~ 100 fish of each target species/age-class in each of two sites. Site length would be extended to satisfy fish marking quotas. Captured fish are counted, measured, marked by clipping a small portion of their caudal fin, and held to recover for 20 minutes in buckets supplied with bubblers then carefully returned to slow velocity holding water along the shore of the site. On day two, the crew would electrofish the same sites using the similar effort and method used during index sampling.

The number of marked fish captured on day 2 in relation to the number marked on day one is used to estimate capture efficiency using Equation 1. We estimate the number present in the site by dividing the total catch on day 2 (unmarked and marked fish) by the estimated capture efficiency.

$$CE = R_{\text{day 2}} / M_{\text{day 1}}$$

where CE is the capture efficiency, R is the number marked fish recaptured on day 2, and M is the number of fish marked on day 1.

We estimated the average capture efficiency for several sites by calculating the mean of each capture efficiency estimates, not by calculating mean capture efficiency by the sum of all recaptured / sum of all marked across all sites. This approach gives equal weight to each site in spite of differences in the numbers of marked or recaptured fish. It is useful at the exploratory stage where it is more important to reflect the maximum variability between sites.

2.3.2 Pool mark-recapture methods

Mark-recapture methods for pools use the same procedure as for mainstem sites but with an additional step. Prior to electrofishing on day 2, each crew member would visually estimate the number of fish visible (all species, age-classes and mark groups combined), record their estimate. The mean of the estimates would represent the estimate of the number of fish visible in the pool. All species and age-classes were combined to maintain the same approach used during stranding monitoring. To increase the range of abundance levels in pools of similar size, more common smaller than $10m^2$, we increased the number of marked fish with captures from adjacent mainstem habitats.

To estimate the observer efficiency of visually assessing abundance, we first estimated the number of fish by species in each pool using the Chapman estimator (Equation 2, Krebs 1999). We then estimated observer efficiency as the number estimated visually divided by sum of all fish in the pool.

$$N = [(M+1)(C+1)/(R+1)]-1$$

where N is abundance on day 2, M is the number marked on day 1 and R is the number of marked fish recaptured on day 2, and C is the total catch on day 2.

Each sampling site included an entire pool with a target of marking 100-200 fish per site. We chose sites to include a similar range of pool sizes and depths as encountered during stranding surveys.

2.4 Survival in isolated pools

The categories for pool stranding in CASP (2018) were based on pool size/depth and fish abundance. We also hypothesize that survival varies with the duration that pools are isolated, and water temperature and quality. For example, the standing impacts are likely high for rampdowns during the summer, when higher water temperatures could lead to sub-optimal or lethal water quality and the duration of isolation is in the order of weeks or months. Contrast this with rampdowns during late fall and winter when higher frequent rain events reduce the isolation period and lower temperatures result in higher water quality. We introduced this additional study component to evaluate how survival varies with depth and time of year (temperature) but not the duration of isolation, which was held constant thus far. This involved comparing abundance in pools following a rampdown to the abundance just prior to a flow increase capable of reconnecting pools to the mainstem. We estimated abundance using the same mark-recapture method as for evaluating observer efficiency. Survival was calculated individually for each species / age-class for pools using the three maximum depth categories: <25, 25-50, >50 cm, the same depth categories used for the pool stranding categories.

2.5 Adult and redd stranding

Monitoring the impact of rampdowns on adult spawners or redds was by way of recording the number and location of stranded adults and stranded redds encountered during each of the post-rampdown pool surveys. In the case of the September 20, 2019 rampdown, redd counts were replaced by estimating the area of redds above water level due to difficult counting such a large number of redds. A pre-ramp spawner survey was added following the September 20, 2019 rampdown to assess general spawner abundance and their use of habitats potential impacted by the rampdown. Salvage crew size and equipment needs were based on this information.

3 Results

3.1 Interpreting results

After one year of monitoring, all results should be considered preliminary with a potential to change as year two monitoring increases replication of ramp types and sample size in general. Conclusions based on these results may indeed stand up over time but this can only be confirmed through additional monitoring. Summaries of stranding were provided to BCH following each rampdown and this information was used as the basis to implement mitigation strategies to reduce stranding during the year-one monitoring period.

In this revised report, we use the change in the relative risk to evaluate the two primary hypotheses. This is in response to lack of consensus between BCH, stakeholders and regulators in the approach to assessing risk and how the level of stranding relates to population level risk. We no longer attempt to directly test the hypothesized risk levels in Table 3 due to the incompatibility between their hypothesized absolute risk levels and the relative risk levels generated using the current monitoring methods. However, we can still evaluate the underlying trends represented in the Table 3 using relative risk. For example, an increase in the stranding density on bars for rampdowns at lower versus higher discharge ranges would support one of the two underlying hypotheses of Table 3. These tests are largely qualitative and do not incorporate the uncertainty of individual estimates or between estimates. This is because we lack the replication or sample size to adequately estimate all sources of uncertainty (sampling and process error).

We used the density of fish in the mainstem to assess the level of bias that abundance could have on the primary bar or pool stranding risk metrics. To do this, we include estimates of mainstem abundance along with both pool and bar stranding statistics, and for some bar stranding events, we also include the number stranded in relation to the estimated abundance at the site (proportion stranded). We also attempted to incorporate mainstem abundance into graphical comparisons between stranding risk and the hypothesized risk factors.

Evaluating the benefit of mitigation measures or influence of key factors such as season, wetted history, or discharge level depend on comparing only specific rampdowns that differ in the level of a given variable either through planned or ad hoc manipulation while other variables were held constant. Generally, inferences are stronger and more reliable when a) rampdowns differ by only a single variable, and 2) when the contrast in that variable is large. When the compared rampdowns differ by more than one variable, it becomes unclear which variable was the most likely to have caused the stranding effect, particularly with the small number of ramps that were monitored to this point.

3.2 Monitored rampdowns

Sixteen rampdowns have been monitored, including the August 20, 2018 event that preceded the development of the CASP but shared many of the monitoring methods of the subsequent rampdown assessments. Table 6 lists the flow characteristic for each rampdown. Hydrographs for each rampdown are shown in Figures 2 through 12. Rampdowns ranged widely in terms of ramp magnitude (5-90 m^3/s), discharge range and rates (1.8-16 cm/h). Wetted history also varied from 3-10 days for rampdowns following rain-caused high flow events to weeks or months when the rampdown was in response to gradual decreases in inflows (i.e. lower snow melt) or a shift to lower minimum flows. For the most part, rampdowns September-November included a wider range of rampdown magnitudes (5-90 m^3/s) and slightly more variation in the flows at the start of a rampdown (20-130 m^3/s) than those in the spring and summer of 2019 (magnitude: 5-35 m^3/s; pre-rampdown flow: 50-130 m^3/s). Two of the three high flow events October-November 2018 were in response to storm events. For these, peak flows in the day prior to the monitored rampdowns were over 100 m^3/s higher than flow at the start of the rampdown (peaking at 200-250 m^3/s). The third rampdown followed a planned pulse flow intended to initiate upstream dispersal of Chum spawners and was preceded by relatively stable but high flows. The two-day rampdown in September 2019 also followed a high rainfall event but peak flows were only 10 m^3/s higher than the pre-ramp flows. The duration of high flows prior to the fall rampdowns ranged from three to seven days. With the exception of the November 9, 2018 rampdown (20-15 m^3/s), all rampdowns during the fall of 2018 were at the WUP rate of 13 m^3/s per hour. While fall rampdowns shared common flows either at the outset or end of rampdowns, rampdowns differed in terms of magnitude and range of rampdown.

Rampdowns during the spring and summer were typically in response to reduced inflows as a result of cooling temperatures and/or reduced snowpack. For several rampdowns, BC Hydro manipulated either the rampdown range or the ramping rate to provide the contrast necessary to evaluate the benefit of mitigation measures. The first of these were for an 80-20 m^3/s reduction. Using the same rampdown rates, the reduction was done in one day on October 22, 2018 while two weeks later the drop was split over two days: November 6 (80-40 m^3/s) and November 7 (40-20 m^3/s). A key manipulation also occurred between several rampdowns in late spring. As a mitigation measure in response to high levels of stranding from the May 17, 2019 rampdown, the ramp rate was greatly reduced while the rampdown range and pre-ramp flow levels were kept consistent. This combined with the similar fish abundance greatly increased the strength of inferences about the effect of ramp rates that otherwise wouldn't have been possible if other factors varied as well. After May 17, 2019, all subsequent rampdowns were ramped with a target rates of 5 cm/h or 2.5 cm/h (actual ramp rates vary measured at the Brackendale gauge station varied from the modelled values). Additionally, the August reduction from 40-20 m^3/s was divided into three rampdown steps spread over 12 days. A similar strategy was used to reduce the magnitude of the mid-September, 2019 rampdown that followed a high rainfall event.

Due to an unusually cold and dry winter and early spring there were no operational rampdown reductions out of Daisy Lake Dam during December 2018 - April 2019. As such, no stranding related data was collected during this period.

3.3 Mainstem fish abundance during rampdowns

The shoreline abundance sampling that followed each rampdown provided valuable information about species and life stages present, emergence timing, freshwater residence, and abundance levels in monitored habitats during time periods that have had little prior monitoring effort. Tables 7&8 include the sum of electrofishing captures and abundance estimates of Chinook, Coho and Steelhead fry in the mainstem adjacent to bar survey sites. Since electrofishing occurred after the rampdown for almost all rampdowns, we estimated the pre-rampdown abundance by adding the number stranded on the adjacent bar to the post-rampdown abundance estimate (Table 8). Figure 13 and Table 9 show the estimated mean density (fish per meter or shoreline) prior to each rampdown.

For Chinook, shoreline sampling indicated that emergence began more than two months earlier than the late January date that was part of the reasoning for assigning a lower stranding risk level to the October 15 – February 15 period in the CASP (Fell and Melville 2016). While we speculate that only a small portion had emerged in early November, these results indicate that this sensitive life-stage was present during three of the four months considered a lower stranding risk. Sampling likely did not capture the peak abundance given the lack of rampdowns and subsequent monitoring between December and May. Density decreased from a high of 3 fish/m of shoreline to near zero between early May and late June. Mean forklength of Chinook fry exceeded 40 mm by late May and reached 60 mm by late June (Table 10).

For Coho, density of the 2018 cohort declined from 0.59 fish/m to zero between August and December. This could reflect both mortality, some possibly ramp related, and a shift away from the shallow mainstem habitat that we selected for bar surveys. While the gap in rampdown monitoring between December and May missed the beginning of emergence, Coho captured as part of mark-recapture calibration suggest emergence is underway by early April. From April to late June, density continued to increase with density more than doubling during June and peaking late June (15.4 fish/m). Mean forklength remained relatively unchanged from early May to the peak abundance in late June (34-37 mm, Table 10).

Similar to Coho, Steelhead fry abundance dropped between August (13 fish/m) and the end of May (0.4 fish/m) when this cohort becomes categorized as age-1 parr instead of age-0 fry. The decrease in density between October (5.1 fish/m) and early May (0.82 fish/m) represents a survival rate of 16 %, which is comparable with the 20% average overwinter survival from MON-3 reporting (Korman and Schick, 2018). However, the over 50% decrease between August 21, 2018 and October 22, 2018 and then again between early and late May had not been previously documented. This could reflect changes in abundance (natural and stranding mortalities) in conjunction with dispersal to a wider range of habitats. It was also interesting that the reduction in mainstem abundance between Aug 9 & 20, 2019 was far greater than what could be accounted for from bar stranding alone. Abundance dropped by 45% for Coho and 35% for Steelhead fry over this period yet we estimated that 0.5% or less were stranded on bars during any of the three rampdowns.

3.4 Bar stranding

For rampdowns from August 2019-September 2019, the count of juvenile salmonids found stranded across all six bar sampling sites ranged from 0 to 47 fish, by species and age-class, and when expanded to account for the unsampled portion of the site, the total number of fish stranded ranged from 0 to 261 fish (Table 11). This was comprised almost entirely of Steelhead, Coho or

Chinook fry based on forklength, in descending order of abundance. Average forklength of stranded fish was 35mm. 92% of stranded fish were less than 50mm forklength and all were less than 100mm forklength. Only one Steelhead was found that could be considered an age-1 parr based on forklength and date (May 17, 2019 survey; FL 95mm). No Chum or Pink fry were encountered during this monitoring period, which is to be expected considering no monitored ramps occurred during their emergence or outmigration period. By species and age-class, stranding density ranged from 0 to 0.31 fish/m^2 (Table 12). Other species found stranded included Three Spine Stickleback (1 fish), Pacific Lamprey (3 fish) and Coastrange Sculpin (2 fish).

There was some support for a seasonal influence. The median stranding density was higher during the Feb-15 to Oct-14 period than Oct-15 to Feb-14 for Coho and Steelhead fry at ramp rates both above and below 5 cm/h, however it was also highly variable (Figure 14a-b). The high variance for this period suggests factors other than ramp rate and season affect stranding density. Stranding densities were consistently at the very low end of the range for the Oct-15 to Feb-14 period regardless of ramp rate, though there was only a single ramp event under low ramp rate conditions. For Chinook, there was no indication of a seasonal influence on stranding and that stranding density remained relatively low during both periods and all ramp rates. This was likely a product of the consistently low mainstem abundance during the rampdowns, which largely missed the peak emergence and outmigration period. Because of the low value and low variance of stranding densities, we have not included Chinook fry in the remainder of the analysis. It is important to consider that the low stranding densities during the Oct-15 to Feb-14 period only reflect rampdowns up to the end of November and not the latter period, which is approaching the peak occurrence period for newly emerged Chinook and Pink fry as estimated by Fell and Melville (2016) and reproduced in Appendix 10.

There was no support that stranding density increased with ramp magnitude. For rampdowns with ramp rates both above and below 5 cm/h, ramp magnitude explained little to none of the variance in stranding density for Coho fry (R^2 = 0.02-0.05, Figure 15a-b) and for Steelhead fry, indicated a weak negative relationship (R^2 = 0.17-0.18). Furthermore, the highest stranding densities occurred at the lower ramp magnitudes, but so did many low stranding densities. The negative relationship was likely an artifact of the influence of mainstem fish abundance, which coincidentally for Steelhead fry, was highest for several of the low magnitude rampdowns.   This is not to say that ramp magnitude is unlikely to have an influence on stranding density, but that so far, it has not had a relatively large influence. This is largely due to a lack of contrast in ramp magnitudes under similar fish densities. It is also important to note that stranding density will underestimate the effect of ramp magnitude compared with metrics such as total fish stranded or linear density of stranding (fish/m shoreline). This is because stranding density normalizes the number stranded by the area dewatered. For example, if twice the fish are stranded for a high vs. low magnitude rampdown but the area dewatered is also twice as large, the two will still have the same stranding density. In this case, replacing stranding density with the total number stranded did not substantially increase the influence of ramp magnitude.

The strongest support that relatively high stranding levels can occur over a wide discharge range is from the comparison of the August 20, 2018 and May 17, 2019 rampdowns. These rampdowns had similar stranding densities (0.31 and 0.24 fish/ m^2, respectively; Table 12) in spite of their widely differing discharge ranges (Aug-20: 38-20 m^3/s and May-17: 125-90 m^3/s). There was also little support of an effect of post-ramp discharge on stranding density but this is difficult to assess considering it is likely confounded with mainstem abundance. For Steelhead fry, while stranding density was highest at low post-ramp discharge, it was also just as often low (Figure 16a-b).

There was support that stranding density increased when ramp rates increased above 5cm/h but only under moderate to high mainstem abundance levels (Figure 17). Stranding density remained relatively low even at rates above 5 cm/h did if mainstem abundance was also low. When ramp rates were less than 5 cm/h, mainstem abundance was a far stronger predictor of stranding density for Steelhead fry ($R^2 = 0.73$, Figures 18) than ramp rate and ($R^2 < 0.01$, Figure 19). The lack of an any increase with ramp rate suggests that reductions in ramp rates below 5 cm/h provide little added protection from stranding. For Coho fry, both were poor predictors of stranding density. The magnitude of the reduction in stranding density due to lowered ramp rates was most evident from the comparison of the May 17, 2019 and June 4, 2019, and the August 2018 and 2019 rampdowns. Within each pair, different ramp rates were used but they were similar in terms of fish abundance and size, total discharge change, and time of year but differed in the ramp rate used. For the May 17 and June 4 rampdowns, the rate was lowered from 16 to 4.7 cm/h and stranding density was reduced from 2.4 to 0 fish/m^2 (Table 12). For the August

2018 and 2019 events, the rate was lowered from 8.4 to 1.8-4.6 cm/h. The August 2019 reduction included three rampdowns over 12 days. Stranding density for the August 2018 rampdown was 0.32 fish/m^2 and averaged 0.05 fish/m^2 for the three August 2019 rampdowns (Table 13). In terms of the proportion of mainstem abundance stranded, this represented a reduction from 12% to 1-2% stranding rate. We report the percent stranded in this case since we have reliable capture efficiency estimates for Coho and Steelhead fry at this time of year. While percent stranded is not intended as a population level indicator, it does reflect site-level impacts, and in this case, the benefits of reduced ramp rates.

We do not yet have sufficient information to assess the significance of wetted history because wetted history was largely confounded with season, abundance, fish size or ramp rate. While the two rampdowns with the relatively high stranding rates (Aug 20, 2018 and May 17, 2019) were preceded by at least five days of stable or slowly decreasing flows (Figures 2 and 7) there were no comparable ramps – in terms of abundance, fish size or ramp rates – that were preceded by a short period of high flows. Similarly, all of the ramp rampdowns preceded by variable flows occurred during the fall and early winter, which have so far been characterized by larger and fewer fish.

3.5 Pool stranding

Pool and side-channel stranding occurred throughout the all 5 of the study reaches of the Cheakamus River but were most common in reaches 2, 3 and 5 (Figure 20). Rampdowns with reach 1 included in surveys were only from November 6, 2018 to May 1, 2019 and the September 2019 surveys. Unless otherwise noted, results reported are for reaches 2-5. Even over the one year study period, both the location of some pools as well as the and minimum flow required to isolate them varied due to natural bed movement and, in the case for two constructed off-channel habitats, maintenance work. All surveys included reaches 2-5. For this area, we consider that surveys included the vast majority of habitat isolated at the time of the survey.

The total length of habitat isolated ranged from 14 to 1,650 m (Table 14) and generally decreased as the post-ramp discharge increased (Figure 21 and Table 15). While the amount of isolated habitat was lower for very low magnitude ramps (smallest dots well below trend line, < 10 m^3/s), the influence was variable above this level (Figure 21). The amount of habitat isolated for single or multi-step rampdowns ending at or near 20 m^3/s was 1,102-1,650 m (excluding

November 8) compared to those with a ramp magnitude greater than 20 m^3/s that ended at 40 m^3/s or higher (639-805 m, Table 14). The low amount of habitat isolated November 8, 2018 (839 m) compared to the two prior reductions to 20 m^3/s were the result of several side-channel habitats remaining connected to the mainstem whereas they were considered isolated during the August 20 and October 22 surveys. This could have been due to changes in channel morphology during this time period or the three hour later survey start time for the earlier dates, which would have allowed for a greater drop in pool stage prior to these surveys.

Table 14 lists the minimum and maximum number of observed fish in isolated pools and side-channels for reaches 2-5. Note that the range between the minimum and maximum was fixed at a 10-fold difference to fit with order-of-magnitude approach to assessing abundance. It does not incorporate any empirical information about the uncertainty in the proportion of fish present that are observer (observer efficiency). The range of each abundance category was anticipated to include the actual abundance. When comparing pool stranding data, we chose to compare only the maximum rather than reporting the minimum or range for simplicity, however the relative difference between ramps is the same regardless which metrics are compared.

The total number of fish observed per survey varied widely (maximum stranded per ramp: 30-35,000 fish, Table 14) and was largely influenced by high stranding counts in five or less pools/side-channels per survey, with the exception of August 20, 2018 when it was a high of 16 pools. Generally, the stranding counts were far higher for rampdowns at discharges below 40 m^3/s (and particularly those that spanned 20 m^3/s) than those 80-40 m^3/s. For instance, maximum observed abundance was 30-370 for rampdowns ending at or above 40 m^3/s whereas it was 1,170-1,570 for those in the 40-20 m^3/s range. However, there were also notable exceptions. After the May 1, 2019 rampdown (50-45 m^3/s), up to 3,150 fish were observed whereas only a maximum of 40 were observed July 12 (75-47 m^3/s). This is perplexing considering the rampdown rate was lower May 1 (2.5 cm/h) than July 12 (4.8 cm/h) and mainstem abundance was almost two-fold higher in July. It is possible that this was the result of changes in channel morphology during the spring freshet considering that the amount of habitat isolated was 143 m May 1 and only 14 m July 12.

For the Figures 22-24, stranding counts were combined for flow reductions that included rampdowns distributed over more than one day to reduce the influence of ramp magnitude when assessing the effect of season, mainstem abundance and post-ramp discharge. There is little support of a strong seasonal influence on the number of fish observed in isolated habitats based on the similarity of the median value of the counts of stranded fish between the two seasons given the wide range in stranding counts (Feb-15 to Oct-14 period and Oct-15 to Feb-14, Figure 22). While the two rampdowns with substantially higher stranding counts than all others did occur during the Feb-Oct season (August 2018 and 2019 ramps), the remaining rampdowns were all within the range and distributed similarly to those in the Oct-Feb season. The influence of season may have been confounded by ramp rate considering that the ramps during Feb-Oct with the lowest stranding counts were also ramped at the lowest rates.

To help distinguish between the influence of temperature and post-ramp discharge, we categorized ramps by water temperature (above and below 9° C) and post-ramp discharge (15-40 m^3/s and 41-100 m^3/s). Even with this blocking, relationships were generally weak or were highly influenced by a single ramp event. However, it was still possible to identify some ramp conditions that tend to result in higher or low observed abundance. At temperatures above 9 °C, the stranded count was consistently low when the post-ramp discharge was 41-100 m^3/s and was unaffected by mainstem density (lower right column, Figure 23). The effects of post-ramp discharge or mainstem abundance were not apparent below 9 °C, however this may be been a product of very low and variable observer efficiency for surveys during this period.

There was some support of a positive relationship between observed abundance and post-ramp discharge but only when above 9° C (Figure 24). In this case, observed abundance decreased as post-ramp discharge increased but the strength of the relationship was low to moderate ($R^2 = 0.39$). However, the L shaped distribution suggests another variable is important at low post-ramp discharge levels, possibly fish size or wetted history. The three ramps that ended below 25 m^3/s all occurred during August and September, had ramp magnitude of at least 19 m^3/s, and moderate to high mainstem abundance of which majority were Steelhead fry. The two ramps with high observed stranding occurred in August 2018 and 2019 whereas the ramp with much lower observed stranding occurred late September 2019. Mean Steelhead forklength for the August ramps was 31-38mm whereas it was 52mm in late September (Table 10). Wetted history was also far longer for the August ramps (>4 weeks) than for the September ramp (1 week). For rampdown below 9° C, there was no evidence of a strong trend between observed abundance and post-ramp discharge but this may have been influenced the low observer efficiency during this period.

Without combining stranding counts across multi-step ramp events, there was no support of an effect of ramp magnitude for ramps that ended below 40 m^3/s ($R^2 < 0.02$, Figure 25), which was possibly more influenced by mainstem density. There was some support of a positive effect for ramps ending above 40 m^3/s. At temperatures above 9 °C, ramp magnitude explained about half of the variance in observed abundance ($R^2 = 0.49$), however the significance of this relationship is low considering it included ramps where relatively few fish were observed stranded. At temperatures below 9° C and ending above 40 m^3/s, observed stranding did increase with ramp magnitude however this is a relatively weak comparison considering it compared only two ramps.

Similar to bar stranding, several rampdowns were considered important for understanding the benefits of mitigation. Two of these relate to the minimum flow change in August 2018 and 2019 that included a flow change through the 40-20 m^3/s range. Flow reductions in August 2018 and 2019 ramp rates were reduced from 8.1 cm/h to 4.6-1.8 cm/h as well as the days for the reduction increased from one day to three ramps spread over 12 days. For all of the three rampdowns in August 2019 combined a greater number maximum observed stranded in isolated habitats was greater in 2019 (max observed = 48,080 fish) than 2018 (max observed = 34,840 fish) in spite of mitigation (Table 16). However, in 2018 there were more than twice as many fish in the shallowest pool category than in 2019 (max observed: 2018 = 6,540 and 2019 = 2,800, Table 16). This was offset by the three-fold more fish in moderate depth pools in 2019 (max observed: 2018 = 2,100 and 2019 = 8,900). Preliminary survival estimates suggest near complete mortality for Coho and Steelhead fry in the shallowest category and in moderate depth pools, near complete mortality for Steelhead fry and 53% survival for Coho (Table 20). While precision of survival estimates is low, it is possible that mortalities were similar for the August ramps in 2018 and 2019.

The May 17 and June 4 rampdowns were another pair initially considered useful for understanding the effect of reduced ramp rates on bar stranding. Both spanned what we initially considered similar discharge ranges (125-90 m^3/s May 17 and 130-100 m^3/s June 4, Table 16). The May 17 rampdown was at 16 cm/h and the June at 4.7 cm/h. The maximum stranding count was 12,320 on May 17 and only 1,000 on June 4 and reflected the reduction in the amount of habitat isolated during each event (250 m and 65 m, respectively). The 10 m^3/s lower post-ramp discharge on May 17 isolated one pool that was not isolated June 4 that accounted for 10,000 of the 12,320 considered stranded May 17. This same pool was present June 4 but was still connected to the mainstem at the time of the survey and not considered likely to become isolated, thus observed abundance was not incorporated into the total. At the time of this ramp, we were not repeating the pool surveys 1-3 days after a rampdown so we do not know if this pool eventually became isolated from the June 4 rampdown. This highlights the sensitivity of this survey approach to the order-of-magnitude estimation method in combination with relatively small difference in post-ramp discharge.

Surveys included reach 1, which extends downstream of Fergie's Bridge to the Squamish River confluence, November 6, 8, 9, 29, May 5, September 19, 20. The maximum number of fish observed in reach 1 was always 100 or less, and was 10 or under for six of the seven surveys (Table 17). Including this section would have increased the reach 2-5 total maximum observed abundance 0-6% for all but the September 19, 2019 survey when it would have increased the stranding count by 33%, but represented only an additional 10 fish. The low observed abundance November to early May was a primary reason for discontinuing surveys of this area. Surveys in reach 1 also necessitated a change to more of a sampling approach than the census of all isolated habitats for reaches 2-5. This was because crews were not able to survey the entire extensive side-channel and braid network in reach 1 with available resources. Based on the low stranding counts, it would also be more difficult to distinguish stranding effects across ramp types since all counts were relatively low. For consistency across ramps, surveys in reach 1 only included the visible width or within ~50m from the primary mainstem channel leading to the Squamish River. For instance, there are at least two side-channels branch off of the Cheakamus and enter directly into the Squamish River that were never surveyed. This would underestimate both the total amount of fish stranding and habitat isolated.

Visual abundance estimates were generally poor indicators of abundance at warmer water temperatures June-September ($R^2 < 0.25$) and had no predictive ability at colder temperatures during March-April ($R^2 < 0.01$, Figure 26). Though highly variable, mean observer efficiency (OE) generally increased with water temperature based on trials in March, April and June (mean OE: 0.06, 0.12, 0.49; temperature: 4.6, 7.4, 13.1 °C respectively; Table 18). For example, if an observer saw 60 fish in a pool during March it would reflect an abundance of 1,000 fish whereas during June, when temperatures are considerably higher, seeing 60 fish would reflect an abundance of only 123 fish. Without accounting for this, visual estimates are not reliable indicators of abundance across temperatures, which coincides with seasons and provide no indication of abundance below 7 °C. For instance, at temperatures near or below 5 °C, only 0 to 1 fish were observed in three pools even though abundance ranged from 190-350 (Table 19). Additional monitoring will help quantify observer efficiency over a range of water temperatures, as well as how it varies with other factors, such as pool depth or substrate but it still may not allow for adequate comparisons across all temperatures or seasons.

We estimated survival for Coho and Steelhead fry in pools over a 21-day period starting two to three days after the August 20 rampdown and ending September 11-12, just prior to a forecasted high rainfall event that would have reconnected the pools to the mainstem. For Coho fry, mean survival was 53% in pools 25-50 cm deep and 22% in pools greater than 50 cm, while for Steelhead fry survival was 0% in pools up to 25cm, 1% for pools 25-50 cm and 43% in pools over 50 cm (Table 20). For Coho, only a single pool (depth>50cm) had 0% survival and the remaining five sites survival ranged from 30-67% whereas for Steelhead, five of the eight sites had 0% survival with the remaining sites ranging from 5-50% (Appendix 7). The delay between the August 20 rampdown and beginning the trial was to ensure pools depth had come to equilibrium with mainstem levels since the intention was to measure survival in pools that would remaining wetted until the next high-water event. While this does accurately reflect survival rates over the study time period, it over estimates survival compared with if the trial had started on the day of the rampdown. This is because pool depth decreased for several days following a rampdown. Pools within the deep category would shift to moderate or possibly even shallow category when measured several days later and shallow or moderate pools could become dewatered during this time. Survival would likely be lower than reported if the assessment started the day of the rampdown since many of the pools included in the study would have become dewatered whereas none were dewatered over the 21-day trial.

Observing the reduction in the number and depth of pools 2-3 days after the rampdown was a key learning in itself since stranding monitoring did not include repeat pool surveys several days after rampdowns. Prior to this, our assumption was that pool levels decreased at a relatively similar rate to mainstem levels. We accounted for post-survey reductions of less than 10 cm but not the 30-40 cm change that likely occurs in pools with greater separation from the mainstem. Thus, stranding risk was likely higher than reported as abundance for a number of pools would shift to a lower depth category.

3.6 Adult and Redd stranding

Adult and redd stranding risk levels were not originally a component of the CASP. All adult and redd stranding were restricted to November 2018 and September 2019. During November 2018, a total of 30 live Chum and Coho adults were observed in isolated pools 20-150 cm deep following three rampdowns (Table 21). Over 80% of these were found within 1km of the Road's End survey start point. As well, nine redds were stranded either on bars or in isolated pools after rampdowns during this period. For both adults and redds, stranding followed rampdowns that included a wide range of discharge levels (20-15, 80-40, 130-40 m^3/s) and ramp rates (1.1-11.8 cm/h). No adult or redd stranding was observed for rampdowns May-August.

Approximately 2,400 Pink adults (minimum estimate) were stranded during rampdowns September 19-20, 2019. The September 19 rampdown from 60-44 m^3/s at a rate of 3.9 cm/h stranded 18 adult Pinks and 6 redds whereas the September 20 ramp from 43-22 m^3/s stranded approximately 2,380 pre- and post-spawned adult Pinks. Approximately 150 of these were returned alive the day of the rampdown with a priority being placed on salvaging female fish. We also estimated that 7,700 m^2 of bar habitat containing redds became dewatered as a result of the flow reduction. Based on observations, we estimated redd densities of 0.2-1.5 redds/m^2, which equates to 1,500-9,200 stranded redds. No enumeration of Pink adults has occurred since 1997. Estimated maximum run size from DFO NuSEDS database for 1952-1997 is 75-555,000 adults and a mean of 61,000 adults. Assuming equal sex ratio and average run size, the stranded redds would represent 5-30% of all redds.

The September 2019 rampdowns followed a seven day period of high flows that exceeded 45 m^3/s for 5 days. The overall flow trends were similar to the Squamish River, however for the Squamish, the relative increase was higher and over a shorter time period (Figure 27). Though there were reports of stranded Pinks on the Squamish, we lack information to estimate whether the level of stranding was greater on the Cheakamus than Squamish river. The decision to return flows to the WUP minimum (20 m^3/s) August 19-20 was to reduce the time period that Pinks could spawn in habitat that would become dewatered at base levels. Extending the high flow period may have increased the proportion of Pinks able to spawn (reduced adult stranding mortalities) but would also increase the number of redds at risk of stranding.

4 Discussion

4.1 Effectiveness of monitoring

4.1.1 Study approach

The study approach has provided considerable variation in the rate, range, magnitude and season of rampdowns, all of which are important for gaining an understanding of their impact on stranding. The variability observed during year one also highlighted the large number of monitored rampdowns required to distinguish between the many factors that could influence stranding. Given these challenges, learnings from year one were improved largely through BC Hydro's Generation System Operation's ability to limit the number of variables that rampdowns differed by, so that certain flow reductions could be paired with those that differed only by a single variable. Examples of these include:

- May17 and June 4 - different ramp rates but similar wetted history, and ramp range
- August 2018 and August 2019 – different ramp rates and stepped versus single reduction. This compares two ramp types rather than individual factors.
- Aug 2018 and May 17, 2019 – similar ramp rates and wetted history but different ramp magnitude and range.
- Oct 22, 2018 and Nov 6-8, 2018 – Similar ramp range and rate but a stepped versus singe reduction. Different wetted history is a confounding factor though.

The study approach to assess bar stranding risk based primarily on the highest risk habitats increases the likelihood of detecting relationships between ramp types and stranding risk. It also increases the likelihood of detecting a response from mitigation measures since the magnitude of the reduction in stranding level is potentially larger than if surveys included all habitat risk levels. The trade-off is that it is not easily used to assess river-wide stranding impacts since sites do not reflect average risk levels across a wider range of habitat types. This could lead to both under and over estimating risk depending on a) the amount of habitat across a range of habitat risk levels, b) how this varies with discharge and c) how stranding risk varies with habitat variables such as bar slope and substrate size.

4.1.2 Bar monitoring

The bar sampling methodology has been an effective approach for estimating the number and density of fish stranded in interstitial spaces on low angle bars. The random assignment of sampling quadrates within the dewatered area at the sites in combination with searching each quadrate down to the fine substrate provides an unbiased estimate of the number of fish stranded at the sites. Thus, fish per square metre, the primary metric for rating stranding risk level on bars, is likely a reliable indicator of the probability of finding a stranded fish in a given area of high risk habitat. Marking the wetted edge late in the day prior to a rampdown allowed for quantification of the area dewatered at each site. The dewatered are is needed to estimate the total number of stranded fish in a site and the number stranded per meter of shoreline. If used in combination with random site selection, this metric is compatible with area-based approaches to estimating reach or river-wide stranding. However, for comparing site-level differences in stranding risk, this metric will underestimate the effect of ramp magnitude. This is because site width, and thus site area will tend to increase as ramp magnitude increases. This would not occur if stranding density is based on the number of fish stranded per metre of shoreline, which is unaffected by differences in site width. If reliable mainstem abundance estimates are available, using the proportion of fish present at the site that were stranded is also an effective index that controls for differences in fish abundance. We generally avoided using this approach in year one largely due to insufficient capture efficiency information to expand electroshocking captures to abundance estimates for some species and time periods (late summer and early spring). This information is easily collected with additional mark-recapture trials. We also avoided this metric as it becomes very imprecise when fish abundance is low (approximately <0.5-1 fish/m) and when the sampled proportion of the site is also low (<20%).

4.1.3 Pool and side-channel sampling

Pool and side-channel monitoring was designed as a rapid assessment tool to provide a synoptic overview of the number of fish impacted through isolation in partial or completely dewatered habitats as well as to quantify the amount of habitat isolated for each rampdown. It was also designed as a method to have crews cover the entire river so that qualitative unanticipated impacts could be observed and recorded. It was expected to detect only order-of-magnitude differences in the abundance of impacted fish. To this end, this method, given its current uncertainties, has limited use for assessing the hypothesised relationships and the benefits of mitigation. It also is limited for assessing overall risk especially between periods when abundance or temperature differ.

There are two main limitations of using the current pool monitoring methods to assess the benefit of mitigation or the significance of stranding risk to fish populations, even at the most general level. First, the relationship between the number of fish in a pool and the number observed remains highly uncertain at temperatures below 11°C, which represents a substantial portion of the year, and becomes uninformative below 7.5 °C (Table 18). Methods based on a sampling approach using more quantitative enumeration methods such as 1-pass electrofishing have the potential to produce more reliable results . These methods are well understood and are likely to yield unbiased and adequately precise estimates with even only modest increases in sampling effort. Second, it is difficult to assess the benefit of mitigated ramps on pool isolation when the compared rampdowns do not span near identical discharge ranges. This is because a greater area could become isolated for one ramp than another, influencing the number of fish isolated. This confounds the opportunity for fish to enter habitats at risk of isolation with the probability of becoming stranded if in a pool. While the former is important for understanding the importance of discharge on stranding potential, the latter is the ideal metric to evaluate the benefits of mitigation measures. An additional limitation of the current survey methods is that it underestimates both the area and number of fish isolated as a result of the 1-2 days for water levels in pools and side-channels to reach equilibrium with the mainstem levels. This effect wasn't apparent until members of the Squamish Watershed Society surveyed several pool stranding locations 1-2 days after the rampdown to find that pools that were connected to the mainstem on the day of the ramp were now isolated and, in some cases, completely dewatered (pers. comm. Francesca Knight).

The relationship between abundance in pools, pool depth, water temperature, or the duration of isolation and survival are key uncertainties that presently limit our ability to accurately assess pool and side-channel stranding impacts. Estimating survival becomes more important if mitigation is unsuccessful and rampdowns cannot be avoided, such as during high rainfall events. This is important for assessing impacts during the winter when large rampdowns are more frequent but may have a lower per-ramp impact if cooler temperatures allow for sufficient habitat quality. This is particularly important for large and deep pools, which account for the majority of fish observed. Trials to quantify survival rates have started to improve our ability to interpret pool survey data.

Pool monitoring was useful for quantifying the amount of habitat isolated by a rampdown. At its most basic level, the length of pool and side-channel habitat dewatered or isolated is an indication of the potential for stranding to occur. Generally, we found that the length of pool or side-channel isolated by ramping increased as discharge decreased, in particular at flows below 40 m^3/s (Table 16). We also found that even during just one year of monitoring the discharge that specific individual pools became isolated changed.

4.1.4 Adult and redd sampling

The present monitoring approach of counting stranded adults is likely adequate but may require additional crew if high numbers of adults and/or redd stranding is expected. Our approach of counting stranded adults since all isolated habitats are surveyed. It is more difficult to assess whether redd stranding is adequately quantified with the present methods since downstream surveys do not search all possible redd stranding locations. However, surveys do target known or expected standing locations. At high stranding levels, we would need to test the effectiveness of the area based redd estimation method using redd counts at a subsample of stranding locations. Whether the present approach is adequate also depends on the type and certainty of information needed to evaluate the impact of ramping on adult and redd stranding.

4.2 Effectiveness of mitigation measures

4.2.1 Reduced ramp rates

There is strong support that reducing ramp rates to 5 cm/h or less greatly reduces bar stranding risk based on the reductions between paired rampdowns that differed primarily in the ramp rates used. The stranding density for Coho fry was reduced from 0.24-0.00 fish/m^2 when the ramp rate was reduced from the WUP maximum rate of 16cm/h (May 17) to 4.7 cm/h (June 4, 2019). For the August 2018 and 2019 rampdown, stranding density was reduced from 0.32 fish/m^2 to 0.05 fish/m^2 when rates were lowered from 8.3cm/h to 1.8-4.6 cm/h. However, considering mitigation in 2019 included lower ramp rates and splitting the overall spill reduction into three rampdowns distributed over 12 days, it is less clear that the benefit was the result of the lower rates alone. There was also no indication that rampdown rates near 2.5 cm/h were any more effective than 5cm/h based on the different ramp rates used during the August 2019 rampdown. Instead, the level of bar stranding for ramps at or below 5 cm/h were more likely a product of difference in mainstem abundance than ramp rates.

There was little to no support that reduced rampdown rates reduce pool stranding. First, there was no reduction in the total number of fish observed stranded for the three August 2019 ramps combined (ramp rates < 5cm/h) than with August 2018 (WUP ramp rate = 8.1cm/h). Second, mortalities for these rampdowns were likely similar enough to conclude that mitigation had no effect considering the preliminary survival estimates. This trial suggested near compete mortality over the 21 day period following the August 2019 rampdowns for Steelhead fry in pools less than 50 cm deep and Coho fry in pools less than 25 cm deep, and 22-53% survival in the deeper depth classes. The May 17 and June 4 rampdowns also provide little support that ramp rates affect pool stranding rates even though very different ramp rates were tested. Pool stranding was lower June 4 because of higher post-ramp discharge, which allowed for connectivity and a single pool that accounted for approximately 90% of strandings May 17.

These results are only partially consistent with experiments using artificial streams and known numbers of fish. Bradford et al. (1995) found similar reductions in bar stranding when rampdown rates were reduced from 30 cm/h to 6 cm/h. They also found that at 6 cm/h most fish left pools prior to them becoming isolated, which is somewhat counter to our results. However, they used fish with a mean forklength of 58-89mm, which is far larger than the average size of the youngest age-classes present in the Cheakamus River during monitored rampdowns (mean forklength = 30-70mm).

4.2.2 Distributing a flow reduction over more than one day

Distributing an overall flow reduction over two or more days appeared to have little effect at reducing either bar or pool / side-channel stranding risk. Effects were either confounded with another mitigation approach (rate reduction), other flow variable or a lack of clear benefit. There were two potential comparisons where abundance, temperature, and range of the total flow reductions were similar enough to evaluate the benefits of reducing the daily total change in flow. In the case of October 22 and November 6-8, 2018, there was no clear benefit from mitigation. Both reductions spanned somewhat similar discharge ranges (65-20 m^3/s and 80-20 m^3/s, respectively) and ramp rates (9.5-9.9 cm/h) but were completed over one and two days, respectively. Risk levels were moderate for pool stranding and low for bar stranding for both time periods suggesting no benefit. Using the number of fish observed in isolated habitats, there was support of a benefit to the two-day reduction. However, the reduction could have been in response to the highly variable pre-rampdown flows for the November event. Flows were stable at the pre-ramp level for three days prior to October 22 whereas flows were receding both naturally and by rampdown (unmonitored) from a high of 200 m^3/s prior to November 6. The August 2018 and 2019 ramps also provide little support for this mitigation measure since the multi day reduction was confounded with reduced ramp rate. There was greater support that the reduced bar stranding was more a product of reduced ramp rates rather than the increased number or days for the flow reduction considering the large reduction in bar stranding with reduced ramp rates (May 17- to June 4) and the lack of a clear response to bar stranding (October 22 to November 6-8).

4.2.3 Salvage

Salvage was not explicitly tested as part of CASP monitoring other than limited and reactive adult salvage on September 20, 2019. Prior to this, salvage was primarily in the form of removing the small number of adults from isolated pools during downstream pools surveys. Juveniles captured in pools as part of quantitative sampling were returned to the mainstem once sampling, observer efficiency or survival trails were complete. Salvage may be useful for mitigation if it can be successfully scaled in relation to the expected level of stranding, however, this has not been tested yet. As described by Korman et al. 2018, we can use the estimated capture efficiency for backpack electrofishing to assess the logistical requirements for juvenile salvage. Preliminary capture efficiency estimates for pools range from 6-55% depending on the species, temperature and density. Undertaking salvage capable of up to a 50% reduction in juvenile mortality following the late August rampdown would require at least eight, and possibly more, two-person electroshocking crews even if just targeting pools less than 50 cm in depth (high probability of mortality). This assumes two passes of each pool to capture 50% of stranded fish and each crew salvaging 60 m of the 500 m or more of pool habitat. Effort would need to be increased further for high abundance levels possibly during peak fry emergence periods if reductions in mortality larger than 50% were targeted, or if all pool/side-channels were targeted. Measuring the effectiveness of this approach would require additional monitoring to understand what proportion of stranded fish were salvaged not only at the individual pool level but for the entire river.

A dedicated salvage effort was only undertaken on one occasion thus far. Following the September 20, 2019 rampdown, a crew of four captured and returned alive approximately 150 adults over a six hours period. Assuming a crew could salvage 250 fish in a full day and with improved efficiency, it would require approximately 10 crews to salvage all of the estimated 2400 stranded adults estimated to have become stranded that day. Assembling a team of 40 people with the necessary training and equipment would represent a significant logistical challenge that far exceeds the scale of fisheries work on the Cheakamus to date. A number of additional changes to both ramping and salvage may increase the proportion of fish successfully salvaged but at this point have none been tested. Deploying salvage crew during the rampdown rather than afterwards could further increase the proportion of live returns. This would require either shifting the ramp to daylight hours or training and equipping the salvage crews for night work. Reducing the daily stage change could also reduce the number of stranded per day, which could reduce either the number of crews or the time needed for salvage. Estimating spawner abundance is a key component for matching crew and resources to the predicted stranding level. Whether the pre-ramp survey now implemented during adult spawning provides sufficient information has not been evaluated and should be expected to be less precise as abundance increases as is commonly the case for visual estimation methods.

4.2.4 Physical works

Physical works includes bar contouring to reduce the area isolated during a rampdown as well as side-channel intake contouring to maintain flows at lower mainstem discharge levels. Maintenance of side channels included lowering the invert at the intake to the Mykiss and Wountie side-channels mid-August 2019 providing an example of a successful physical works project. Lowering the invert allowed both of these channels to remain wetted and connected to the mainstem at one or both ends at 17 m^3/s; whereas in the fall of 2018, both became isolated at flows below ~30 m^3/s, and at 20 m^3/s the entirety of both channels were considered isolated. This likely avoided considerable mortalities considering that approximately a third of the 3,500-34,840 fish observed in isolated habitats following the August 20, 2018 rampdown were from these two channels, with most counted as mortalities in the dewatered intake to the Mykiss side-channel. This work is traditionally completed as part of a larger maintenance project of several enhanced off-channel areas with project funding secured by Squamish River Watershed Society and funded by the Fish and Wildlife Compensation Program (Edith Tobe. pers. com). The frequency of this type of work would likely have to increase, as well as securing reliable long term funding, for this to be a viable stranding mitigation approach.

Bar contouring trials on the Columbia River suggest they would reduce stranding in isolated habitats but also that they would likely require ongoing maintenance to remain effective (Irving *et. al.* 2015). This study did not assess the impact of contouring on habitat quality or quantity while wetted.

4.3 Alternative mitigation measures

4.3.1 Ramp avoidance

There is potential to reduce the magnitude and possibly the number of rampdowns by modifying how changes in inflows are managed through storage in Daisy Lake, power generation and spill from Daisy Lake dam. This could have potentially high costs in terms of lost power generation and operational flexibility. It could also lead to unintended consequences for downstream biota through further stabilization of the hydrograph. In addition, because of the small storage capacity in Daisy Reservoir, opportunities for increased flow management are limited in particular during large inflow events and low-flow periods. Ramp avoidance is a more likely option when ramps are to optimize generation or in response to smaller or more gradual changes in inflow. Within these constraints, there may be opportunities to reduce the frequency of ramps either during time periods when particularly sensitive life-histories are present and in particular, when ramps extend below 40 m^3/s. Below 40 m^3/s, pool stranding risk increases regardless of ramp rate, and ramp rates are less effective at reducing bar stranding (September and possibly earlier). Based on the amount of pool and side-channel habitat isolated, reductions in pool stranding could be highest when up and down ramp avoid the 15-25 m^3/s and still high for the 25-45 m^3/s range (Table 23). Ramp avoidance is included as an additional mitigation measure to test when post ramp discharge is below 40 m^3/s and outside of the Chum and Coho spawning period since short duration pulse flows are associated with improved Chum egg-fry survival (Middleton *et al*. 2018).

4.3.2 Conditioning flow

Conditioning flow (a rapid decrease and increase in flow prior to a rampdown) appeared to reduce pool stranding on the Columbia River (Irving et al. 2009) but has not been tested further largely due to the risk that while they may trigger fish to leave pools, they could also lead to increased bar stranding in the process (Golder 2017).

4.3.3 Increased minimum flows

Increasing the minimum flow to near 40 m^3/s, would greatly reduce the number of fish stranded in pools/side-channels. Though not a modification of ramping, a minimum flow of 40 m^3/s could have reduced observed abundance for all but one of the ramps 1,000's-10,000's of fish to 100's-1000's or lower. However, this could lead to reduced egg-fry survival for fall and winter spawning species if flows cannot be maintained throughout the entire period spawning and incubation period; i.e. incubation success not being affected by redd dewatering, which is somewhat unclear. Modelling the probability of maintaining higher minimum flows would improve the understanding of the trade-offs of this approach. Increasing the minimum flow during fall and winter to even 25 m^3/s would eliminate the highest impact ramps in terms of the amount of habitat isolated for a given ramp magnitude.

5 Key uncertainties and data gaps

There are different levels of uncertainty related to quantifying stranding impacts and how dam operations impact the level of stranding. First is whether the study approach is capable of providing the necessary information for decisions around the type and level of mitigation to meet fisheries management objectives. This involves how risk is assessed, baseline conditions set and the decision framework for type and timing of mitigation. While these are important components of the CASP, they are beyond the scope of this report. Second is whether the monitoring methods have provided useful information for assessing risk based on the current CASP (2018) approach. In particular, are results adequately unbiased and do they provide useful information for the presently stated objectives. Third is how the amount of contrast and replication in ramp types affects reliability and applicability of inferences we draw from the results.

5.1 Monitoring methods

5.1.1 Bar stranding

The primary uncertainty is whether the number and location of sampling sites is adequate to characterize stranding for high risk habitats. We have high confidence that sampling methods provided unbiased estimates of stranding density at the site level but less so for high risk habitats in general. Present uncertainties include:

a) Does sampling at six sites provide adequate precision? Precision has not yet been quantified but will be in the next reporting cycle. The question also depends on the level of precision that is needed for decision making. For instance, much lower precision is necessary if the objective is to differentiate between three equally wide risk categories that span the range of stranding densities collected so far (range: 0-0.3 fish/m^2). However, much higher precision is required when the range in effects is much smaller, as is the case when evaluating the effectiveness of ramp rates or mitigation approaches for rampdowns using rates of 5 cm/h or less (range: 0-0.6 fish/m^2).
b) Does non-random site selection provide an adequately unbiased estimate of stranding risk for high risk habitat? Addressing this requires a description of high

risk habitat characteristics (i.e. bank slope, gradient and substrate size) and a comparison of average stranding levels for this habitat type and those based 'selected' sites. Random site selection is the preferred method but also requires a high level of habitat mapping. We anticipate this will be possible with further development of the TELEMAC 2D model.

5.1.2 Pool stranding

Generally, we question whether the current pool stranding assessment method is capable of providing the level of information necessary to understand stranding impacts or the benefit of different levels or types of mitigation. Uncertainties include:

a) The inability to compare stranding risk across seasons and temperatures considering in influence of temperature on observer efficiency.
b) High uncertainty in observer efficiency across all water temperatures but particularly at low temperatures.
c) Low ability to assess how risk is influenced by differences is survival across seasons and pool depths.
d) The amount that stranding risk has been underestimated as a result of the continued reductions in pool depth in the days following the rampdown and pool survey, and whether this varies with season, ramp magnitude and post-ramp discharge level.
e) Whether mainstem abundance estimates based on sampling in shallow habitats is a good indicator of abundance in habitats prone to pool stranding and whether this varies by season, species or age-class.

5.2 Ramp contrast and replication: ramp range, season and mitigation types

Our confidence that the assessment of stranding risk factors are not due to chance, coincidence or were confounded other factors increases with the number of monitored ramp and amount of replication. Inferences will become more applicable as a wider range of ramp types across more of the year are included. The rate data gaps will be reduced will increase with purposeful selection or active manipulation of ramps to include absent ramp types. Ramp types not yet included in monitoring:

a) Year round –pre-ramp discharge above 130 m^3/s and post-ramp discharge above 100 m^3/s
b) December-April – All ramp types. One flow reduction was monitored in February 2020 that is not included in this report.
c) December-July – Post-ramp discharge below 40 m^3/s.
d) July-October – Post-ramp discharge above 70 m^3/s

If the anticipated change to a more quantitative pool/side-channel monitoring method and whole-river estimates of stranding impacts occurs, there would be a need to establish baseline stranding levels using the new stranding metric and baseline ramping conditions. If baseline rampdowns use a fixed ramp rate and maximum daily stage change, baseline monitoring could target ramps that contrast in terms of post-ramp discharge, season and possibly abundance. With five post-ramp discharge categories and two season, using Table 3 as an example, there are 10 permutations (unique combination) that would ideally be represented in baseline monitoring. If abundance is considered too variable within each seasonal category and thus, considered as an additional factor, this would increase the number of permutations by the multiple of the number of abundance categories. Even with two abundance categories, the number of permutations (20) likely exceeds the current annual funding levels. How to allocate monitoring resources to provide the most useful baseline information depends on several factors.

First, will fish abundance be factored into a metric to quantify stranding or treated as an additional risk factor? We already have good information that abundance influences the number of fish stranded on bars. Also, outmigration monitoring (MON-1) and standing stock estimates (MON-3) indicated abundance varies greatly between species, and across seasons and years (Table 22). Considering this, it is important to control for it when assessing the effect of the other hypothesised risk factors. Options to do this include incorporation abundance into the stranding metric, such as the number stranded in relation to the number present, target ramps with similar abundance, or to include a range of abundance levels within each season in the matrix of ramp types. Using a stranding metric that is independent of abundance would provide the greatest flexibility in ramp timing than if the influence of abundance was controlled through targeting only those with similar abundance. While both would still require some level of abundance monitoring, a metric that incorporates abundance could require greater precision in the abundance estimates than if it is just used to confirm whether abundance is similar enough for it not to overly influence the stranding metric. Monitoring across a range of abundance levels allows for high flexibility but also greatly increases the number of ramp types to monitor under baseline conditions.

Second, can monitoring focus on common ramp types and/or those scheduled to occur every year. This reduces the opportunity to develop more general relationships between stranding risk and the risk factor but would focus resources on ramp types that are most likely to occur and thus, are more likely to be replicated and tested with a different type or level of mitigation. Examples include:

- August reduction in minimum flows from 38-20 m^3/s – This ramp will likely continue until there is change in when or how minimum flows are set. It is also a relatively unique ramp type in that pre-ramp flows are relatively stable for weeks or months, ramp magnitude is consistent, and post-ramp discharge is relatively low.
- Fall and winter high magnitude flow reductions following spills resulting from high rainfall events – While historically these were completed in a single ramp, they are now comprised of a number of smaller ramps spread over server days. These are characterized by moderate to high pre-ramp discharge, post-ramp discharges near or below 20 m^3/s, and a wetted-history on the order of days rather than weeks.
- Late spring and early summer decreases due to changes in snow melt and precipitation – These ramps are characterized by post-ramp discharge above 40 m^3/s, and wetted-history greater than a week.

A third consideration is whether baseline ramp conditions will be the rule until monitoring has included all ramp types. The question is whether a ramp type that has already been monitored would go un-monitored if it reoccurred, allowing budget to be reserved for monitoring a different type, or would a different ramp rate or daily magnitude be tested and monitored prior to completing all baseline monitoring? For example, if there were multiple fall/winter spills that would result in similar rampdowns, would the ramp rate or daily magnitude be increased after the first event or would baseline conditions be continued and the ramp go un-monitored?

6 Summary of monitoring

6.1 Bar stranding

For bar stranding, we use stranding density (fish stranded/m^2 dewatered bar) as the index of relative stranding risk. WUP maximum ramp rates in combination with stable high flows and moderate to high mainstem abundance resulted in relatively high stranding density at both low (40-20 m^3/s) and high flows (125-90 m^3/s). Reduced ramp rates (15cm/hr to 5 cm/hr and 8.3 cm/hr to 1.8-4.7 cm/hr) were effective at lowering stranding density for both Steelhead and Coho fry. The reduced ramp rates (<5cm/hr) were effective a reducing stranding density over a wide range of flow reductions (130-100 m^3/s, 100-67 m^3/s, 60-40 m^3/s and 43-22 m^3/s). Reducing ramp rates to 1.8-2.4 cm/h provided no clear reduction in stranding risk compared with rates near 5 cm/h. When using stranding rates at or below 5 cm/h, stranding risk was more influenced by mainstem abundance than stranding rate. Stranding risk was uniformly low when mainstem abundance was less than 1fish/m.

Support for a seasonal influence is weak once we account for differences in ramp rates. Mean stranding density was higher during Feb-15 to Oct-14 period than Oct-15 to Feb-14 for Coho and Steelhead fry at ramp rates both above and below 5 cm/h, however it was also highly variable. There is no support that stranding density increased with ramp magnitude. However, stranding density is relatively insensitive to the increases in the number of fish stranded across as magnitude increase because it normalize stranding by area. Expanded total or linear density would be better indices for assessing in influence of ramp magnitude. Thus, there is little support for dividing one rampdown into several smaller rampdowns spread over several days reduces bar stranding. We do not yet have sufficient information to assess the significance of wetted history because wetted history was largely confounded with season, abundance, fish size or ramp rate.

6.2 Pool stranding

We used the number of fish observed in isolated pools and side-channels in reaches 2-5 as the index of relative pool stranding risk. Based on total number of fish observed, the estimated impact of rampdowns varied widely (maximum observed per ramp: 30-35,000 fish) and was largely influenced by high abundance in five or less pools/side-channels per survey, with the exception of August 20, 2018 when it was high in 16 pools. Generally, the index of pool stranding was far higher for rampdowns with post-ramp discharges below 40 m^3/s (and particularly those that dropped below 20 m^3/s) than those that ended in the 80-40 m^3/s. For instance, maximum observed abundance was 30-370 fish for rampdowns ending at or near 40 m^3/s whereas it was 1,170-1,570 for those that ended in the 40-20 m^3/s range. There was some support of a positive relationship between observed abundance and post-ramp discharge but only when above 9° C. There was no support that reduced ramp rates in combination with distributing a ramp over several reduced stranding based on comparison the August 2018 (8.4 cm/h and 1 day ramp) and 2019 (1.8-4.7 cm/h and 3 rampdowns) rampdowns, which had similar mainstem abundance and total discharge change. A greater number of fish were observed in isolated pools in 2019 (max observed = 48,080 fish) than 2018 (max observed = 34,840 fish) in spite of mitigation. Considering preliminary survival estimates, though precision of these is low, it is possible that mortalities due to pool stranding were similar for the August ramps in 2018 and 2019 in spite of mitigation.

Observing the reduction in the number and depth of pools 2-3 days after the rampdown was a key learning in itself since stranding monitoring did not include repeat pool surveys several days after rampdowns. Prior to this, our assumption was that pool levels decreased at a relatively similar rate to mainstem levels and was largely complete by the time surveys were commenced. We accounted for post-survey reductions of less than 10 cm but not the 30-40 cm change that was observed at some locations 2-3 days after the rampdowns. Thus, stranding risk was likely higher than reported since a number of pools would shift to a lower depth category and some pools considered connected on the day of the rampdown would have later become isolated.

Visual abundance estimates were generally poor indicators of abundance at warmer water temperatures June-September ($R^2 < 0.25$) and had no predictive ability at colder temperatures during March-April ($R^2 < 0.01$). Though highly variable, mean observer efficiency (OE) generally increased with water temperature based on trials in March, April and June (mean OE: 0.06, 0.12, 0.49; temperature: 4.6, 7.4, 13.1 °C , respectively). Without accounting for this, observed abundance estimates are not reliable indicators of abundance between seasons and are uninformative below 7 °C.

Survival estimates for pools over a 21-day period starting two to three days after the August 20, 2019 rampdown indicated survival increased with pool depth. For Coho fry, mean survival was 53% in pools 25-50 cm deep and 22% in pools greater than 50 cm, while for Steelhead fry survival was 0% in pools up to 25cm, 1% for pools 25-50 cm and 43% in pools over 50 cm. While this does accurately reflect survival rates over the study time period, it over estimates survival compared with if the trial had started on the day of the rampdown. This is because pool depth decreases for several days following a rampdown. Pools within the deep category would shift to moderate or possibly even shallow category when measured several days later and shallow or moderate pools could become dewatered during this time. Survival would likely be lower than reported if the assessment started the day of the rampdown since many of the pools included in the study would have become dewatered whereas none were dewatered over the 21-day trial.

6.3 Adult stranding

Adult salmon and redds stranding was only encountered during November 2018 and September 2019. The level of stranding varied widely. During the three November 2018 rampdowns, a total of 30 live Chum and Coho adults were observed in isolated pools 20-150 cm deep following three rampdowns. Over 80% of these were found within 1km of the Road's End survey start point. As well, nine redds were stranded either on bars or in isolated pools after rampdowns during this period.

Approximately 2,400 Pink adults (minimum estimate) were stranded during rampdowns September 19-20, 2019. The September 19 rampdown from 60-44 m^3/s at a rate of 3.9 cm/h stranded 18 adult Pinks and 6 redds whereas the September 20 rampdown from 43-22 m^3/s stranded approximately 2,380 pre- and post-spawned adult Pinks. The rampdowns followed a seven day period of high flows that exceeded 45 m^3/s for 5 days. The overall flow trends were similar to the Squamish River, however for the Squamish, the relative increase was higher and over a shorter time period. Though there were reports of stranded Pinks on the Squamish, we lack information to estimate whether the level of stranding was greater on the Cheakamus than Squamish river. The decision to return flows to the WUP minimum (20 m^3/s) August 19-20 was to reduce the time period that Pinks could spawn in habitat that would become dewatered at base levels. Extending the high flow period may have increased the proportion of Pinks able to spawn (reduced adult stranding mortalities) but would also increase the number of redds at risk of stranding.

7 Recommendations

7.1 Monitoring

7.1.1 Bar stranding

- Continue using the current sampling approach. At the site level, this likely provides an unbiased estimate of the number of fish stranded.
- Consider a stratified random site selection approach. Random site selection is the most reliable method to accurately estimate the average stranding level. Stratifying bar habitat into categories based on shoreline slope (e.g. low < 5%, high ≥ 5%) would likely reduce the sampling effort required for adequate precision compared to without stratification. A stratified random sampling approach also allows for river-wide stranding estimates when expanded by the amount of each habitat type.
- To reduce uncertainty of low density fish abundance estimates when calculating stranding rates, increase the number or size of shoreline sampling sites when the expected fish abundance is below 1 fish/m. This applies primarily to late fall and early winter surveys when abundance was generally below 1 fish/m in shallow habitats.

7.1.2 Pool stranding

- Consider replacing visual surveys with calibrated single-pass electrofishing to quantify stranding. This method would be most useful if a stratified random sampling was used in the site selection process. These recommendations will improve the accuracy and precision of stranding estimates and be more likely to provide the level of precision necessary to evaluate the influence of stranding risk factors (e.g. ramp rate, range and season).
- Consider measuring stage change in each reach but particularly above and below the reach 1-2 boundary. This would help assess the assumption that stranding risk is similar or lower in reach 1 compared to upstream reaches.

- Continue the post-ramp surveys 1-2 days after rampdowns until there is sufficient data to adequately predict which pools and side-channels connected on the ramp day eventually become isolated as well as the extent to which pool depth decreases over this period.

7.1.3 Mainstem abundance monitoring

- Continue with calibrated single-pass electrofishing to assess mainstem abundance at bar stranding sites. If bar stranding site selection uses as stratified random approach and electrofishing continues at each bar site, abundance estimates could be considered unbiased for the habitat types included in sampling.
- Consider expanding abundance sampling to all habitat types but particularly habitats at risk of becoming isolated during rampdowns. Abundance estimates for habitats prone to isolation are needed to understand whether abundance at bar stranding sites (typically shallow shorelines) adequately reflects relative abundance at pool stranding sites and whether this varies with season, species or age-class.
- Consider monthly abundance monitoring during periods without ramp monitoring or MON3 juvenile monitoring. Understanding how abundance and fish size changes seasonally will improve our ability to predict stranding throughout the year.

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9 Tables

Table 1. Relationship between rampdown WUP rampdown rates based on discharge and rates based on stage change at the Brackendale gauge on the Cheakamus River (WSC station 08GA043). Changes in stage and the stage-based ramp rate assume inflows from tributaries downstream remain constant during the rampdown. When local inflows are decreasing during a rampdown, as is typical for post-storm rampdowns, stage-based rates would be higher than reported in this table.

Daisy Lake Discharge (m^3/s)	Start Discharge (m^3/s)	End Discharge (m^3/s)	WUP rate ($m^3/s/hr$)	Start Stage (m)	End Stage (m)	Change in Stage (cm)	Duration of rampdown (hrs)	Ramp Rate (cm/hr)
> 62[1]	150	100	78	2.01	1.70	30.9	0.64	48.3
> 62[1]	100	80	78	1.70	1.55	15.0	0.26	58.3
10-62	100	80	13	1.70	1.55	15.0	1.54	9.7
10-62	80	60	13	1.55	1.38	17.4	1.54	11.3
10-62	60	38	13	1.38	1.14	23.7	1.69	14.0
10-62	38	20	13	1.14	0.88	26.5	1.38	19.2
< 10	20	15	1	0.88	0.78	9.8	5.00	2.0

[1] WUP rampdown rate is 13 m^3/s per 10 minutes when discharge from Daisy Dam is above 62 m^3/s. While up to six 13 m^3/s decreases per hour (78 m^3/s) are permitted, the maximum was two 13 m^3/s decreases within one hour for any rampdown during 2018-2019.

Table 2. Recommended DFO rampdown rates by time of year (corresponding to juvenile life history stage) and time of day (Cathcart 2005).

Time of year	Life Stage History	Day Ramp Rate	Night Ramp Rate
April 1- July 31	Fry Emergence	0 - 2.5 cm/h	2.5 - 5 cm/h
August 1 - October 31	Rearing until temp < 5 °C	0 - 2.5 cm/h	5 - 10 cm/h
November 1 - March 31	Winter Rearing	0 cm/h	0 - 5 cm/h

Table 3. Hypothesized fish stranding risk categories on the lower Cheakamus River based on discharge level and season (CASP 2018).

	Season	
	15-Feb. to 14-Oct.	15-Oct. to 14-Feb.
Discharge Range	Fry emergence, downstream migration, rearing	Adult spawning, egg incubation, and juvenile O/W
15.5-21	High	Moderate
21-42	High	Moderate
42-80	Moderate	Low
80-110	Moderate	Low
>110	Low	Low

Table 4. Relative fish stranding risk levels for high risk bar habitats as proposed in CASP (2018). Stranding risk was based on the stranding density estimated as the number of stranded fish per square meter of dewatered habitat at high risk stranding sites.

	Stranded Fish Density (fish/m^2)		
	<0.02	0.02-0.1	>0.1
Risk	Low	Moderate	High

Table 5. Relative fish stranding risk in isolated pools in lower Cheakamus River as proposed in CASP (2018). Stranding risk was based on the number of pools of a given risk level. Risk varied by observed fish abundance and residual pool depths. A rampdown was considered high risk if one or more pools were in the high range of the risk matrix and moderate risk with five or more pools in the moderate risk range. Ramp risk was considered low if there were four or less pools in the moderate range and any number of pools in the low range.

	Estimated Number of Fish in Isolated Pool				
Pool Size	0	1-10	11-100	101-1,000	1,001-10,000
Small/Dry	Low	Moderate	Moderate	High	High
Medium/Shallow	Low	Low	Moderate	Moderate	High
Large/Deep	Low	Low	Low	Moderate	Moderate

Table 6. Cheakamus River flow conditions and rampdown characteristics for each monitored rampdown. All flow measures were based on levels at the WSC Brackendale gauge (08GA043). Water temperature was measured at the Suspension Bridge using at data logger recording on 1-hour intervals that were averaged for each ramp day. Ramp rates were the average rate over the period of sustained rapid flow change at the Brackendale gauge and may differ from modeled values.

Date	Peak flow previous 48 hrs	Rampdown range (m^3/s)	Ramp rate (cm/h)	Ramp Magnitude (m^3/s)	Flow over prior 5 days	Water temp (°C)	Mitigation: Ramp rate	Mitigation: Magnitude
20-Aug-18	40	40-21	8.3	19	stable	14.5	none	none
22-Oct-18	80	65-20	9.9	45	variable	8.8	none	none
06-Nov-18	200	80-40	9.5	40	variable	8.2	none	3 ramps over 3 day
08-Nov-18	80	40-20	9.5	20	variable	6.5	none	3 ramps over 3 day
09-Nov-18	20	20-15	1.1	5	variable	6.6	none	3 ramps over 3 day
29-Nov-18	200	130-40	11.8	90	variable	5.8	none	none
01-May-19	56	50-45	2.5	5	stable	7.8	none	none
17-May-19	165	125-90	16.0	35	stable	8.3	none	none
04-Jun-19	168	130-100	4.7	30	stable	10.2	lower rate (5cm/h)	none
20-Jun-19	105	100-67	3.6	33	stable	12.3	lower rate (5cm/h)	none
12-Jul-19	75	75-47	4.8	28	stable	12.9	lower rate (5cm/h)	none
09-Aug-19	45	45-31	4.6	14	stable	14.2	lower rate (5cm/h)	3 ramps over 12 days
15-Aug-19	33	33-26.5	2.4	7	stable	14.2	lower rate (2.5cm/h)	3 ramps over 12 days
20-Aug-19	26	25-17	1.8	8	stable	14.3	lower rate (2.5cm/h)	3 ramps over 12 days
19-Sep-19	68	60-44	3.9	16	variable	12.0	lower rate (5cm/h)	2 ramps over 2 days
20-Sep-19	68	43-22	2.0	21	variable	11.8	lower rate (2.5cm/h)	2 ramps over 2 days

Table 7. Sum of electrofishing captures, site length, and water temperature from mainstem sites adjacent to bar stranding sites sampled one to several days after each rampdown.

Date	Temperature (°C)	Shoreline sampled (m)	Electrofishing captures		
			Chinook fry	Coho fry	Steelhead fry
19-Aug-18	14.5	176	0	24	695
23-Oct-18	8.6	180	0	4	278
12-Nov-18	6.0	180	2	1	223
03-Dec-18	4.3	180	20	0	46
02-May-19	7.2	180	202	108	45
21-May-19	8.4	177	43	94	17
04-Jun-19	10.2	206	32	180	0
21-Jun-19	11.6	210	3	420	0
17-Jul-19	13.4	180	7	184	491
08-Aug-19	14.1	174	2	159	983
14-Aug-19	14.1	180	0	161	841
19-Aug-19	14.4	176	2	89	653
21-Sep-19	11.8	180	0	32	530

Table 8. Abundance estimates for mainstem sites adjacent to bar stranding sites electrofished either the day before or one to several days after each rampdown and capture efficiency values used to expand captures. Capture efficiency values for Steelhead fry for temperatures >5° were based on values from CMSMON3. Capture efficiency values for Coho and chinook fry and Steelhead fry <5° were from a small number of mark-recapture trials completed winter and spring 2019 and should be considered preliminary. Shading distinguishes between Cohorts that emerged primarily during 2018 or 2019.

Date	Temperature (°C)	Shoreline sampled (m)	Abundance for all sites combined		
			Chinook fry	Coho fry	Steelhead fry
19-Aug-18	14.5	176	0	104	2172
23-Oct-18	8.6	180	0	17	869
12-Nov-18	6.0	180	6	4	697
03-Dec-18	4.3	180	56	0	144
02-May-19	7.2	180	561	831	141
21-May-19	8.4	177	119	723	53
04-Jun-19	10.2	206	89	1385	0
21-Jun-19	11.6	210	8	3231	0
17-Jul-19	13.4	180	19	1415	1534
08-Aug-19	14.1	174	6	691	3072
14-Aug-19	14.1	180	0	700	2628
19-Aug-19	14.4	176	6	387	2041
21-Sep-19	11.8	180	0	139	1656
		Capture efficiency used for abundance estimates			
Fall	>5		21%	23%	32%
Winter	< 5		21%	23%	26%
Spring	>5		36%	13%	32%

Table 9. Estimates of the pre-ramp linear density for mainstem sites adjacent to bar stranding sites electrofished either before or one to several days after each rampdown for chinook, Coho and Steelhead age-0 fry as well as for all combined. Shading distinguishes between Cohorts that emerged primarily during 2018 or 2019. To estimate pre-ramp density if electrofishing occurred post-rampdown, the total numbers of stranded fish were added to the abundance estimates for each rampdown event.

	Estimated pre-ramp density (fish/m)			
Ramp date	Chinook fry	Coho fry	Steelhead fry	Combined
20-Aug	0.00	0.59	12.34	12.93
22-Oct	0.00	0.17	4.97	5.14
06-Nov	0.03	0.15	3.87	4.05
08-Nov	0.03	0.02	3.87	3.93
09-Nov	0.03	0.02	3.87	3.93
29-Nov	0.39	0.00	0.80	1.19
01-May	3.12	4.64	0.78	8.54
17-May	0.76	5.07	0.32	6.14
04-Jun	0.43	6.73	0.00	7.16
20-Jun	0.04	15.43	0.00	15.47
12-Jul	0.11	7.94	8.54	16.59
09-Aug	0.04	3.98	17.82	21.84
15-Aug	0.00	3.90	14.66	18.55
20-Aug	0.03	2.20	11.63	13.86
19-Sep	0.00	0.79	9.27	10.05
20-Sep	0.00	0.77	9.25	10.02

Table 10. Mean forklength of chinook, Coho and Steelhead fry captures at mainstem sites adjacent to bar survey sites. Shading distinguishes between Cohorts that emerged primarily during 2018 or 2019. After June 1, the 2018 Cohort of Steelhead fry were categorized as 1+parr.

	Electroshocking		
	Mean Forklength (mm)		
Species	Chinook fry	Coho fry	Steelhead fry
19-Aug-18	-	38	31
21-Aug-18	-	39	33
23-Oct-18	-	61	51
12-Nov-18	35	54	52
03-Dec-18	37	-	53
02-May-19	40	35	64
21-May-19	43	34	69
04-Jun-19	51	37	-
21-Jun-19	60	35	-
17-Jul-19	68	46	34
08-Aug-19	70	49	34
14-Aug-19	-	49	37
19-Aug-19	55	52	38
21-Sep-19	-	59	52

Table 11. Summary of bar sampling surveys for each monitored rampdown including the total dewatered area for all sites, area sampled, proportion of total site area sampled (sampled areas/site area), the count stranded fish found and the expanded total number stranded in sites (count / proportion sampled).

Date	Site area	Sampled area	Sampled proportion	Counts of stranded			Expanded total fish stranded at sites		
	m^2	m^2		Chinook fry	Coho fry	Steelhead fry	Chinook fry	Coho fry	Steelhead fry
20-Aug-18	851	150	0.18	0	0	46	0	0	261
22-Oct-18	2006	151	0.08	0	1	2	0	13	27
06-Nov-18	1940	176	0.09	0	2	0	0	22	0
08-Nov-18	956	168	0.18	0	0	0	0	0	0
09-Nov-18	180	162	0.90	0	0	0	0	0	0
29-Nov-18	4006	282	0.07	1	0	0	14	0	0
01-May-19	100	85	0.84	0	4	0	0	5	0
17-May-19	726	197	0.27	4	47	1	15	173	4
04-Jun-19	706	170	0.24	0	0	0	0	0	0
20-Jun-19	1593	175	0.11	0	1	0	0	9	0
12-Jul-19	777	225	0.29	0	4	1	0	14	3
09-Aug-19	614	446	0.73	1	1	22	1	1	30
15-Aug-19	444	358	0.81	0	1	8	0	1	10
20-Aug-19	309	271	0.88	0	0	6	0	0	7
19-Sep-19	502	203	0.40	0	1	1	0	2	2
20-Sep-19	903	203	0.22	0	0	2	0	0	9

Table 12. Relative stranding risk based on the average stranding density (fish stranded per m^2 of dewatered bar) at the six high-risk bar stranding sites for rampdowns August 2018-September 2019. Flow characteristics of are based on the Brackendale gauge on the Cheakamus River (WSC station 08GA043). Ramp rate is the average sustained rate. Mainstem fish density was based on one-pass electrofishing of the mainstem shoreline adjacent to each of the six bar sampling sites either one day prior or one to three days after each rampdown.

				Fish abundance pre-ramp (fish/m)			Stranding density (fish/m^2)		
Date	Rampdown range (m^3/s)	Ramp rate (cm/h)	Ramp magnitude (m^3/s)	Chinook fry	Coho fry	Steelhead fry	Chinook fry	Coho fry	Steelhead fry
20-Aug-18	40-21	8.3	19	0.00	0.59	12.34	0.000	0.000	0.307
22-Oct-18	65-20	9.9	45	0.00	0.17	4.97	0.000	0.007	0.013
06-Nov-18	80-40	9.5	40	0.03	0.15	3.87	0.000	0.011	0.000
08-Nov-18	40-20	9.5	20	0.03	0.02	3.87	0.000	0.000	0.000
09-Nov-18	20-15	1.1	5	0.03	0.02	3.87	0.000	0.000	0.000
29-Nov-18	130-40	11.8	90	0.39	0.00	0.80	0.004	0.000	0.000
01-May-19	50-45	2.5	5	3.12	4.64	0.78	0.000	0.047	0.000
17-May-19	125-90	16.0	35	0.76	5.07	0.32	0.020	0.239	0.005
04-Jun-19	130-100	4.7	30	0.43	6.73	0.00	0.000	0.000	0.000
20-Jun-19	100-67	3.6	33	0.04	15.43	0.00	0.000	0.006	0.000
12-Jul-19	75-47	4.8	28	0.11	7.94	8.54	0.000	0.018	0.004
09-Aug-19	45-31	4.6	14	0.04	3.98	17.82	0.000	0.002	0.049
15-Aug-19	33-26.5	2.4	7	0.00	3.91	14.74	0.000	0.007	0.058
20-Aug-19	25-17	1.8	8	0.03	2.20	11.64	0.000	0.000	0.027
19-Sep-19	60-44	3.9	16	0.00	0.79	9.27	0.000	0.000	0.005
20-Sep-19	43-22	2.0	21	0.00	0.77	9.25	0.000	0.000	0.010

Table 13. Comparison of the bar stranding for the scheduled reduction in minimum flows for August 2018 and 2019.

Date	Rampdown range	Stranding density		Fish density	Percent stranded
	cms	fish/m^2	fish/m	fish/m	stranded/present
20-Aug-18	40-21	0.320	1.55	12.93	12%
9-20 Aug 19	45-17	0.052	0.393	21.84	1-2%

Table 14. The minimum and maximum numbers of fish observed in isolated pools based order-of-magnitude visual estimates following rampdowns. For consistency, pool lengths and fish observations are for the reaches 2-5 (river km 4-16) even though some surveys continued to the confluence with the Squamish River. Ramp range, magnitude and rate reflect changes at the Brackendale gauge on the Cheakamus River (WSC 08GA043). Note that numbers observed are likely not comparable across wide differences in water temperature due to increased concealment at low temperatures that reduces observer efficiency. For flow reductions that included more than one rampdown, pools isolated on prior ramps were not included in for assign the risk level for individual ramps.

Date	Water temperature (°C)	Ramp range (m^3/s)	Ramp rate (cm/h)	Ramp magnitude (m^3/s)	Fish observed		Total length isolated (m)	Mainstem abundance (fish/m)
					Minimum	Maximum		
20-Aug-18	14.5	40-21	8.3	19	3,492	34,840	1,650	12.9
22-Oct-18	8.8	65-20	9.9	45	1,468	14,650	1,637	5.1
06-Nov-18	8.2	80-40	9.5	40	37	370	805	4.1
08-Nov-18	6.5	40-20	9.5	20	116	1,160	839	3.9
09-Nov-18	6.6	20-15	1.1	5	30	300	584	3.9
29-Nov-18	5.8	130-40	11.8	90	20	200	639	1.2
01-May-19	7.8	50-45	2.5	5	314	3,140	143	8.5
17-May-19	8.3	125-90	16.0	35	1,232	12,320	250	6.1
04-Jun-19	10.2	130-100	4.7	30	100	1,000	68	7.2
20-Jun-19	12.3	100-67	3.6	33	289	2,890	443	15.5
12-Jul-19	12.9	75-47	4.8	28	4	40	14	16.6
09-Aug-19	14.2	45-31	4.6	14	2,392	23,920	527	21.8
15-Aug-19	14.2	33-26.5	2.4	7	1,492	14,920	314	18.7
20-Aug-19	14.3	25-17	1.8	8	924	9,240	262	13.9
19-Sep-19	12.0	60-44	3.9	16	2	20	810	10.1
20-Sep-19	11.8	43-22	2.0	21	156	1,570	1,039	10.0

Table 15. The mean length of pool and side-channel habitat isolated in reaches 2-5 of the Cheakamus River based on post-rampdown discharge for rampdowns from August 2018 to September 2019 (Brackendale gauge station). The count is the number of rampdowns used to estimate the mean and CV is the coefficient of variation.

Post ramp discharge (m^3/s)	Mean length isolated (m)	Count	CV
15-24	1142	6	0.38
25-34	684	2	0.33
35-44	751	3	0.13
45-54	79	2	1.16
55-64	-	-	-
65-74	443	1	-
75-84	-	-	-
85-94	250	1	-
95-104	68	1	-

Table 16. The maximum number of juvenile fish observed in isolated pools/side-channels by depth class and flow reduction events used to assess the benefit of one or more mitigation measures. Surveys included all isolated pool/side-channel on the Cheakamus River between Road's End (~km 16) and Fergie's Bridge (~km 4).

					Maximum number of fish Pool Depth (cm)			
Date	Ramp range (m^3/s)	Ramp rate (cm/h)	Number of ramps	Days for reduction	<25	25-50	>50	Total
20-Aug-18	40-21	8.31	1	1	6,540	2,100	26,200	34,840
9-20, Aug 19	45-17	1.4-4.6	3	12	2,480	8,900	36,700	48,080
17-May-19	125-90	16.0	1	1	100	10,120	2,100	12,320
04-Jun-19	130-100	4.7	1	1	0	1,000	0	1,000
22-Oct-18	65-20	9.9	1	1	390	2,140	12,120	14,650
6-8 Nov	80-40	9.5	2	3	230	180	1,130	1,540

Table 17. The total maximum number of juvenile fish observed in isolated pools/side-channels for the Cheakamus River in reach 1 (Fergie's Bridge - Squamish River confluence) and reach 2-5 (Road's End to Fergie's Bridge). Surveys downstream of Fergie's Bridge were suspended following the May 1 survey in part due to the low observed abundance compared to reaches 2-5 prior to this. Surveys were reinstituted in reach 1 September 2019 in response to questions about relative impacts in this reach.

	Maximum number of fish observed		
Date	Reach 1	Reach 2-5	% of Reach 2-5
06-Nov-18	0	370	0%
08-Nov-18	10	1170	1%
09-Nov-18	10	550	2%
29-Nov-18	10	200	5%
01-May-19	10	3150	0%
19-Sep-19	10	30	33%
20-Sep-19	100	1570	6%

Table 18. Mean and coefficient of variation of observer efficiency (CV) of visually estimating fish abundance in pools across a range of water temperatures. CV represents the relative precision of observer efficiency estimates based on the variation among individual estimates. Smaller CV values represent higher precision.

Month	Average Temperature (°C)	Mean observer efficeiency	CV of observer efficiency	Number of sites
March	4.6	6%	1.95	4
April	7.4	12%	0.61	4
June	13.1	49%	0.36	5
August	17.6	24%	0.70	8
September	15.8	38%	0.85	4

Table 19. Backpack electrofishing base mark-recapture estimates of abundance in pools, the visual estimates of the number if fish visible prior to the recapture pass and the observer efficiency for each trial.

Date	Site	Area (m^2)	Max depth (cm)	Temp (C)	N	Observed fish	Observer efficiency
15-Mar-19	485	26	32	5.3	120	30	25%
15-Mar-19	490	192	35	5.2	356	0	0%
15-Mar-19	511	90	30	3.5	187	0	0%
15-Mar-19	520	66	35	4.4	220	1	0%
12-Apr-19	205.1	81	60	7.4	401	35	9%
12-Apr-19	205.2	51	55	7.4	414	15	4%
12-Apr-19	294.1	9	20	7.4	76	15	20%
12-Apr-19	294.2	54	45	7.4	283	50	18%
11-Jun-19	117	91	20	16.7	610	140	23%
11-Jun-19	520.1	3	20	12.3	91	40	44%
11-Jun-19	520.2	4	20	11.9	128	70	55%
11-Jun-19	520.3	19	30	12.3	153	110	72%
11-Jun-19	520.4	9	25	12.3	137	70	51%
22-Aug-19	516.4	9	36	17.7	248	43	17%
22-Aug-19	516.43	13	32	17.7	142	50	35%
22-Aug-19	516.46	72	35	18.5	2078	435	21%
22-Aug-19	516.47	4	42	17.2	96	10	10%
22-Aug-19	516.48	3	8	17.7	48	22	47%
22-Aug-19	516.49	0	10	17.2	14	7	50%
23-Aug-19	480	135	80	15.8	305	27	9%
23-Aug-19	490	221	67	15.8	892	61	7%
12-Sep-19	516.43	8	10	13.2	28	24	86%
12-Sep-19	516.47	2	40	13.2	39	8	21%
11-Sep-19	480	364	80	12.3	159	35	22%
11-Sep-19	490	188	35	12.3	468	107	23%

Table 20. Average survival for Coho and Steelhead age-0 fry in isolated pools from Aug 22-23 to September 11-12 by maximum pool depth. CV reflects the variation in survival rates between sites. Count is the number of sites used for each estimate.

		Survival		
Species	Max depth (cm)	Mean	cv	Count
Coho	0-25	-	-	0
Coho	26-50	53%	0.30	4
Coho	>50	22%	1.41	2
Steelhead	0-25	0%	0.00	2
Steelhead	26-50	1%	2.00	4
Steelhead	>50	43%	0.24	2

Table 21. The count of adult salmon (Chum and Pink) and redds stranded on bars or in isolated pools. For the September 20, 2019 survey, the area of dewatered redds was estimated instead of the number of redds.

Date	Discharge change (cms)	Adults isolated (alive and dead)	Redds isolated and/or dewatered
22-Oct-18	80-20	0	0
06-Nov-18	80-40	7	3
08-Nov-18	40-20	0	0
09-Nov-18	20-15	11	6
29-Nov-18	130-40	12	0
01-May-19	50-45	0	0
17-May-19	125-90	0	0
04-Jun-19	130-100	0	0
20-Jun-19	100-67	0	0
12-Jul-19	72-46	0	0
09-Aug-19	42-27	0	0
15-Aug-19	29-23	0	0
20-Aug-19	23-17	0	0
19-Sep-19	60-44	18	6
20-Sep-19	43-22	2380	7743 m^2

Table 22. The range of abundance during late summer for Steelhead fry (2008-2018), and during spring outmigration for young-of-the-year chinook (2001-2018), Pink (2002-2018) and Chum (2007-2018).

	Abundance		
Species	Minimum	Maximum	Season
Steelhead fry	150,000	390,000	Late summer
Chinook YoY	16,000	800,000	Spring
Pink	80,000	30 million	Spring
Chum	2 million	12 million	Spring

Table 23. Estimates of the mean length of pool habitat (m) isolated when spill decreases from Daisy Dam result in end of ramp discharges at Brackendale within discrete 10 m^3/s bins. For an event to be counted as isolating pool habitat, discharge must cross from at least one discharge bin to another.

Post ramp discharge at Brackendale (m^3/s)	Mean length of isolated pools and side-channels (m)	Count of ramp events	Coefficient of variation
14-May	-	-	-
15-24	1,142	6	0.38
25-34	684	2	0.33
35-44	751	3	0.13
45-54	79	2	1.16
55-64	-	0	-
65-74	443	1	-
75-84	-	0	-
85-94	250	1	-
95-104	68	1	-
105-114	-	0	-
115-124	-	0	-

10 Figures

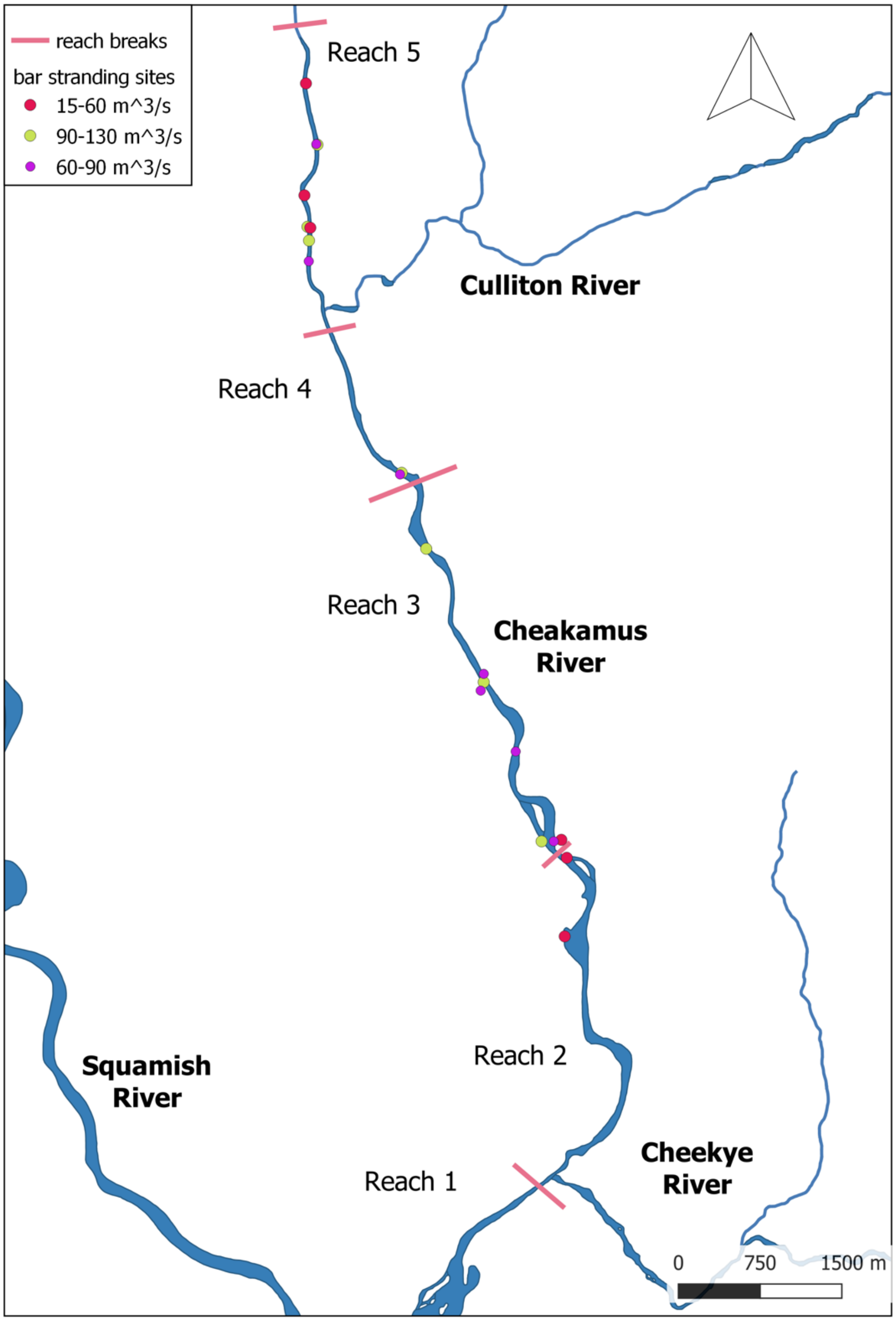


Figure 1. Map of the Cheakamus River study area showing the locations of the upstream limit of reach breaks (pink bars). The core stranding monitoring included reach 2-5. Reach 1 was not surveyed May 17 – August 20, 2019 due to relatively low amounts of pool stranding and the assumption that risk based on reaches 2-5 would reflect risk in reach 1.

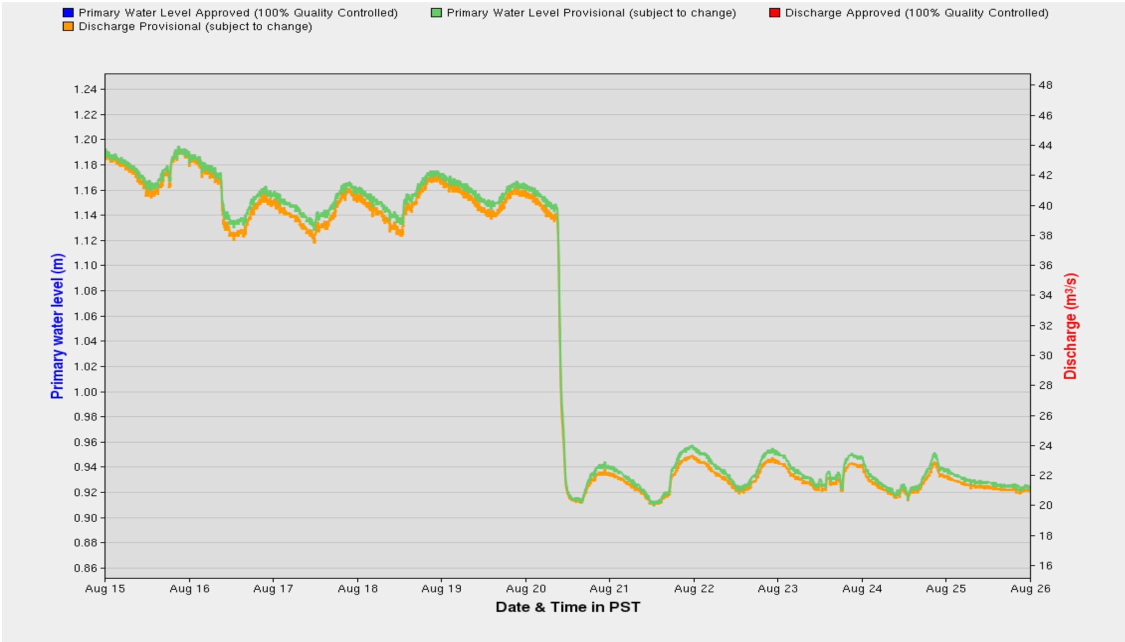


Figure 2. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the Aug 20, 2018 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

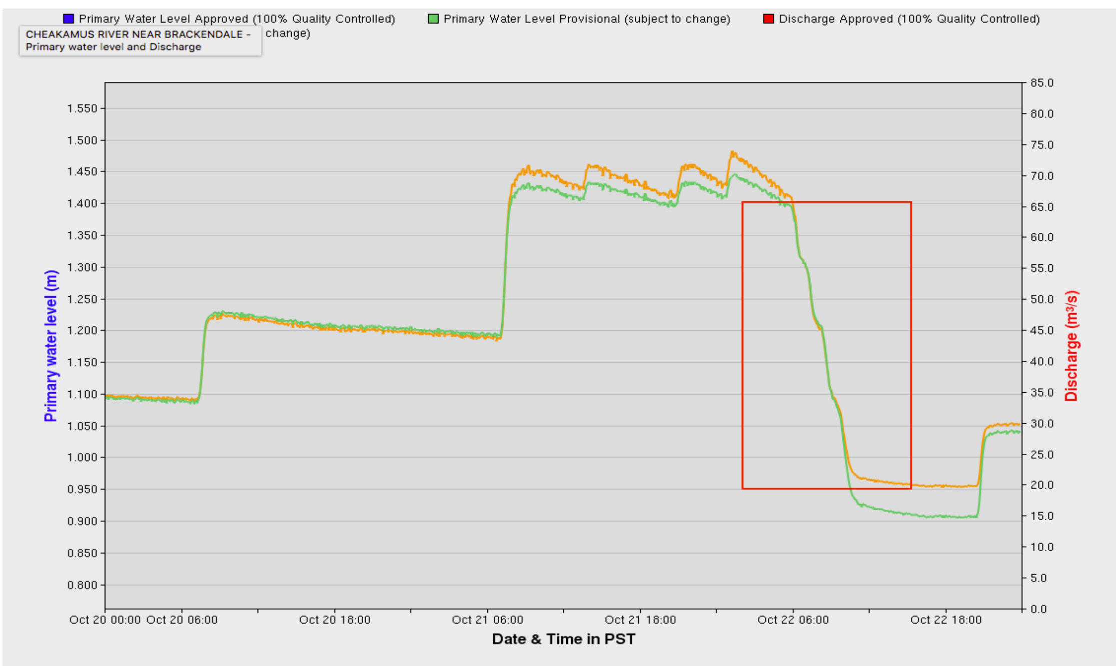


Figure 3. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the Oct 22, 2018 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

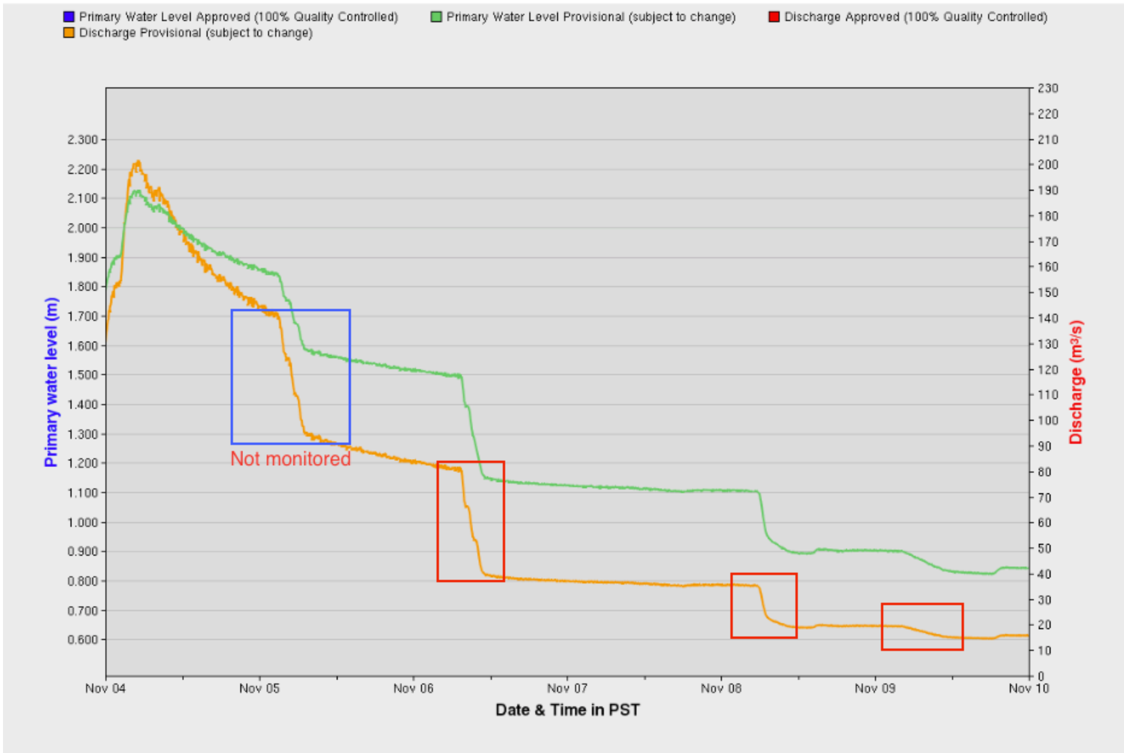


Figure 4. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the Nov 6-9, 2018 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

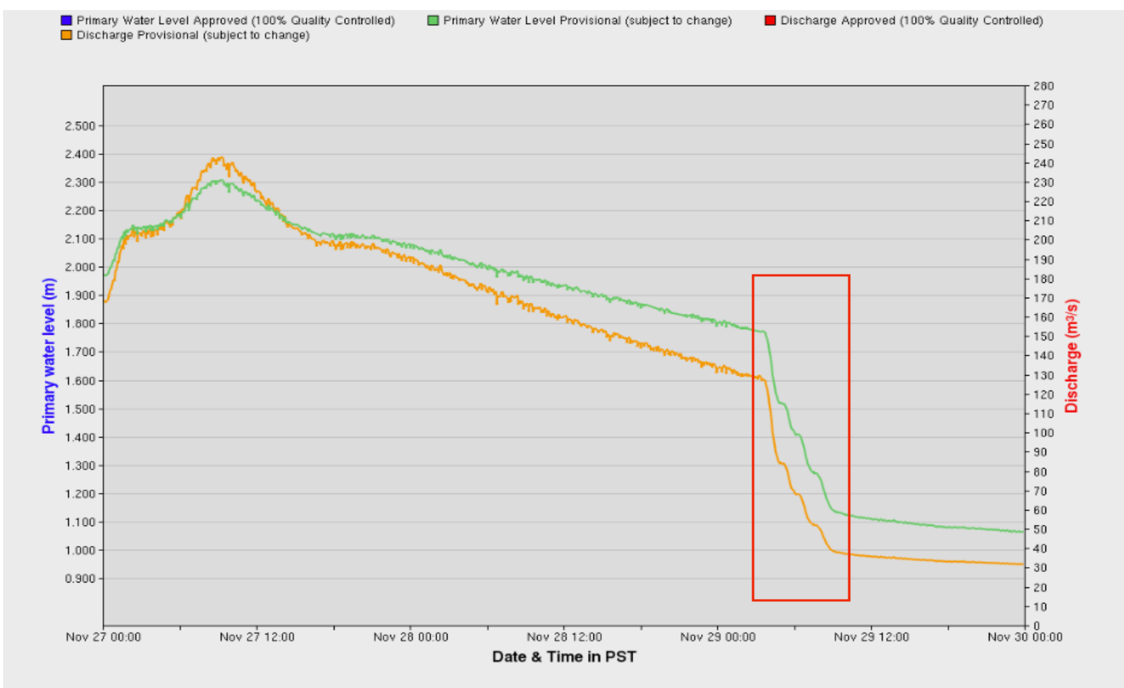


Figure 5. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the Nov 29, 2018 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

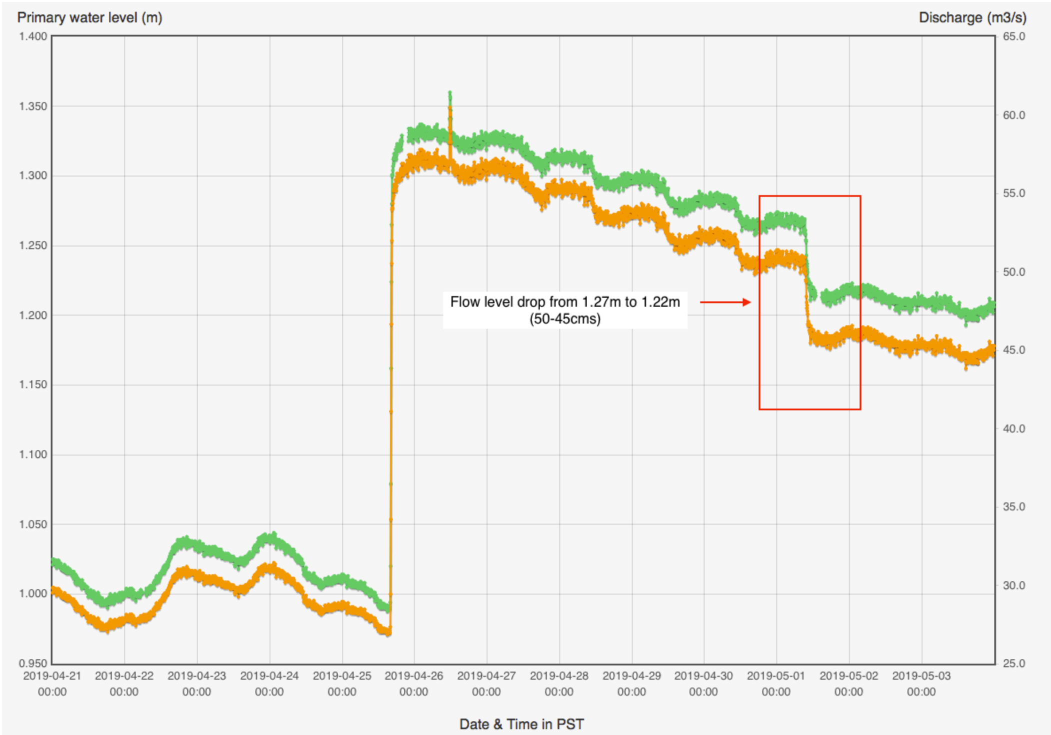


Figure 6. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the May1, 2019 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

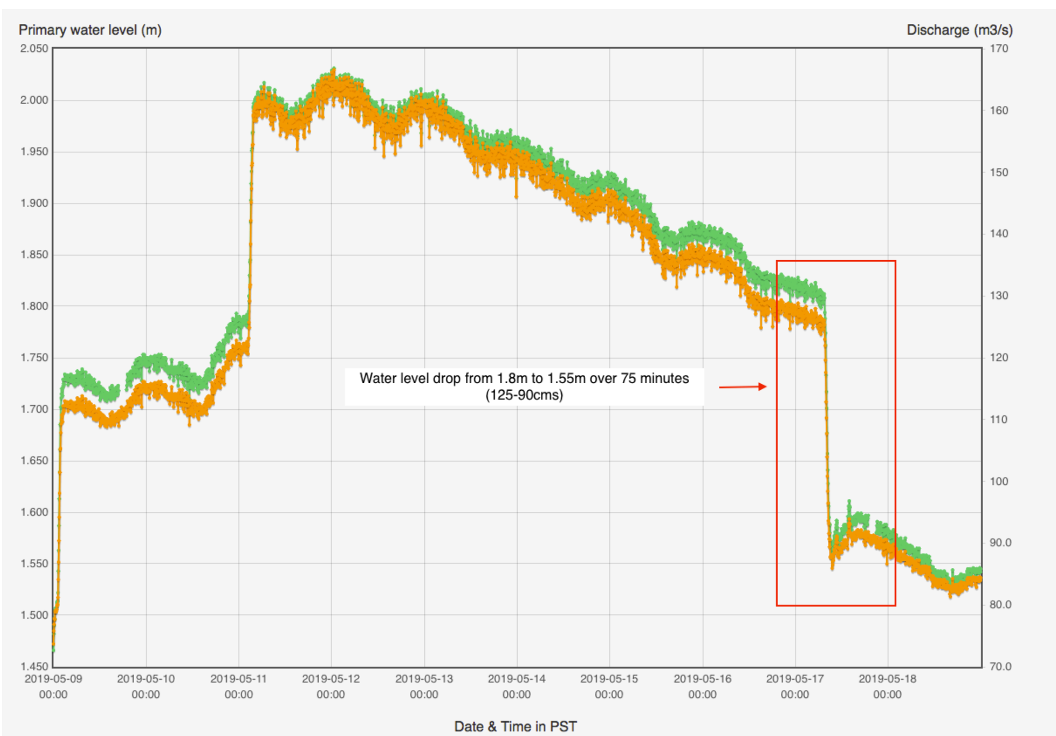


Figure 7. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the May17, 2019 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

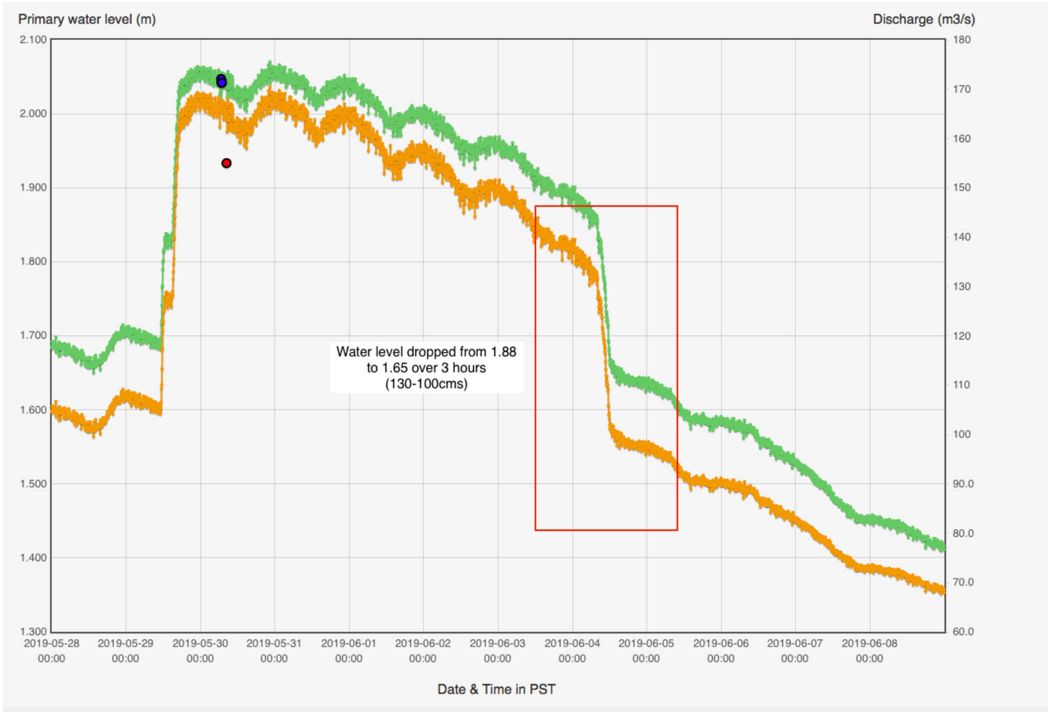


Figure 8. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the June 4, 2019 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

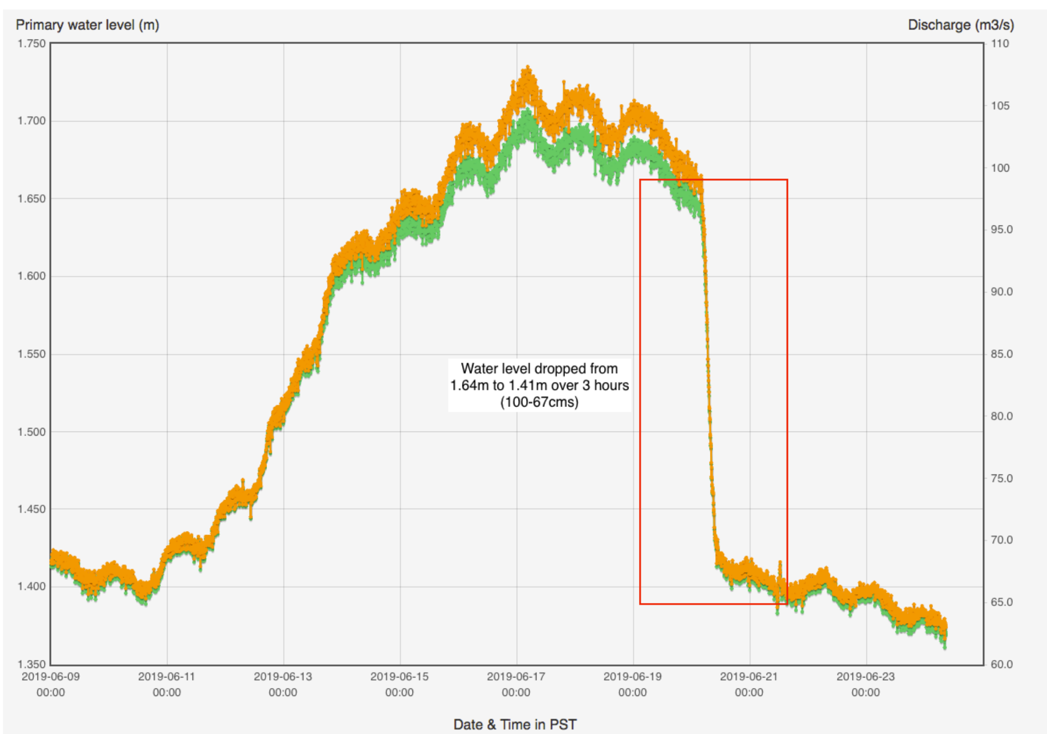


Figure 9. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the June 20, 2019 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

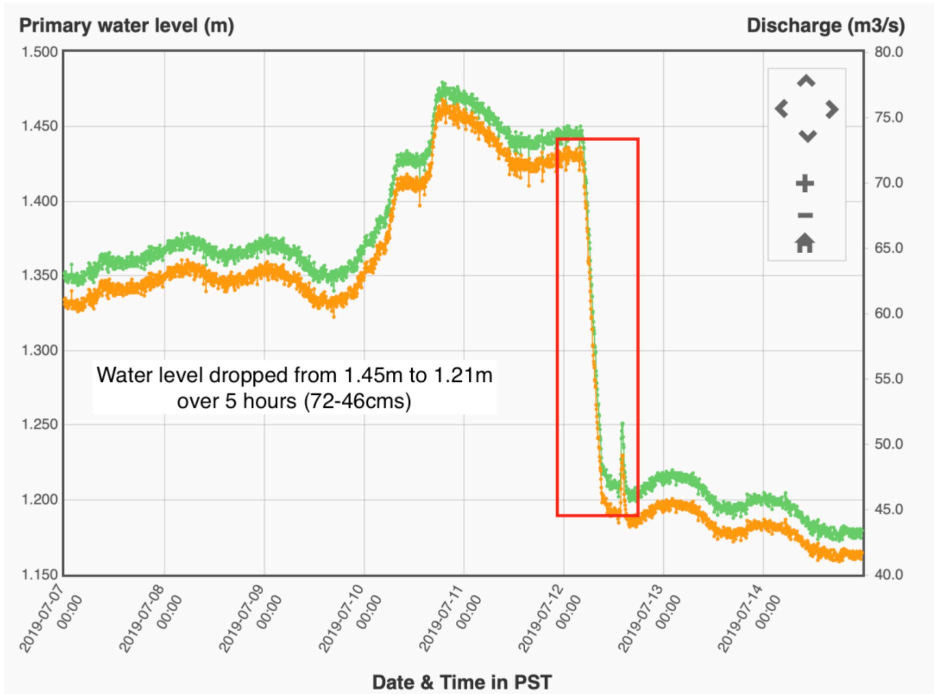


Figure 10. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the July 12, 2019 ramping event. Orange lines are discharge (m^3/s) and green lines are stage (m).

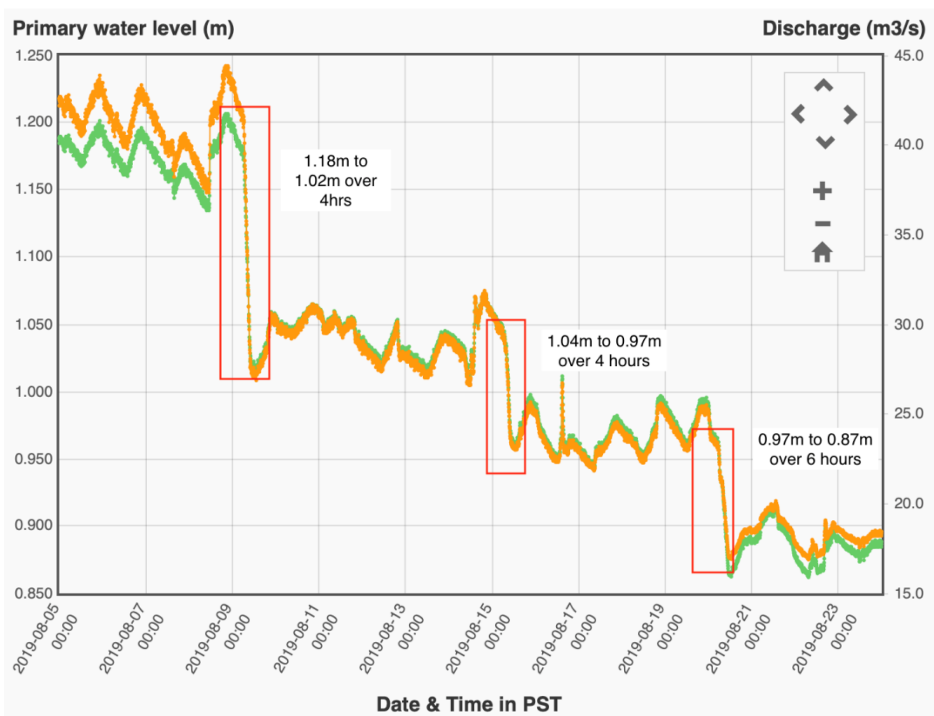


Figure 11. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the August 9-20, 2019 ramping events. Note that gauge readings on September 20, 2019 indicated flows did not fall below 20 m^3/s at Brackendale, however discharge was shifted downward after are discharge reading August 29, 2019. Orange lines are discharge (m^3/s) and green lines are stage (m).

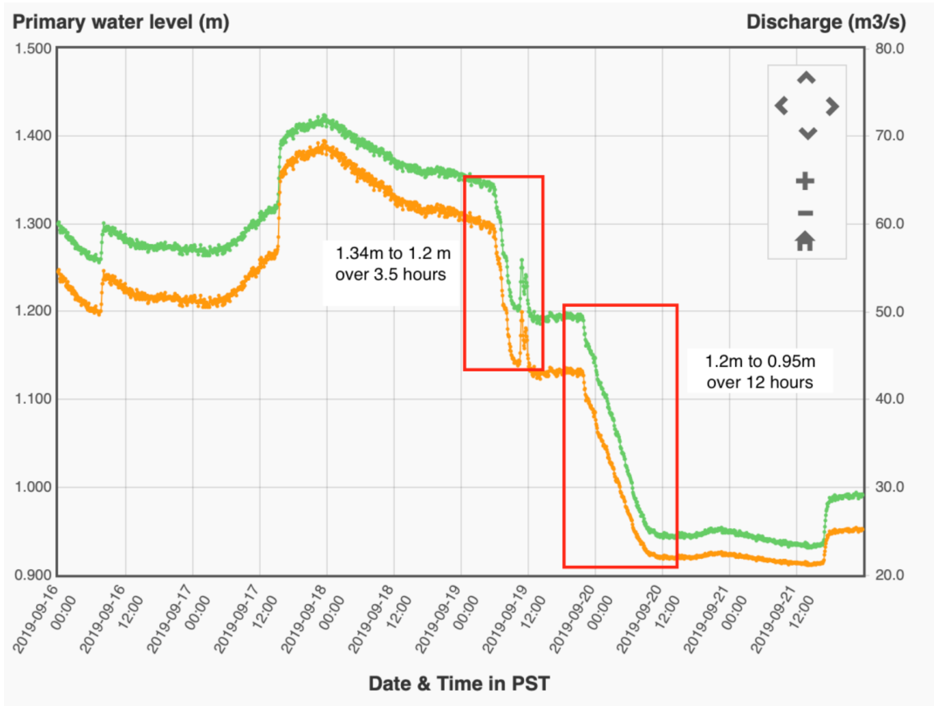


Figure 12. Hydrograph at the Brackendale gauge on the Cheakamus River before, during, and after the September 19-20, 2019 ramping events. Orange lines are discharge (m^3/s) and green lines are stage (m).

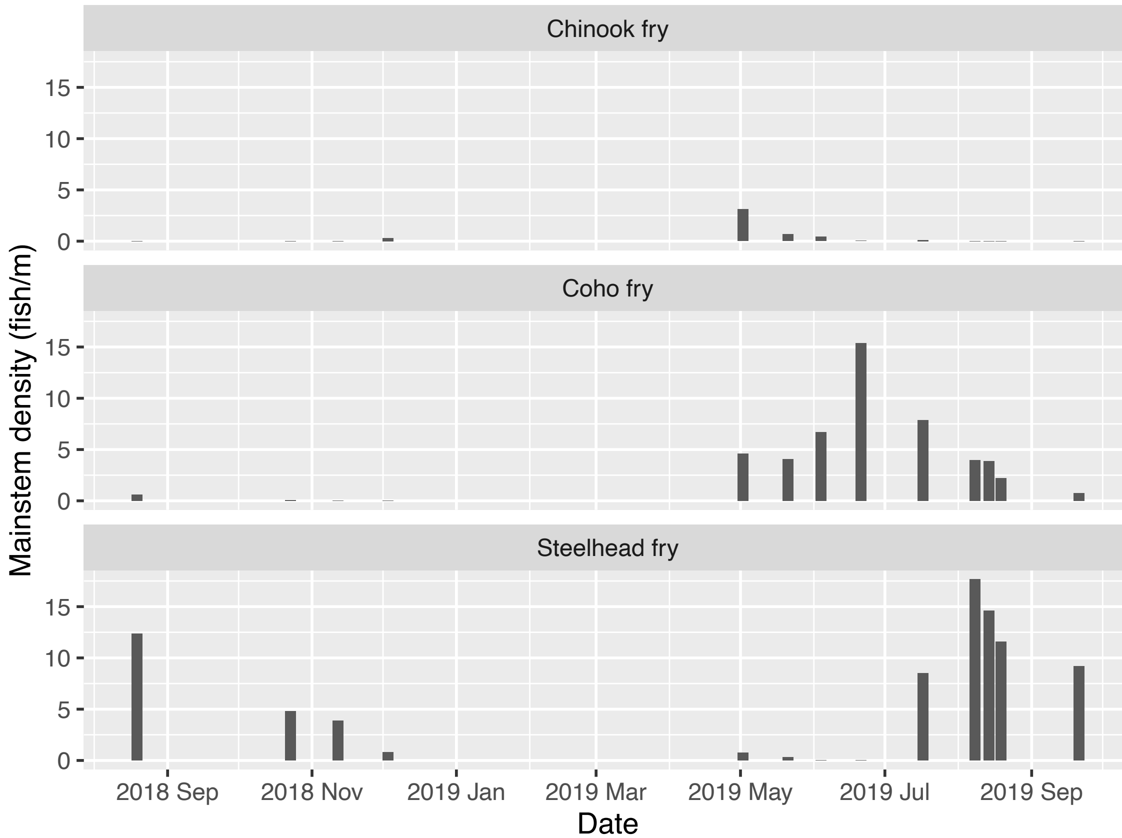


Figure 13. Mean density of Chinook, Coho and Steelhead age-0 fry in the mainstem at the six bar stranding sites. Estimates based on backpack electroshocking captures (Table 7) expanded by estimated capture efficiency (Table 8).

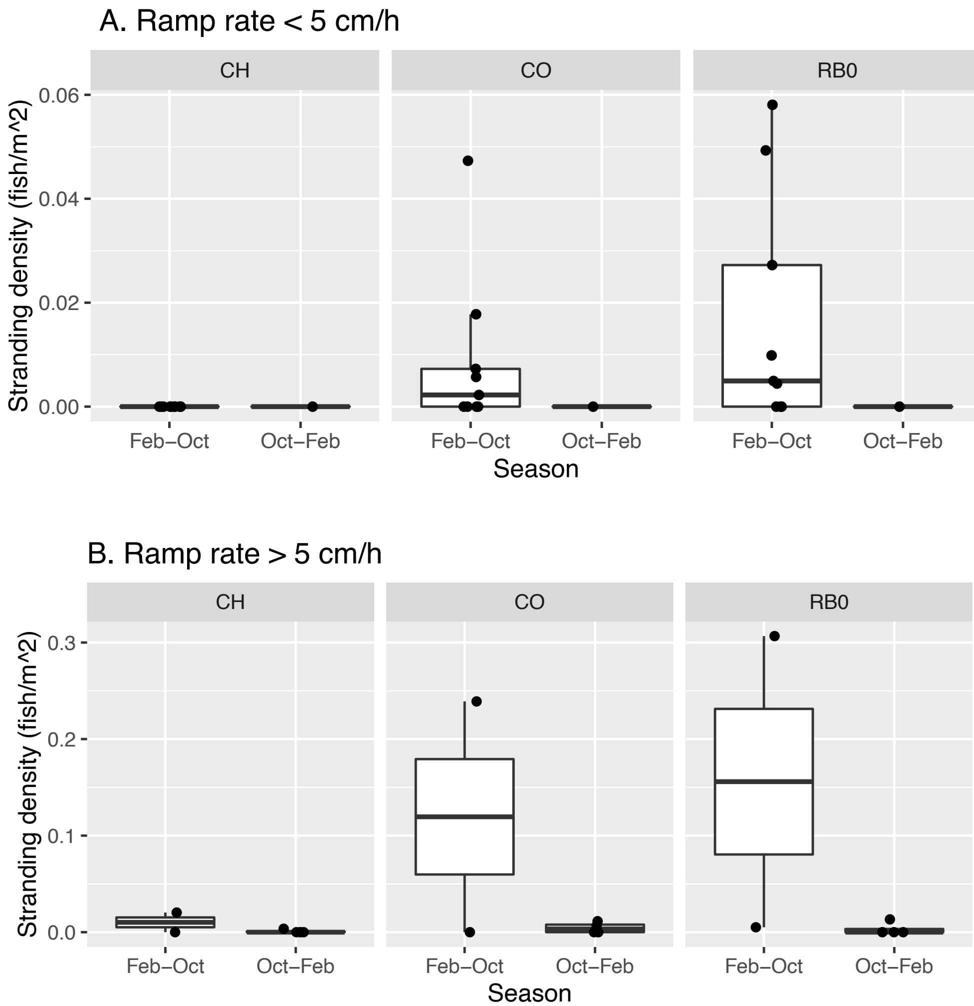


Figure 14a-b. Stranding density (fish stranded/m^2 dewatered bar) by season and species for rampdowns with ramp rate greater than 5 cm/h (A) and less than 5 cm/h (B). Species include Chinook fry (CH), Coho fry (CO) and Steelhead fry (RB0). Seasons correspond to hypothesised periods of higher and lower stranding risk: Feb-15 to Oct-14 and Oct-15 to Feb-14 (CASP 2018). The horizontal line in the boxplot represents the median value. Stranding densities and mainstem fish densities were based on sampling at six sites selected to represent high-risk stranding habitat.

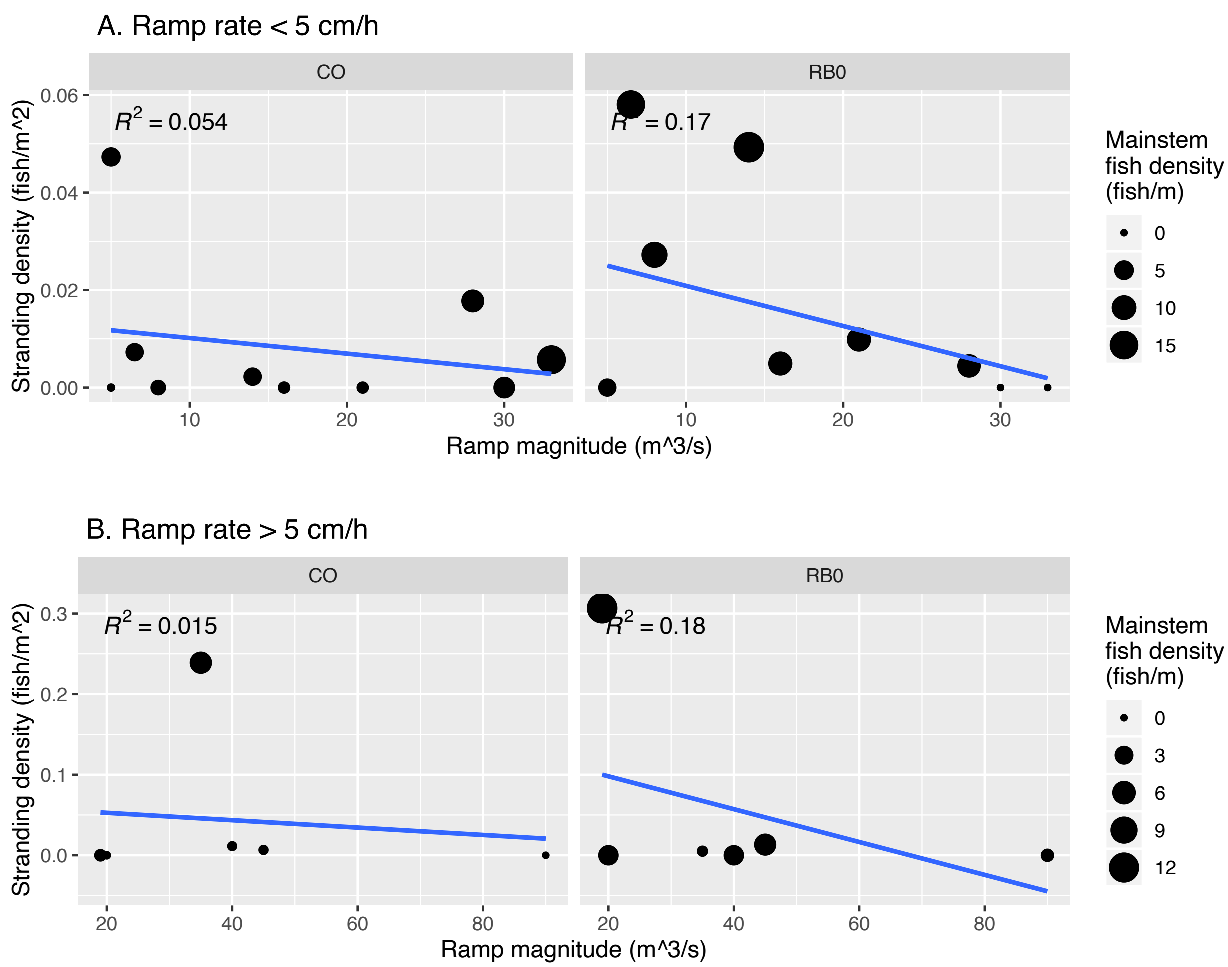


Figure 15a-b. Stranding density (fish stranded/m^2 dewatered bar) in relation to ramp magnitude (m^3/s) and mainstem fish density (fish/m) for Coho fry (CO) and Steelhead fry (RB0) for rampdowns completed with ramp rate greater than 5 cm/h (A) and less than 5 cm/h (B). Rampdowns occurred August 2018 and September 2019. Stranding densities and mainstem fish densities were based on sampling at six sites selected to represent high-risk stranding habitat.

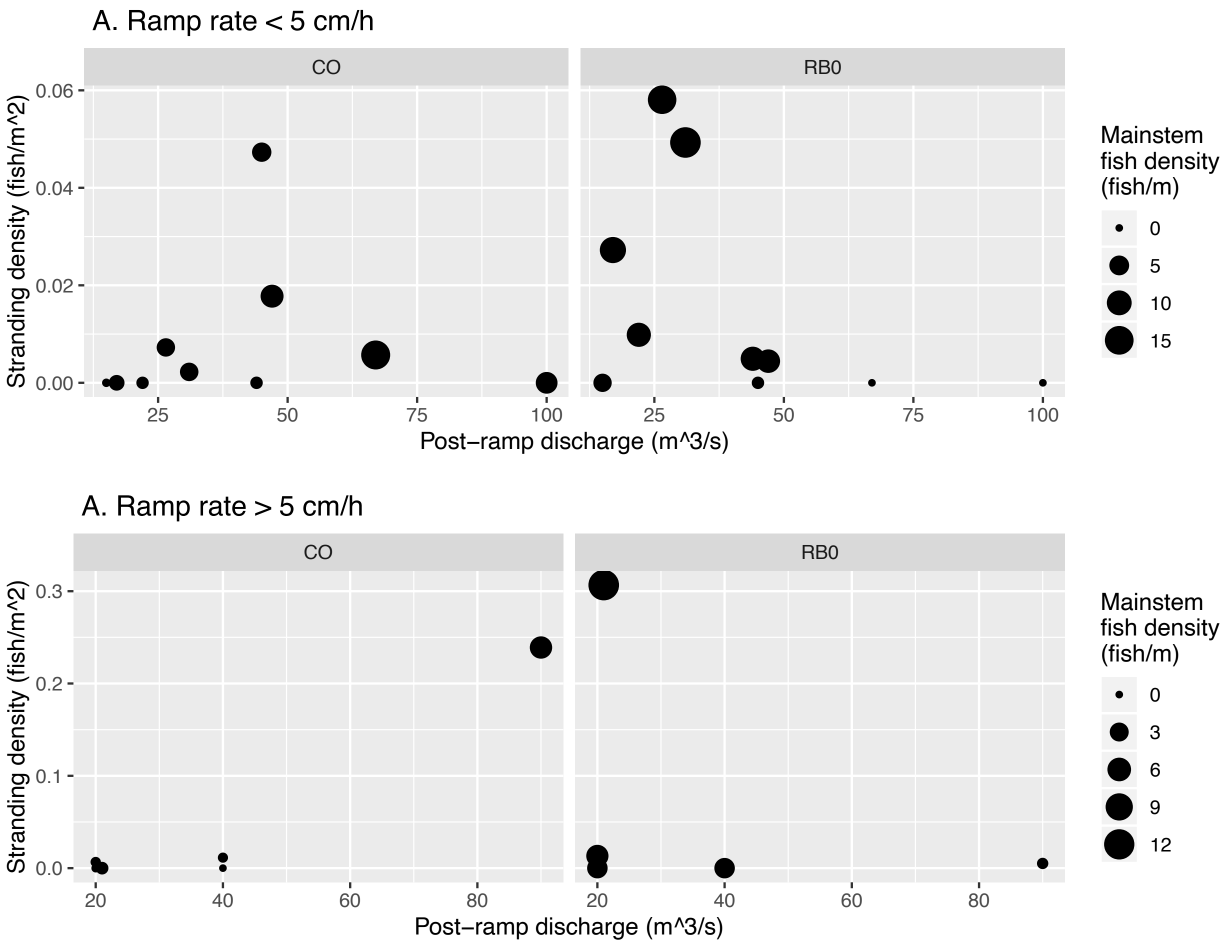


Figure 16a-b. Stranding density (fish stranded/m^2 dewatered bar) in relation to post-rampdown discharge (m^3/s) and mainstem fish density (fish/m) for Coho fry (CO) and Steelhead fry (RB0) for rampdowns completed with ramp rate greater than 5 cm/h (A) and less than 5 cm/h (B). Rampdowns occurred August 2018 and September 2019. Stranding densities and mainstem fish densities were based on sampling at six sites selected to represent high-risk stranding habitat.

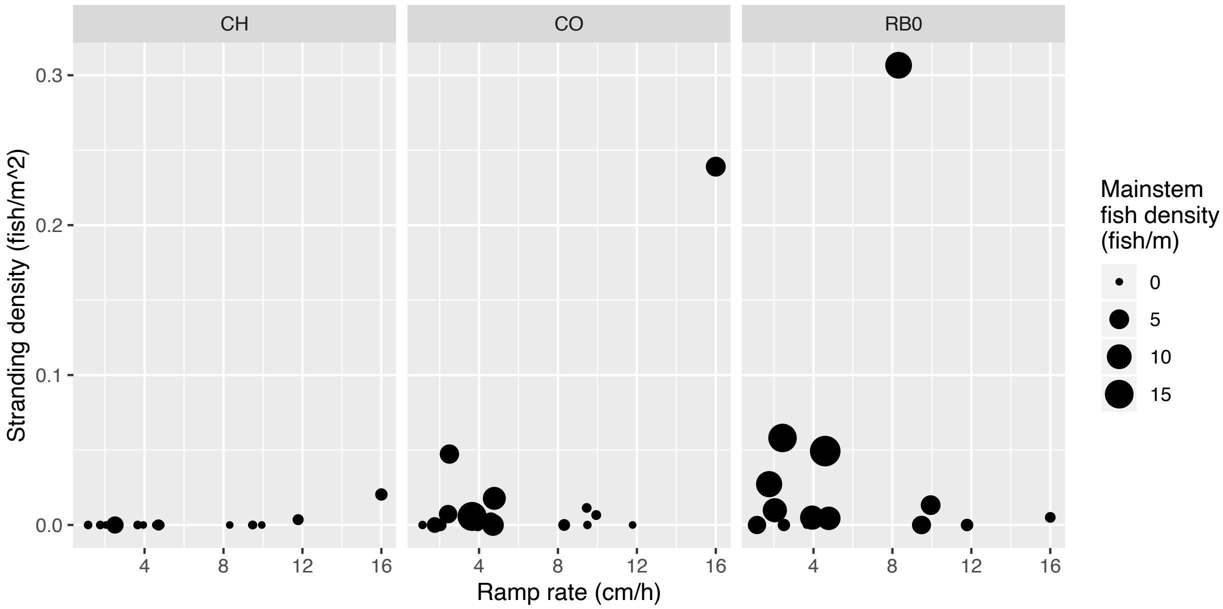


Figure 17. Stranding density (fish stranded/m^2 dewatered bar) in relation to ramp rate (cm/h) and mainstem fish density (fish/m) for Chinook fry (CH), Coho fry (CO) and Steelhead fry (RB0) for rampdowns completed between August 2018 and September 2019 based on sampling at six sites selected to represent high-risk stranding habitat. Stranding densities and mainstem fish densities were based on sampling at six sites selected to represent high-risk stranding habitat.

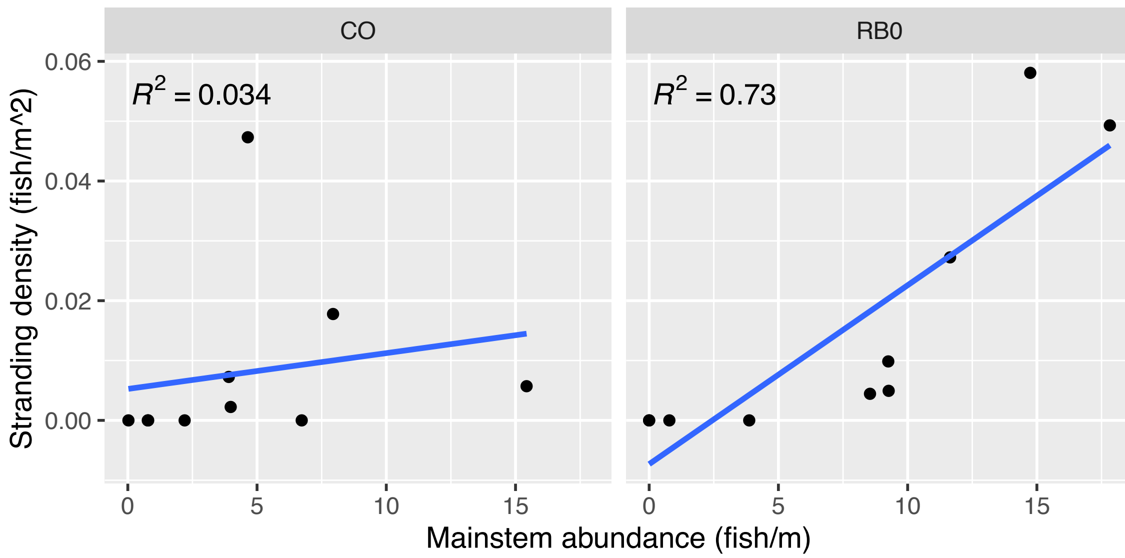


Figure 18. Stranding density (fish stranded/m^2 dewatered bar) in relation to mainstem fish density (fish/m) for Coho fry (CO) and Steelhead fry (RB0) for rampdowns with a ramp rate < 5 cm/h. Rampdowns were completed between August 2018 and September 2019. Stranding densities and mainstem fish densities were based on sampling at six sites selected to represent high-risk stranding habitat.

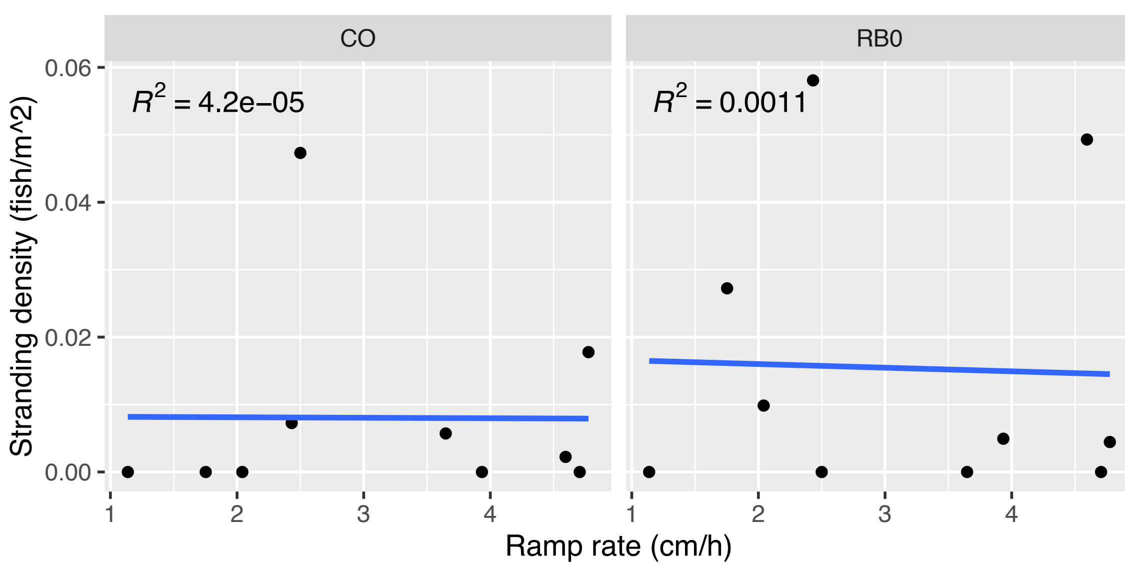


Figure 19. Stranding density (fish stranded/m^2 dewatered bar) in relation to ramp rate (cm/h) for Coho fry (CO) and Steelhead fry (RB0) for rampdowns with a ramp rate < 5 cm/h. Rampdowns were completed between August 2018 and September 2019. Stranding densities and mainstem fish densities were based on sampling at six sites selected to represent high-risk stranding habitat.

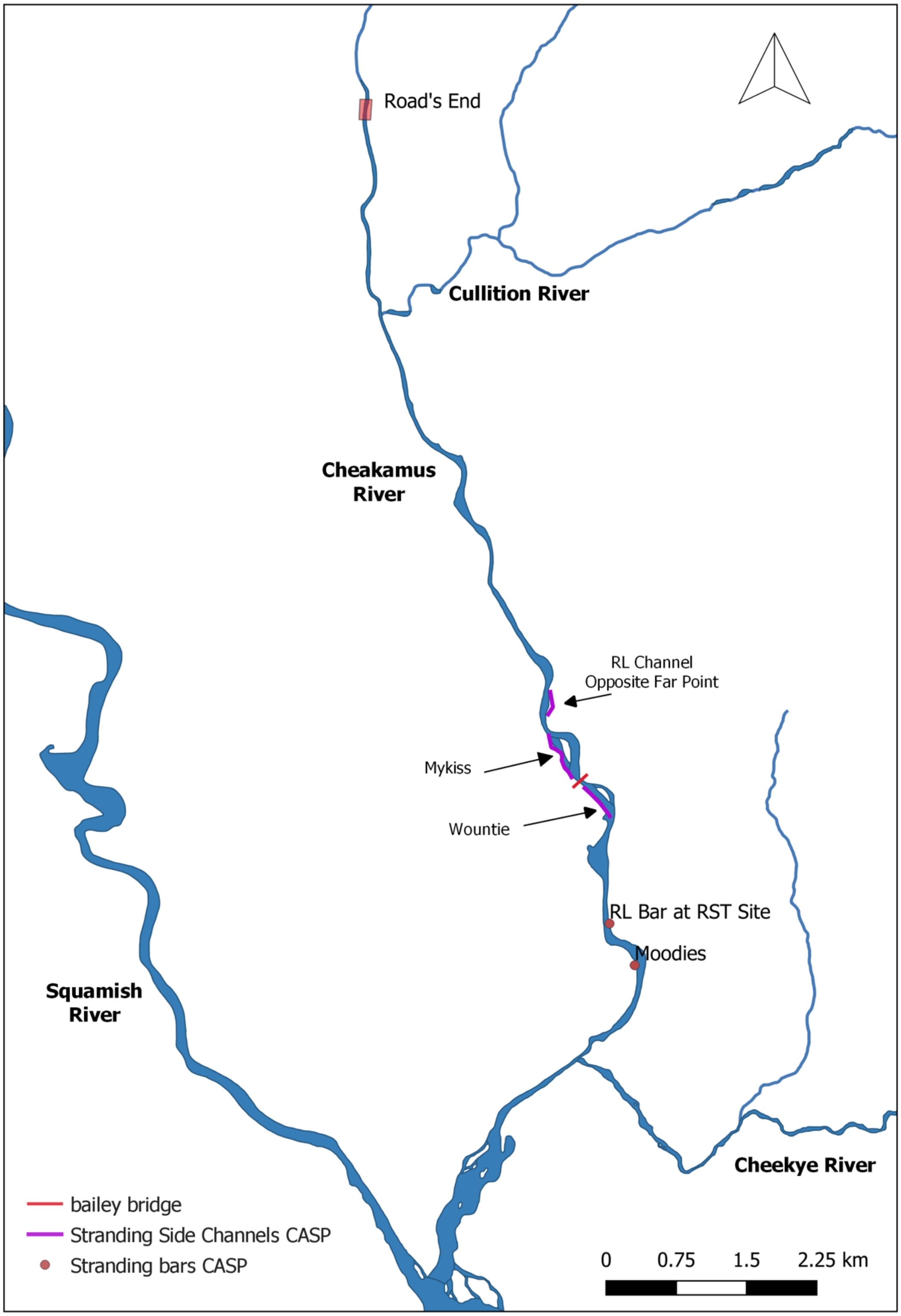


Figure 20. Map of the lower Cheakamus River showing the location of six high risk pool stranding sites that account for a high proportion of juvenile fish strandings in habitats that become isolated from the mainstem following a rampdown: Road’s End, Far Point bar, Mykiss and Wountie side-channels, RST bar and Moodie’s bar.

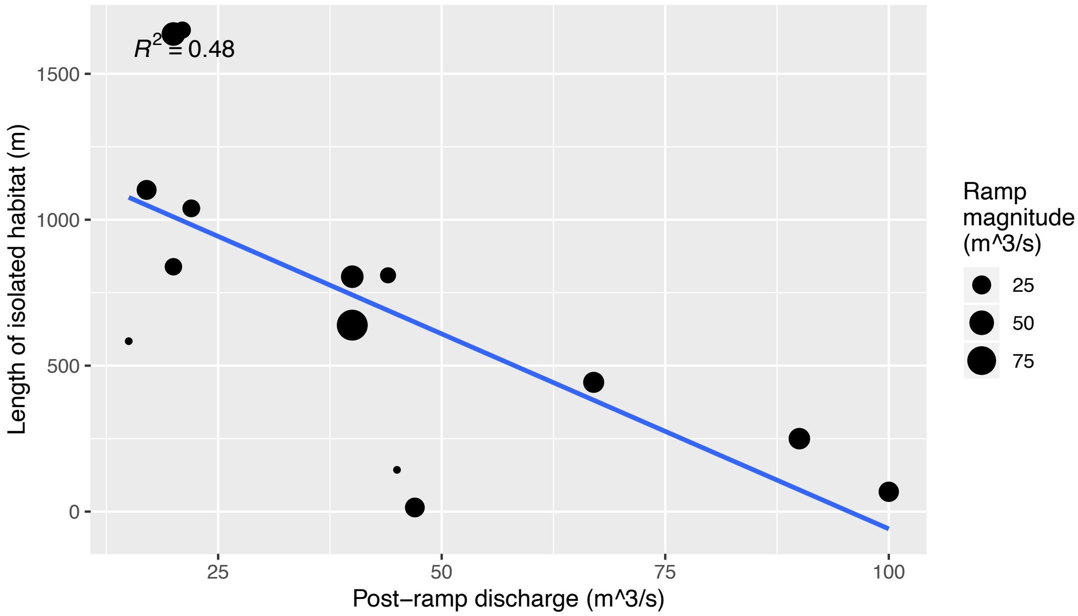


Figure 21. The length of pool and side-channel habitat isolated (m) in relation to post-rampdown discharge (m^3/s) within reaches 2-5 of the Cheakamus River for rampdown from August 2018 to September 2019. Dot size indicates ramp magnitude (m^3/s). The three rampdowns August 9-20, 2019 were combined to reduce the effect of ramp magnitude.

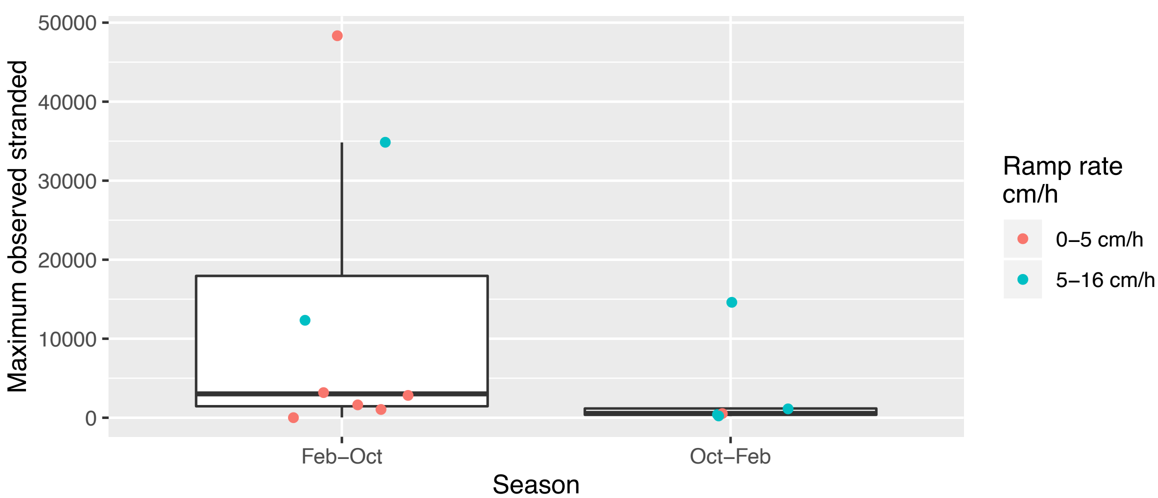


Figure 22. Maximum number of fish observed in isolated pools and side-channels for each reduction event by season and temperature class. Note that for the flow reductions August and September 2019 that consisted of 2-3 rampdowns were combined to better reflect the number stranded per reduction event. Seasons correspond to hypothesised periods of higher and lower stranding risk: Feb-15 to Oct-14 and Oct-15 to Feb-14 (CASP 2018). The horizontal line in the boxplot represents the median value.

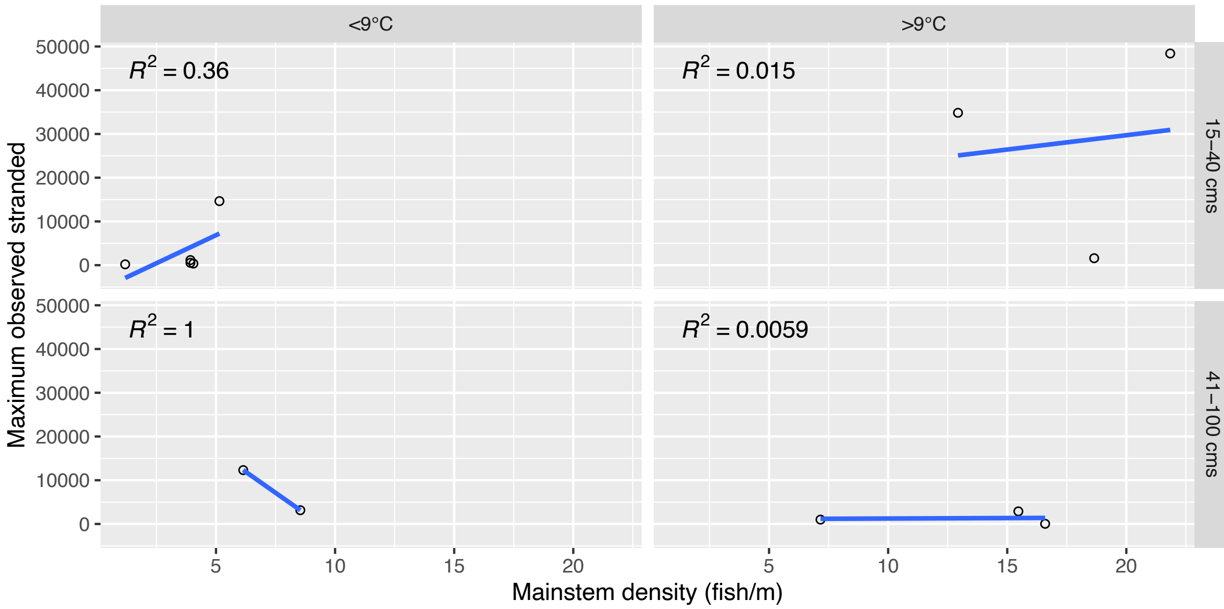


Figure 23. Maximum number of fish observed in isolated pools and side-channels in relation to the mainstem abundance (fish/m) for rampdowns with water temperature above and below 9° C and with a post-ramp discharge of wither 15-40 m^3/s or 41-100 m^3/s. Note that for the flow reductions August and September 2019 that consisted of 2-3 rampdowns were combined to better reflect the number stranded per reduction event. $R^2 = 1$ are the result insufficient sample size of should be interpreted as NA.

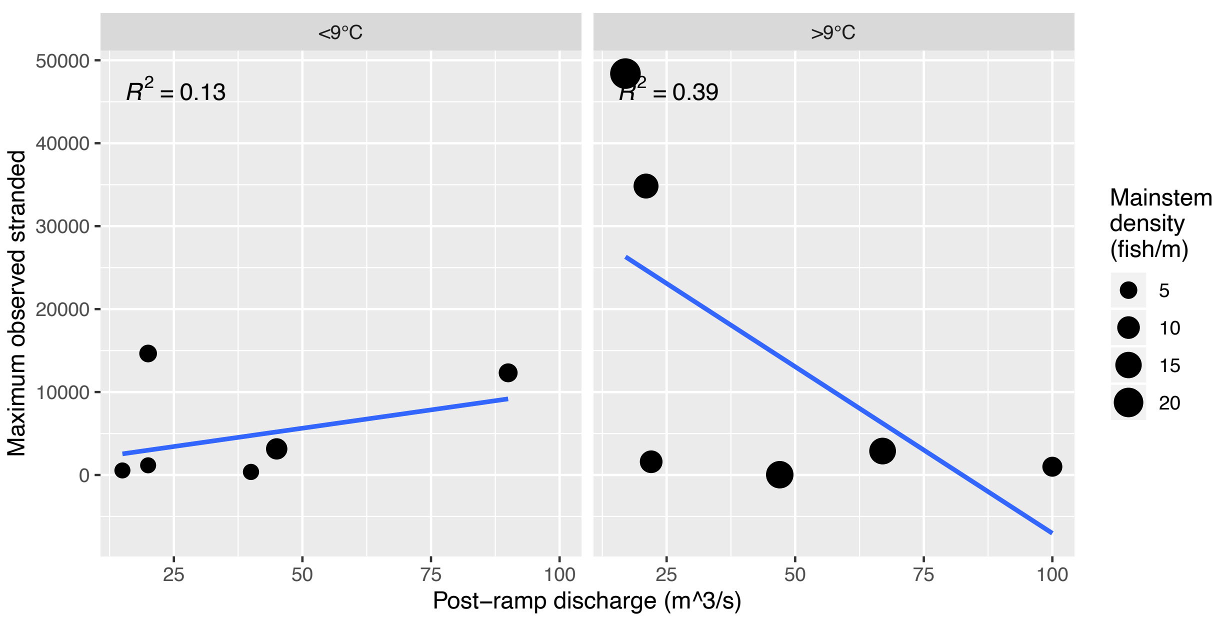


Figure 24. Maximum number of fish observed in isolated pools and side-channels in relation to the post-ramp discharge (m^3/s) for rampdowns with water temperature above and below 9° C. Dot size indicates mainstem density (fish/m). Note that for the flow reductions August and September 2019 that consisted of 2-3 rampdowns were combined to better reflect the number stranded per reduction event.

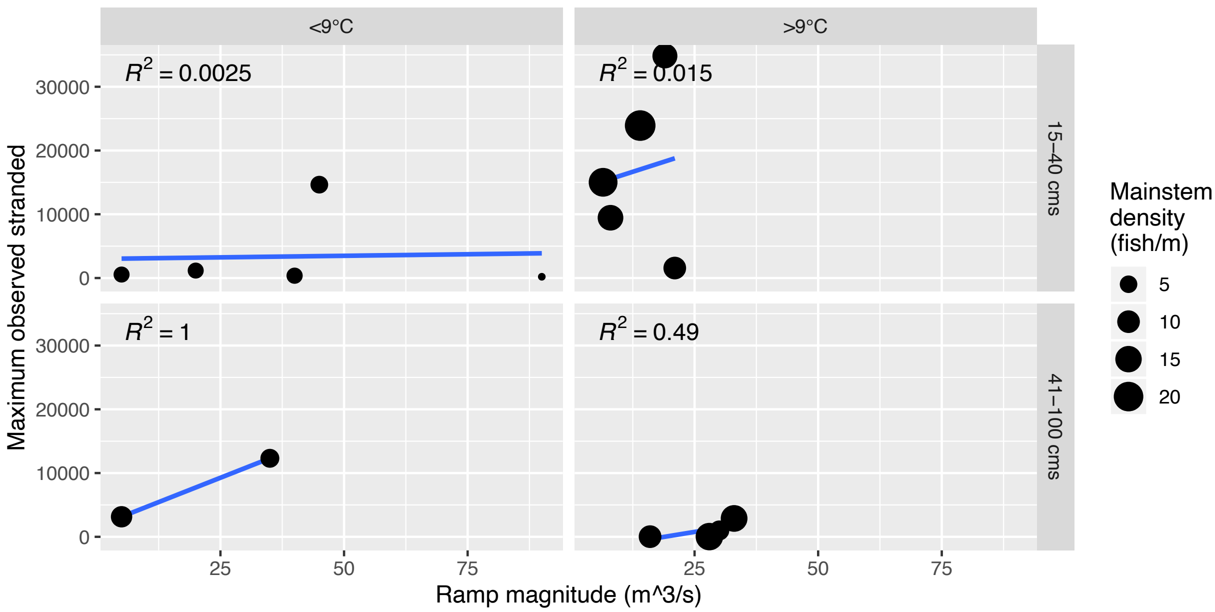


Figure 25. Maximum number of fish observed in isolated pools and side-channels in relation to ramp magnitude (m^3/s) for rampdowns with water temperature above and below 9° C and with a post-ramp discharge of 15-40 m^3/s and 41-100 m^3/s. The three August 2019 and two September 2019 ramps were not combined for this figure. $R^2 = 1$ are the result insufficient sample size of should be interpreted as NA.

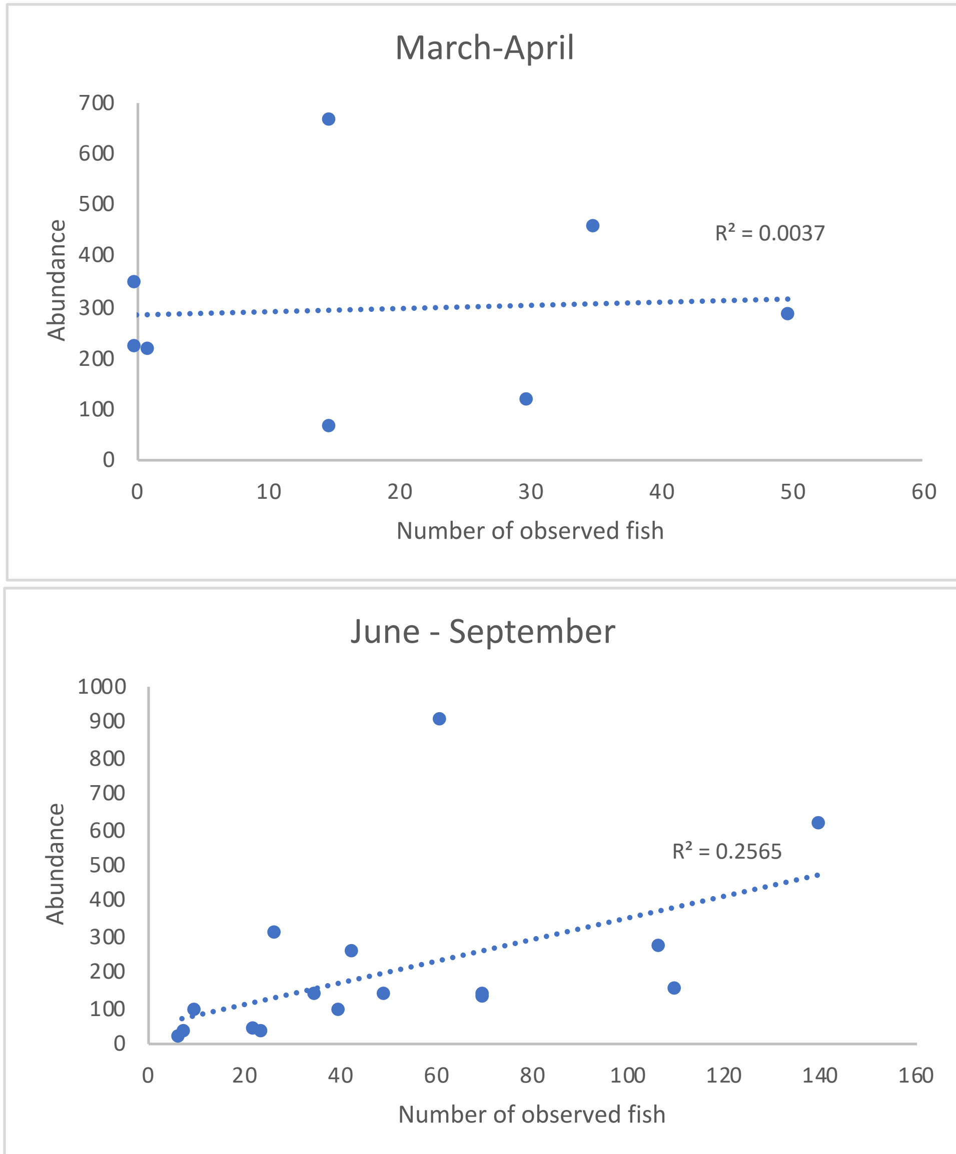


Figure 26. The number of fish in pools seen by observers in relation to mark-recapture estimates of abundance estimated March-April (4.6-7.4 C°) or June-September (13.1-17.6 C°).

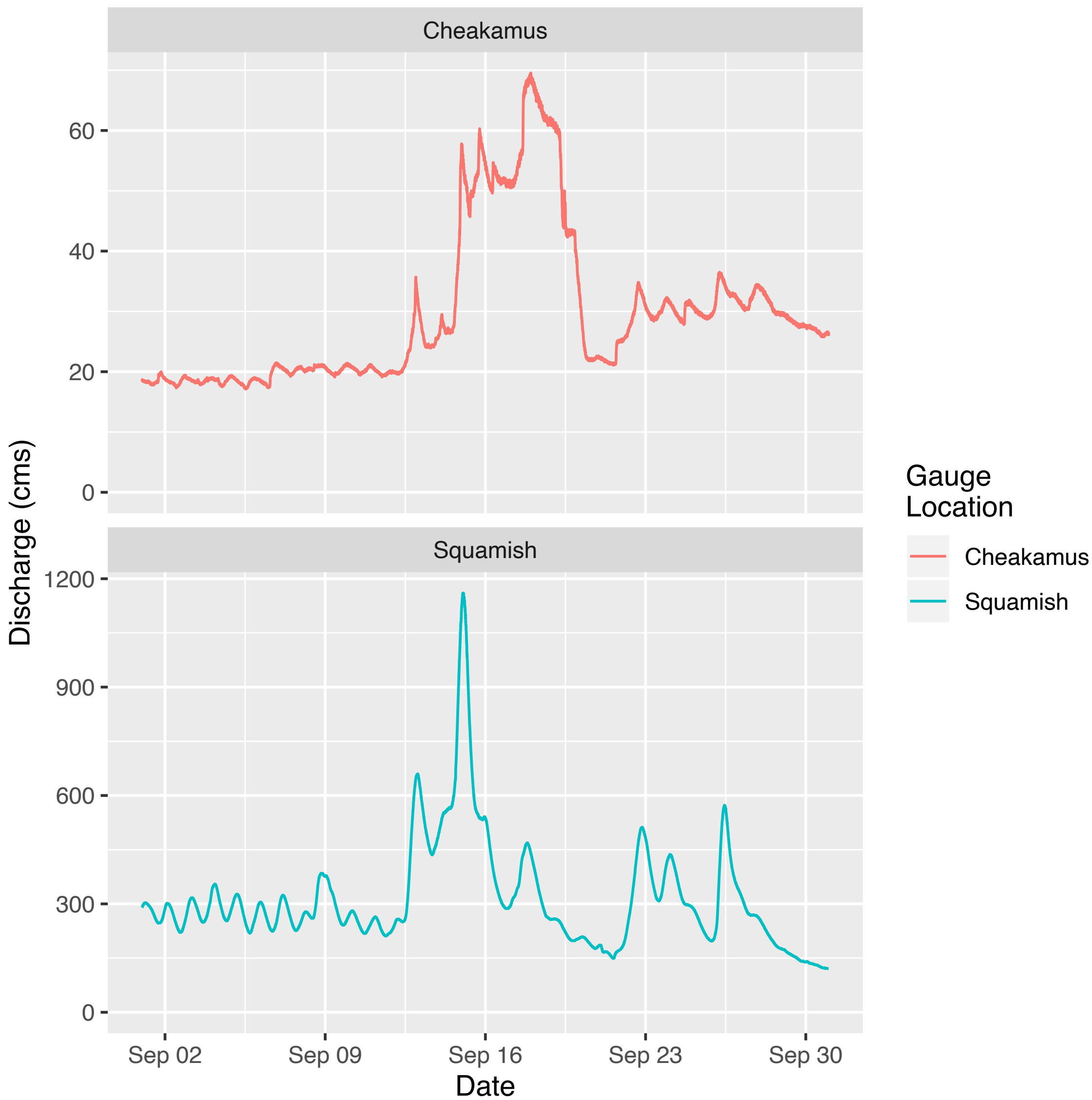
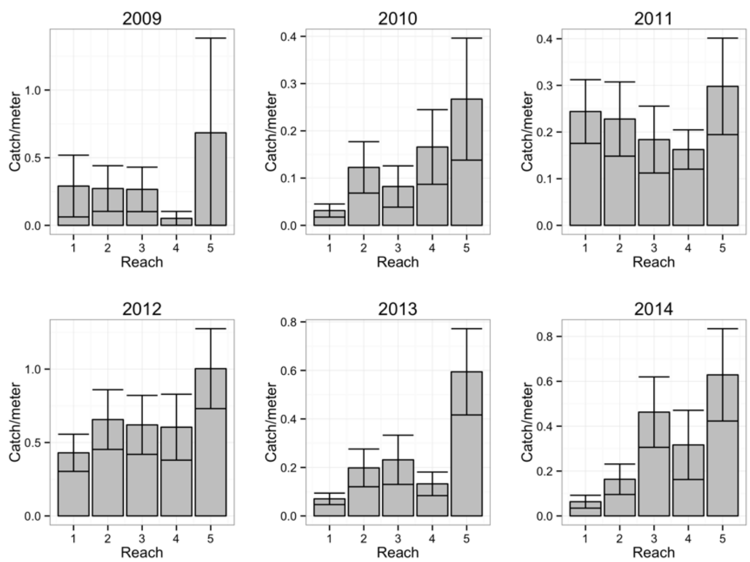


Figure 27. 15 minutes discharge (m^3/s) for the Squamish River (WSC 08GA022) and Cheakamus (WSC 08GA043) rivers from gauge stations near Brackendale for the month of September 2019.

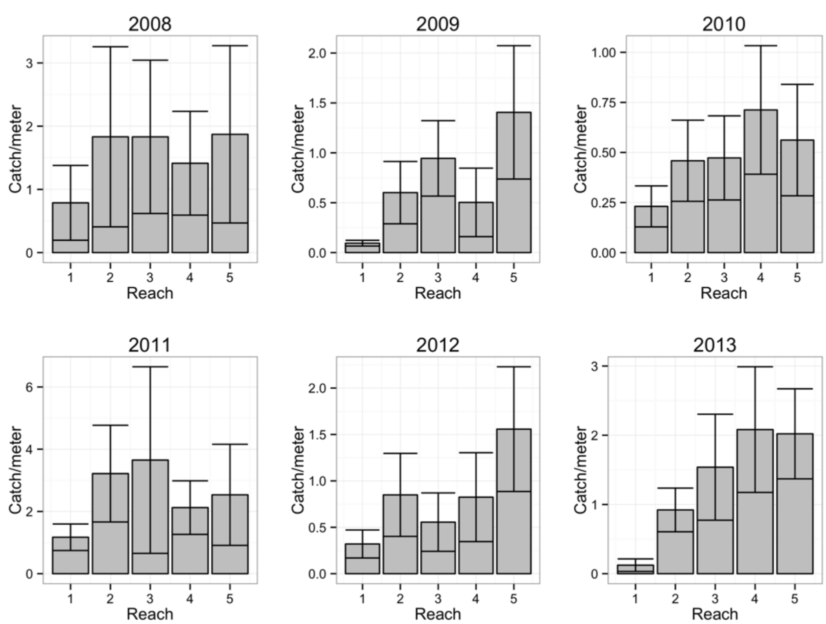
11 Appendices

Appendix 1 a-b. Mean spring and fall catch per meter of shoreline of Steelhead fry by reach for the years 2008-2013. Catch based on one-pass backpack electrofishing for fall estimates and both electrofishing and snorkelling for spring estimates. Reach boundaries are the same as those used for CASP monitoring. Assuming that capture efficiency is constant across reaches and years, catch per meter would be a reliable indicator of relative abundance. Data provided by CMSMON-3.

A. Spring - April



B. Fall - September



Appendix 2. Flow statistics for monitored rampdowns during 2018-2019 at the Brackendale gauge on the Cheakamus River (WSC 08GA043). The start and end time correspond period beginning and end of rapid stage decrease and omits the period at the end of the rampdown between end of the rapid decline and stabilized level.

	Discharge m^2/s		Stage (m)						
Date	Start	End	Start	End	Change	Start time	End time	Duration (h)	Ramp rate (cm/h)
20-Aug-18	40	21	1.15	0.925	0.22	8:50	11:30	2.67	8.3
22-Oct-18	65	20	1.42	0.936	0.49	5:45	10:40	4.92	9.9
06-Nov-18	80	40	1.49	1.159	0.33	7:10	10:40	3.50	9.5
08-Nov-18	40	20	1.10	0.95	0.15	5:45	7:20	1.58	9.5
09-Nov-18	20	15	0.90	0.83	0.07	4:15	10:35	6.33	1.1
29-Nov-18	130	40	1.77	1.14	0.63	3:40	9:00	5.33	11.8
01-May-19	50	45	1.27	1.22	0.05	8:55	10:55	2.00	2.5
17-May-19	125	90	1.82	1.58	0.24	7:30	9:00	1.50	16.0
04-Jun-19	130	100	1.86	1.66	0.20	7:50	12:05	4.25	4.7
20-Jun-19	100	67	1.64	1.407	0.23	4:05	10:20	6.25	3.6
12-Jul-19	75	47	1.44	1.217	0.22	4:40	9:20	4.67	4.8
09-Aug-19	45	31	1.18	1.022	0.15	6:30	9:50	3.33	4.6
15-Aug-19	33	26.5	1.04	0.965	0.07	7:40	10:35	2.92	2.4
20-Aug-19	25	17	0.96	0.864	0.10	5:45	11:10	5.42	1.8
19-Sep-19	60	44	1.34	1.208	0.13	6:05	9:25	3.33	3.9
20-Sep-19	43	22	1.20	0.95	0.25	21:45	9:45	12.00	2.0

Appendix 3. Mean forklength and count of fish stranded at bar monitoring sites.

	Forklength (mm)				Count			
Date	Chinook fry	Coho fry	Steelhead fry	Steelhead parr	Chinook fry	Coho fry	Steelhead fry	Steelhead parr
20-Aug-18	0	0	33	0	0	0	46	0
22-Oct-18	0	41	60	0	0	1	2	0
06-Nov-18	0	55	0	0	0	2	0	0
29-Nov-18	40	0	0	0	1	0	0	0
01-May-19	0	34	0	0	0	4	0	0
17-May-19	44	33	74	95	4	47	1	1
20-Jun-19	0	27	0	0	0	1	0	0
12-Jul-19	0	30	22	0	0	4	1	0
09-Aug-19	34	53	32	0	1	1	22	0
15-Aug-19	0	55	32	0	0	1	8	0
20-Aug-19	0	0	39	0	0	0	6	0
19-Sep-19	0	54	55	0	0	1	1	0
20-Sep-19	0	0	43	0	0	0	2	0

Appendix 4. Captures from single pass open site electrofishing at mainstem sites adjacent to bar survey sites.

Date	Site	Site Length (m)	Chinook		Chum	Coho	Steelhead	
			0+ fry	1+ parr	0+	0+	0+ fry	1+ parr
19-Aug-2018	226	26	0	0	0	4	102	0
19-Aug-2018	248	30	0	0	0	287	573	82
19-Aug-2018	270	30	0	0	0	1	52	0
19-Aug-2018	469	30	0	0	0	4	87	1
19-Aug-2018	478	30	0	0	0	1	217	2
19-Aug-2018	518	30	0	0	0	3	129	4
23-Oct-2018	226	26	0	0	0	0	7	0
23-Oct-2018	248	30	0	6	0	1	34	8
23-Oct-2018	270	30	0	1	0	0	29	3
23-Oct-2018	469	30	0	0	0	1	51	0
23-Oct-2018	478	30	0	0	0	0	75	1
23-Oct-2018	518	30	0	0	0	2	82	0
12-Nov-2018	226	30	0	0	0	0	5	0
12-Nov-2018	248	30	0	5	0	1	46	2
12-Nov-2018	270	30	0	0	0	0	12	1
12-Nov-2018	469	30	0	0	0	0	30	0
12-Nov-2018	478	30	1	0	0	0	38	0
12-Nov-2018	518	30	1	0	0	0	92	0
3-Dec-2018	226	30	4	0	0	0	2	0
3-Dec-2018	248	30	0	2	0	0	2	0
3-Dec-2018	270	30	1	0	0	0	1	1
3-Dec-2018	469	30	4	0	0	0	9	1
3-Dec-2018	478	30	5	0	0	0	11	0
3-Dec-2018	518	30	6	0	0	0	21	0

Appendix 4 continued

			Chinook		Chum	Coho	Steelhead	
Date	Site		0+ fry	1+ parr	0+	0+	0+ fry	1+ parr
2-May-2019	226	30	22	0	2	43	0	0
2-May-2019	248	30	16	0	1	1	7	0
2-May-2019	270	30	39	0	0	9	7	1
2-May-2019	469	30	25	0	0	19	9	0
2-May-2019	478	30	62	0	0	34	8	0
2-May-2019	518	30	38	0	0	2	14	2
21-May-2019	270	30	12	0	0	45	10	2
21-May-2019	345	30	4	0	0	21	1	1
21-May-2019	373	30	5	0	0	14	0	0
21-May-2019	412	30	0	0	0	1	2	0
21-May-2019	462	30	2	0	0	0	3	2
21-May-2019	496	30	20	0	0	13	1	0
4-Jun-2019	270	30	6	0	0	66	0	10
4-Jun-2019	345	30	10	0	0	48	0	5
4-Jun-2019	373	30	1	0	0	4	0	0
4-Jun-2019	406	27	9	0	0	28	0	1
4-Jun-2019	462	30	1	0	0	2	0	4
4-Jun-2019	496	30	5	0	0	32	0	3
21-Jun-2019	270	30	0	0	0	122	0	1
21-Jun-2019	309	45.8	1	0	0	65	0	2
21-Jun-2019	345.1	30	0	0	0	115	0	0
21-Jun-2019	345.2	40	1	0	0	51	0	1
21-Jun-2019	406	30	1	0	0	23	0	1
21-Jun-2019	462	30	0	0	0	8	0	1
21-Jun-2019	496	30	0	0	0	36	0	0

Appendix 4 continued

Date	Site	Site Length	Chinook		Chum	Coho	Steelhead	
		(m)	0+ fry	1+ parr	0+	0+	0+ fry	1+ parr
17-Jul-2019	226	30	2	0	0	30	180	0
17-Jul-2019	248	30	2	0	0	40	83	4
17-Jul-2019	270	30	3	0	0	85	73	5
17-Jul-2019	478	30	0	0	0	11	44	5
17-Jul-2019	518	30	0	0	0	18	89	6
17-Jul-2019	464	30	0	0	0	0	22	3
8-Aug-2019	226	30	0	0	0	12	84	0
8-Aug-2019	248	24	2	0	0	89	214	0
8-Aug-2019	270	30	0	0	0	34	102	0
8-Aug-2019	469	30	0	0	0	0	118	1
8-Aug-2019	478	30	0	0	0	10	308	0
8-Aug-2019	520	30	0	0	0	14	157	1
14-Aug-2019	226	30	0	0	0	12	84	0
14-Aug-2019	248	30	0	0	0	110	110	1
14-Aug-2019	270	30	0	0	0	20	72	0
14-Aug-2019	469	30	0	0	0	11	104	4
14-Aug-2019	478	30	0	0	0	8	264	1
14-Aug-2019	520	30	0	0	0	0	207	2
19-Aug-2019	226	30	1	0	0	6	66	0
19-Aug-2019	248	26	1	0	0	40	101	1
19-Aug-2019	270	30	0	0	0	20	34	0
19-Aug-2019	469	30	0	0	0	4	84	0
19-Aug-2019	478	30	0	0	0	5	114	0
19-Aug-2019	520	30	0	0	0	14	254	3
21-Sep-2019	226	30	0	0	0	0	4	0
21-Sep-2019	248	30	0	0	0	12	106	4
21-Sep-2019	270	30	0	0	0	7	59	3
21-Sep-2019	469	30	0	0	0	6	110	1
21-Sep-2019	518	30	0	0	0	7	171	4
21-Sep-2019	472	30	0	0	0	0	80	0

Appendix 5. Mark recapture results for chinook, Chum, Coho, and Steelhead age-0 fry in mainstem and pool habitat aggregated by month including the number of sites sampled, the total number of fish marked (Marked), the total number of marked fish that were recaptured (Recaptured). Capture efficiency is calculated as the total recaptured divided by total marked for each month.

Mainstem

Species	Month	Number of sites	Marked	Recaptured	Capture efficiency
Chinook fry	February	2	81	15	0.19
Chinook fry	March	2	184	41	0.22
Chinook fry	April	3	214	78	0.36
Chinook fry	June	2	13	4	0.31
Chum fry	March	2	12	1	0.08
Chum fry	April	3	78	13	0.17
Coho fry	April	3	106	14	0.13
Coho fry	June	3	329	41	0.12
Steelhead fry	February	2	181	34	0.19
Steelhead fry	March	2	89	39	0.44
Steelhead fry	April	3	67	31	0.46

Pools

Species	Month	Number of sites	Marked	Recaptured	Capture efficiency
Chinook fry	March	4	129	34	0.26
Chinook fry	April	4	273	74	0.27
Chum fry	March	2	66	46	0.70
Chum fry	April	4	59	15	0.25
Coho fry	March	4	18	6	0.33
Coho fry	April	4	127	7	0.06
Coho fry	June	5	398	162	0.41
Coho fry	August	6	293	76	0.26
Coho fry	September	8	173	96	0.55
Steelhead fry	March	4	174	60	0.34
Steelhead fry	April	4	92	17	0.18
Steelhead fry	August	8	417	89	0.21
Steelhead fry	September	5	53	12	0.23

Appendix 6. Mark-recapture calibration site information and mean capture efficiency for Chinook, Chum, Coho and Steelhead age 0 fry in mainstem and pool habitats. No pool sites were included in the first sampling session due to ice cover.

Habitat	Month	Average Temperature (°C)	Capture efficiency Chinook fry	Chum fry	Coho fry	Steelhead fry
Mainstem	February	2.5	19%			19%
Mainstem	March	4.3	22%	8%		44%
Mainstem	April	6.6	36%	17%	13%	46%
Mainstem	June	11.4	31%		12%	
Pool	March	4.6	26%	70%	33%	34%
Pool	April	7.4	27%	25%	6%	18%
Pool	June	13.1			41%	
Pool	August	17.6			26%	21%
Pool	September	15.8			55%	23%

Appendix 7. Mark recapture results for chinook (CHO), Chum (CM0), Coho (CO0), and Steelhead (RB0) age-0 fry in mainstem and pool habitat aggregated by site including the total number of fish marked (Marked), the total number of marked fish that were recaptured (Recaptured). Capture efficiency (ce) is calculated as the total recaptured divided by total marked for each month. N is the estimated abundance for the site.

Habitat	Species / age	Date	Site	Marked	Recaptured	Captured	ce	N
MAINSTEM	CH0	2019-02-24	248	46	15	58	0.33	172
MAINSTEM	CH0	2019-02-24	469	35	NA	14	NA	NA
MAINSTEM	CH0	2019-03-13	270	36	2	25	0.06	320
MAINSTEM	CH0	2019-03-13	294	148	39	151	0.26	565
MAINSTEM	CH0	2019-04-10	205.1	38	21	74	0.55	132
MAINSTEM	CH0	2019-04-10	205.2	63	19	71	0.30	229
MAINSTEM	CH0	2019-04-10	248.1	113	38	203	0.34	595
MAINSTEM	CH0	2019-06-07	270	11	3	13	0.27	41
MAINSTEM	CH0	2019-06-07	290	2	1	5	0.50	8
MAINSTEM	CH0	2019-06-07	294	NA	NA	5	NA	NA
MAINSTEM	CM0	2019-03-13	270	5	NA	5	NA	NA
MAINSTEM	CM0	2019-03-13	294	7	1	5	0.14	23
MAINSTEM	CM0	2019-04-10	205.1	25	7	77	0.28	253
MAINSTEM	CM0	2019-04-10	205.2	28	1	22	0.04	333
MAINSTEM	CM0	2019-04-10	248.1	25	5	41	0.20	181
MAINSTEM	CO0	2019-03-13	270	2	NA	1	NA	NA
MAINSTEM	CO0	2019-03-13	294	NA	NA	1	NA	NA
MAINSTEM	CO0	2019-04-10	205.1	86	12	104	0.14	702
MAINSTEM	CO0	2019-04-10	205.2	19	2	24	0.11	166
MAINSTEM	CO0	2019-04-10	248.1	1	NA	7	NA	NA
MAINSTEM	CO0	2019-06-07	270	101	18	76	0.18	412
MAINSTEM	CO0	2019-06-07	290	98	7	95	0.07	1187
MAINSTEM	CO0	2019-06-07	294	130	16	144	0.12	1116
MAINSTEM	RB0	2019-02-24	248	93	19	68	0.20	323
MAINSTEM	RB0	2019-02-24	469	88	15	49	0.17	277
MAINSTEM	RB0	2019-03-13	270	54	27	27	0.50	54
MAINSTEM	RB0	2019-03-13	294	35	12	49	0.34	137
MAINSTEM	RB0	2019-04-10	205.1	4	3	11	0.75	14
MAINSTEM	RB0	2019-04-10	205.2	27	10	41	0.37	106
MAINSTEM	RB0	2019-04-10	248.1	36	18	66	0.50	129
MAINSTEM	RB0	2019-06-07	270	NA	NA	4	NA	NA
MAINSTEM	RB0	2019-06-07	290	NA	NA	1	NA	NA
MAINSTEM	RB0	2019-06-07	294	NA	NA	2	NA	NA

Appendix 7 continued

Habitat	Species / age	Date	Site	Marked	Recaptured	Captured	ce	N
POOL	CH0	2019-03-15	485	37	8	12	0.22	54
POOL	CH0	2019-03-15	490	68	24	75	0.35	209
POOL	CH0	2019-03-15	511	23	2	6	0.09	55
POOL	CH0	2019-03-15	520	1	NA	1	NA	NA
POOL	CH0	2019-04-12	205.1	53	20	57	0.38	148
POOL	CH0	2019-04-12	205.2	41	5	12	0.12	90
POOL	CH0	2019-04-12	294.1	59	18	18	0.31	59
POOL	CH0	2019-04-12	294.2	120	31	61	0.26	233
POOL	CM0	2019-03-15	485	3	6	8	2.00	4
POOL	CM0	2019-03-15	511	63	40	42	0.63	66
POOL	CM0	2019-04-12	205.1	13	5	5	0.38	13
POOL	CM0	2019-04-12	205.2	38	7	7	0.18	38
POOL	CM0	2019-04-12	294.1	3	2	2	0.67	3
POOL	CM0	2019-04-12	294.2	5	1	1	0.20	5
POOL	CO0	2019-03-15	485	11	4	8	0.36	21
POOL	CO0	2019-03-15	490	3	NA	1	NA	NA
POOL	CO0	2019-03-15	511	2	NA	NA	NA	NA
POOL	CO0	2019-03-15	520	2	2	6	1.00	6
POOL	CO0	2019-04-12	205.1	53	4	23	0.08	258
POOL	CO0	2019-04-12	205.2	65	2	22	0.03	505
POOL	CO0	2019-04-12	294.1	4	NA	NA	NA	NA
POOL	CO0	2019-04-12	294.2	5	1	2	0.20	8
POOL	CO0	2019-06-11	117	114	34	185	0.30	610
POOL	CO0	2019-06-11	520.1	50	19	35	0.38	91
POOL	CO0	2019-06-11	520.2	70	32	59	0.46	128
POOL	CO0	2019-06-11	520.3	70	29	64	0.41	153
POOL	CO0	2019-06-11	520.4	94	48	70	0.51	137
POOL	CO0	2019-08-22	516.4	29	7	35	0.24	134
POOL	CO0	2019-08-22	516.4	27	9	18	0.33	52
POOL	CO0	2019-08-22	516.5	36	7	57	0.19	267
POOL	CO0	2019-08-22	516.5	22	9	21	0.41	50
POOL	CO0	2019-08-22	516.5	NA	NA	3	NA	NA
POOL	CO0	2019-08-22	516.5	NA	NA	NA	NA	NA
POOL	CO0	2019-08-23	480	143	34	72	0.24	299
POOL	CO0	2019-08-23	490	36	10	40	0.28	137
POOL	CO0	2019-09-11	490	6	NA	13	NA	NA
POOL	CO0	2019-09-12	516.1	3	1	6	0.33	13
POOL	CO0	2019-09-12	516.3	9	7	22	0.78	28
POOL	CO0	2019-09-12	516.4	17	16	38	0.94	40
POOL	CO0	2019-09-12	516.4	14	14	28	1.00	28
POOL	CO0	2019-09-12	516.5	56	30	86	0.54	159
POOL	CO0	2019-09-12	516.5	18	9	17	0.50	33
POOL	CO0	2019-09-12	480	50	19	52	0.38	134

Appendix 7 continued

Habitat	Species / age	Date	Site	Marked	Recaptured	Captured	ce	N
POOL	RB0	2019-03-15	485	17	10	21	0.59	35
POOL	RB0	2019-03-15	490	37	10	37	0.27	130
POOL	RB0	2019-03-15	511	49	12	25	0.24	99
POOL	RB0	2019-03-15	520	71	28	71	0.39	178
POOL	RB0	2019-04-12	205.1	24	4	6	0.17	34
POOL	RB0	2019-04-12	205.2	28	5	5	0.18	28
POOL	RB0	2019-04-12	294.1	10	NA	NA	NA	NA
POOL	RB0	2019-04-12	294.2	30	8	9	0.27	33
POOL	RB0	2019-06-11	520.1	NA	NA	NA	NA	NA
POOL	RB0	2019-08-22	516.4	25	6	31	0.24	118
POOL	RB0	2019-08-22	516.4	14	5	31	0.36	79
POOL	RB0	2019-08-22	516.5	203	32	283	0.16	1755
POOL	RB0	2019-08-22	516.5	9	6	24	0.67	35
POOL	RB0	2019-08-22	516.5	14	3	9	0.21	37
POOL	RB0	2019-08-22	516.5	4	2	8	0.50	14
POOL	RB0	2019-08-23	480	2	1	1	0.50	2
POOL	RB0	2019-08-23	490	146	34	180	0.23	759
POOL	RB0	2019-09-11	490	15	2	50	0.13	271
POOL	RB0	2019-09-12	516.1	11	3	19	0.27	59
POOL	RB0	2019-09-12	516.3	4	NA	1	NA	NA
POOL	RB0	2019-09-12	516.5	22	6	27	0.27	91
POOL	RB0	2019-09-12	480	1	1	1	1.00	1

Appendix 8. Survival rate for Coho and Steelhead age-0 fry in pools over a 21-day period starting August 22-23 and ending September 11-12) by pool depth. Abundance estimates based on Peterson mark recapture estimates.

			Abundance		
Species/age	Site	Max depth (cm)	23-Aug-24	11-Sep-12	Survival
Coho fry	516.48	0-25	0	0	-
Coho fry	516.49	0-25	0	0	-
Coho fry	516.4	26-50	134	40	30%
Coho fry	516.43	26-50	52	28	54%
Coho fry	516.46	26-50	267	159	59%
Coho fry	516.47	26-50	50	33	67%
Coho fry	480	>50	299	134	45%
Coho fry	490	>50	137	0	0%
Steelhead fry	516.48	0-25	37	0	0%
Steelhead fry	516.49	0-25	14	0	0%
Steelhead fry	516.4	26-50	118	0	0%
Steelhead fry	516.43	26-50	79	0	0%
Steelhead fry	516.46	26-50	1755	91	5%
Steelhead fry	516.47	26-50	35	0	0%
Steelhead fry	480	>50	2	1	50%
Steelhead fry	490	>50	759	271	36%

Appendix 9. Site information for pool observer efficiency trials including the sum of mark-recapture abundance estimates for each species (N), the average number of fish observed in each pool and observer efficiency.

Date	Site	Area (m^2)	Max depth (cm)	Temp.	N	Observed fish	Observer efficiency
15-Mar-19	485	26	32	5.3	120	30	0.25
15-Mar-19	490	192	35	5.2	356	0	0.00
15-Mar-19	511	90	30	3.5	187	0	0.00
15-Mar-19	520	66	35	4.4	220	1	0.00
12-Apr-19	205.1	81	60	7.4	401	35	0.09
12-Apr-19	205.2	51	55	7.4	414	15	0.04
12-Apr-19	294.1	9	20	7.4	76	15	0.20
12-Apr-19	294.2	54	45	7.4	283	50	0.18
11-Jun-19	117	91	20	16.7	610	140	0.23
11-Jun-19	520.1	3	20	12.3	91	40	0.44
11-Jun-19	520.2	4	20	11.9	128	70	0.55
11-Jun-19	520.3	19	30	12.3	153	110	0.72
11-Jun-19	520.4	9	25	12.3	137	70	0.51
22-Aug-19	516.4	9	36	17.7	248	43	0.17
22-Aug-19	516.43	13	32	17.7	142	50	0.35
22-Aug-19	516.46	72	35	18.5	2078	435	0.21
22-Aug-19	516.47	4	42	17.2	96	10	0.10
22-Aug-19	516.48	3	8	17.7	48	22	0.47
22-Aug-19	516.49	0	10	17.2	14	7	0.50
23-Aug-19	480	135	80	15.8	305	27	0.09
23-Aug-19	490	221	67	15.8	892	61	0.07
12-Sep-19	516.43	8	10	13.2	28	24	0.86
12-Sep-19	516.47	2	40	13.2	39	8	0.21
11-Sep-19	480	364	80	12.3	159	35	0.22
11-Sep-19	490	188	35	12.3	468	107	0.23

Appendix 10. Cheakamus River salmonid life stage chart prepared from Fell and Melville (2016).

Table 3. Cheakamus River Salmonid Lifestages Chart

Prepared for BC Hydro By InStream Fisheries Research Inc.

Revised April, 2012

Legend
X = peak occurance
x = occurring

Species	Lifestage	January				February				March				April				May				June				July				August				September				October				November				December			
	Adult Upstream Migration																									x	x	x	x	x	x	x	X	X	x	x	x	X	X	x	x								
	Spawning																															x	x	X	X	x	x	x	X	X	x	x							
Oncorhynchus tshawytscha	Incubation	X	X	X	X	X	X	X	X	X	X	X	x	x	x	x	x	x														x	x	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
	Fry Emergence				x	x	x	X	X	X	X	X	X	X	X	X	X	x	x																														
Chinook Salmon	Fry Downstream Migration					x	x	x	X	X	X	X	X	X	X	X	X	x	x	x																													
	Juvenile Rearing												x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x			
(summer & fall runs)	Juvenile Over-wintering	x	x	x	x	x	x	x	x	x	x	x	x	x																															x	x	x	x	x
	Smolt Downstream Migration								x	x	x	x	x	x	x	X	X	X	X	X	X	x	x	x	x																								
Oncorhynchus gorbuscha	Adult Upstream Migration																													x	x	X	X	X	X	x	x												
	Spawning																																x	X	X	X	X	x	x										
Pink Salmon	Incubation	X	X	X	X	X	X	X	X	X	X	x	x																				x	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
(odd years only)	Fry Emergence						x	x	x	X	X	X	X	X	x	x	x																																
	Fry Downstream Migration							x	x	x	X	X	X	X	X	X	x	x	x																														
	Adult Upstream Migration																																					x	x	x	X	X	X	x	x				
Oncorhynchus keta	Spawning																																							x	x	X	X	X	X	x	x		
	Incubation	X	X	X	X	X	X	X	X	X	X	X	X	x	x	x	x	x	x																					x	x	X	X	X	X	X	X	X	X
Chum Salmon	Fry Emergence							x	x	x	x	X	X	X	X	X	x	x	x	x																													
	Fry Downstream Migration							x	x	x	x	X	X	X	X	X	x	x	x	x																													
	Adult Upstream Migration	X	x	x	x																																	x	x	x	x	x	x	x	X	X	X	X	X
	Spawning	X	X	x	x	x	x																																					x	x	X	X	X	X
Oncorhynchus kisutch	Incubation	X	X	X	X	X	X	X	X	X	X	X	X	X	X	x	x	x	x	x	x	x	x																					x	x	X	X	X	X
	Fry Emergence					x	x	x	x	x	x	X	X	X	X	X	X	X	X	X	x	x	x	x	x																								
Coho Salmon	Juvenile Rearing												x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x			
	Juvenile Over-wintering	x	x	x	x	x	x	x	x	x	x	x	x	x																															x	x	x	x	x
	Smolt Downstream Migration													x	x	x	X	X	X	X	X	X	x	x	x																								
	Adults Present	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Oncorhynchus mykiss	Spawning									x	x	x	x	x	x	X	X	X	X	x	x	x	x	x	x																								
	Incubation									x	x	x	x	x	x	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	x	x	x																
Resident	Fry Emergence																					x	x	x	X	X	X	X	X	X	x	x	x																
Rainbow Trout	Juvenile Rearing												x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x			
	Juvenile Over-wintering	x	x	x	x	x	x	x	x	x	x	x	x	x																															x	x	x	x	x
Oncorhynchus mykiss	Adult Upstream Migration					x	x	x	x	X	X	X	X	X	X	X	X	X	X	x	x	x	x																										
	Spawning									x	x	x	x	x	x	X	X	X	X	x	x	x	x	x	x																								
Ocean-Run	Incubation									x	x	x	x	x	x	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	x	x	x																
Steelhead	Fry Emergence																					x	x	x	X	X	X	X	X	X	x	x	x																
	Juvenile Rearing												x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x			
(winter run)	Juvenile Over-wintering	x	x	x	x	x	x	x	x	x	x	x	x	x																															x	x	x	x	x
	Smolt Downstream Migration														x	x	x	X	X	X	X	x	x	x	x																								
	Adults Present	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Adult Downstream Migration																	x	x	X	X	X	X	X	X	X	X	x	x																				
	Adult Upstream Migration																																					x	x	X	X	X	X	x	x				
Salvelinus confluentus	Spawning																																	x	x	X	X	X	X	x	x								
	Incubation	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	x	x	x														x	x	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Bull Trout	Emergence													x	x	X	X	X	X	x	x																												
	Rearing													x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x							