Bridge River Project Water Use Plan

Lower Bridge River Aquatic Monitoring

Implementation Year 10

Reference: BRGMON-1

BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 10 (2021) Results

Study Period: April 1 2021 to March 31 2022

Jeff Sneep
Chris Perrin, Limnotek
Josh Korman, Ecometric Research

Field Studies and Data Collection Completed by:
Vroni Lambinet, Mistiy Diablo, Elijah Michel, Brett Squirrell and Carley Wall,
Coldstream Nature-Based Solutions

April 30, 2022

BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 10 (2021) Results

Report Prepared for:
St'at'imc Eco-Resources

Report Prepared by:
Jeff Sneep
Chris Perrin, Limnotek, and
Josh Korman, Ecometric Research

Field Studies and Data Collection Completed by:
Vroni Lambinet, Mistiy Diablo, Elijah Michel, Brett Squirrell & Carley Wall
Coldstream Nature-Based Solutions

Ref. no. BRGMON-1

April 2022

Executive Summary

Monitoring of juvenile salmonid production, periphyton and benthic communities and their habitat continued in 2021 as part of a long-term process to reduce uncertainty about the ecological response to different flow releases from the Terzaghi Dam. Sampling and measurements in 2021 were the same as in previous years dating to 1996 in Reaches 4, 3, and 2. Twelve fish sampling (juvenile standing stock) sites and four periphyton and benthic sampling sites were added in Reach 1 in 2019 and sampled again in 2020 and 2021. Two reference sites that were established in the Yalakom River in 2018 have also been sampled each year since.

Flow releases to the Lower Bridge River during the post-high flow years (i.e., 2019 to 2021) were most similar to the Trial 2 hydrograph, but with higher peak flows in 2021 (i.e., ~25 $m^3{\cdot}s^{-1}$ instead of the Trial 2 peak of 15 $m^3{\cdot}s^{-1}$). There were slight variations in the timing of ramp up to the peak and ramp down to the summer rearing flows among years, but these slight differences in flow change timing were not anticipated to have a measurable effect (within detection limits) on the results. As such, the flow treatments from Terzaghi Dam to the Lower Bridge River continue to be largely characterized by the differences in the magnitude of the peak flow during the spring and early summer, with generally comparable flows at other times of year among treatments. Given earlier findings that peak flow magnitudes influenced patterns of benthic assemblages and juvenile salmonid production, we focused much of the presentation of results and discussion in this report around comparing the Post-high flow years (2019-2021) with the results from the High Flow years (2016-2018) and Trial 2 (2011-2015).

Measurements of fish, invertebrate, and periphyton metrics among years were used in statistical tests of flow effects on aquatic biological communities. Periphyton was measured as peak biomass accruing on artificial substrata in the fall (called PB) and the community was described using cell counts by lowest reliable taxon. Invertebrate metrics were density of all taxa, density of chironomids, the sum of mayfly, stonefly, and caddisfly densities (called EPT) that are typically considered a group most sensitive to environmental change, and family richness (the count of families) measured during the fall. The set of metrics evaluated for juvenile salmonids (i.e., mykiss fry, mykiss parr, coho fry and Chinook fry) were size (mean weight) and condition factor, abundance and biomass, and stock-recruitment. We also specifically tested for flow effects on juvenile salmonid densities and biomass using a mixed effects model, which affirmed that flow and reach are significant determinants. We incorporated a set of covariates into the model including nutrient (phosphorus) concentrations, benthic invertebrate densities (as a proxy for fish food availability), mean temperatures during the summer growing season, incubation period (for Chinook), and spawner densities to determine whether changes and variations in these parameters explain some of the differences among years and flow treatments in addition to flow effects.

Year 10 analysis and reporting also included an array of modified operations (high flow) components since flow releases above the WUP (Trial 2) peak were implemented in 2021. These

activities included: spot water quality monitoring, kokanee entrainment surveys, bank erosion/substrate recruitment assessments, mitigation effectiveness monitoring, and high flow ramp down monitoring and stranding risk assessment.

Water temperatures in 2019-2021 (called the Post-high flow years) were largely within the range documented for the other flow treatments, particularly during fall, winter and spring. Average summer temperatures in the Post-high flow years were higher than during Trial 2 and lower than the High Flow period. The warmer summer temperatures were within the preferred range for the target salmonid species and likely contributed to conducive rearing conditions in summer.

Fall and winter temperatures resulted in predicted emergence timing for chinook and coho that was on par with the estimated timing for those species under Trial 2. As documented in previous reports, the majority (68%) of chinook spawning has occurred in the lower portion of Reach 3 with an associated median emergence timing of mid-March to early April and the majority (77%) of coho spawning was in Reach 4 with an associated median emergence timing of mid April. The spawning distribution among monitoring locations within reaches 3 and 4 did not substantively change with inclusion of the 2021 redd location data. The number of redds observed for chinook and coho in 2021 were four and nine, respectively.

Variation in dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) concentrations across monitoring years produced shifts in potential limitation of benthic algal growth based on molar N:P ratios at different times and places. During Trial 0 and Trial 1, the molar N:P ratio showed co-limitation by N and P. Molar N:P increased in Reach 2 during and after Trial 2, which showed a progressive time course increase in potential phosphorus deficiency. The Reach 2 N:P during the High Flow years was the highest measured since monitoring began. It showed greatest potential phosphorus deficiency in Reach 2 on record in the Lower Bridge River. With the presence of pink salmon in 2019 and 2021, the Post-high flow molar N:P declined sharply, showing mostly N limitation caused by more P enrichment than N enrichment. Highest P deficiency was in the Yalakom River (based on the years available for that tributary).

The periphyton community was mostly comprised of diatoms that are ubiquitous among mountain rivers. Nutrient enrichment from spawning Pink salmon produced greater PB in Reach 2 than in other reaches. There was no difference in periphyton PB between High Flow years and the other flow treatments regardless of presence or absence of Pink salmon. Over time, however, PB increased, resulting in biomass in the Post-High Flow years being about double what it was during Trial 1.

Benthic invertebrates in the Lower Bridge River were also common to mountain rivers, occurring at densities similar to or greater than those elsewhere. The assemblage mainly included chironomids, other dipterans, and the EPT. Densities increased with distance from Terzaghi Dam, which is consistent with patterns in headwater streams where colonization progresses downstream in association with tributary inflows.

Density of the whole invertebrate assemblage declined by 69% during the High Flow Years. The losses were linked to physical scour during high peak flow in summer, originating as releases from the Terzaghi Dam and lack of recovery in the year of the event. Recovery of the invertebrate community was found a year after the last High Flow Year with densities and composition in the fall of the Post-High-Flow years matching those before the High Flow Years.

Presence of Pink salmon spawners and decomposing carcasses (odd years) supported higher chironomid densities than were found in the absence of Pink spawning (even years). Using concentrations of ammonium and phosphate that are primary decomposition products as proxies for Pink run size, only half of the Pink-on years had sufficient numbers of spawners to cause change in measured nutrient concentrations. To produce an overall chironomid response, the Pink effect had to be large in those years to offset lack of a response when the Pink run was small or negligible. The chironomid response was attributed to change in production from the nutrient loading.

This chironomid response to presence of Pink salmon spawning triggered a second analysis to test nutrient loading effects on benthic invertebrates. A Before-After-Control-Impact (BACI) layout was applied to test whether the mean difference of total benthos density between a “Control” and “Treatment” reach in a group of “Before” years was different from the mean difference in the “After” years. The control reach was Reach 4 where few or no Pink salmon spawners were present at any time based on low NH_4-N and SRP concentrations. The treatment reach was Reach 2 where high concentrations of NH_4-N and SRP from Pink spawners in odd years were found. A “Before” year was an even year preceding an odd year when anomalous nutrient concentrations showed presence of Pink spawners in the treatment reach. The “After” years were those having anomalous and high NH_4-N and SRP concentrations showing the presence of Pink spawners in the treatment reach. The BACI selected for widely contrasting reaches and times of measurement and thus was more sensitive than was the rather broad linear ANOVA model covering all places and times.

Results showed that benthos density doubled with a combined 24-times increase in ammonium concentration, a doubling of nitrate concentration, and a 10x increase in phosphate concentration. This response is consistent with observations from several river fertilization experiments reported in the scientific literature. It shows that nutrient addition can be a useful mitigation tool to aid in recovery of fish food organisms that are lost during extremely high-water releases from Terzaghi Dam.

Mean fish condition factor (Fulton’s K value) has decreased over the course of the study period for each target species and age class. Highest values were in the Pre-flow period and Trial 1 and lowest values were in the High flow (2016-2018) or Post-high flow years (2019-2021). The mean value (for all species combined) in Trial 2 was 1.14 (±0.05 SD), which dropped to 1.11 (±0.01 SD) during the High flow years and has remained at 1.11 (±0.01 SD) in the Post-high flow years to-date. Interestingly, for reference, the condition factor assessed for Age-0+ and Age-1 rainbow trout in Downton Reservoir (collected under BRGMON-7) was 1.29 and 1.26, respectively, which is on par with the highest condition factor values documented in the Lower Bridge River (i.e., during the Pre-flow period). To be clear, however, the reduced condition factor values in the LBR do not suggest that the condition of the fish is currently poor (i.e., when K values are <1) but may indicate that fish are needing to spend more time and energy foraging to meet their growth needs relative to earlier study years which can have survival implications.

Juvenile fish abundance was highest at the lowest flow release (i.e., Trial 1) and declined with increasing peak flows from Terzaghi Dam. Abundance and biomass markedly declined for all target species and age classes during the High flow years, except for chinook, which were already at reduced abundance since early in the flow trials. Results from the Post-high flow years (2019-2021) indicate that despite the resumption of lower flow releases (i.e., at or near Trial 2 flow magnitudes and hydrograph shape), juvenile fish abundance and biomass have not recovered. Mean abundance in the Post-high flow years was approx. 15,000 more juvenile salmonids than during the high flow years but was still approx. 200,000 fewer fish than the Trial 2 average. The modest increase in abundance was entirely attributable to mykiss and coho fry. Mean biomass in the post-high flow years was the lowest for any flow treatment to-date; approx. 188 kg lower than the High Flow period and 767 kg lower than Trial 2. These results suggest that juvenile salmonid abundance and biomass in the Lower Bridge River may not be recovered quickly by the resumption of lower flow releases alone. While flow is clearly an important factor, recovery may require additional actions such as addressing changes to rearing habitats caused by the high flows and/or the trend of declining nutrients from the flow release.

Stock-recruitment curves for the High Flow years suggested poorest recruitment of coho fry per spawner stock size for any of the flow treatments assessed, and equivalently low production of chinook fry (as the other trial flows). Stock sizes during the Post-high flow (2019-2021) recruitment years were again at the low end of the range documented across the years of monitoring but were still on par with stock size estimates that produced more juveniles during previous flow treatments. In other words, based on the existing S-R curves, it does not appear that juvenile recruitment during the High and Post-high flow years was necessarily limited by stock sizes. However, only a very small number of S-R datapoints (*n*= 3 in each case) were available for these flow treatments so uncertainty remains about the steepness of the initial slopes and asymptotes for these curves.

Salmonid abundance data were collected in Reach 1 during each of the Post-high flow years. Results for this period highlighted that all the target species and age classes were present in the reach, of which mykiss fry were the most abundant (mean = ~22,000 fish), followed by coho fry (mean = ~11,000 fish), Chinook fry (mean = 8,000 fish), and then mykiss parr (mean = ~4,000 fish). This pattern of relative abundance among the species in Reach 1 was quite consistent each year and the absolute abundances for each species and age class (except coho fry) increased in Reach 1 each year. The Reach 1 total for all species and age classes (mean = ~46,000) during the Post-high flow years was just over 1/3 of all fish captured in the study area (mean = ~128,000).

Lineal densities in Reach 1 were on par with Reach 4 for mykiss fry, reaches 4 and 2 for mykiss parr, Reach 2 for coho fry, and reaches 3 and 2 for Chinook fry, at this stage of post-high flow recovery.

Modified operations monitoring during the higher peak flow period that extended from May to July suggested the following: 1) higher release flows from Terzaghi Dam can increase total dissolved gas saturation and turbidity levels, and make water temperatures, conductivity and alkalinity in the LBR more uniform downstream of the dam (i.e., lessens the influence of tributary inflows to some extent); 2) seasonal enlargement of the wetted area of the channel can cause erosion at some locations in the LBR, such as the toe of alluvial slides adjacent to the channel which contributes to the recruitment of new substrate materials into reaches 2, 3 and 4; 3) high flows flood additional edge areas, including habitats that become isolated from the mainstem or dewater when flows are reduced, thereby adding to the total numbers of fish stranded across the lower flow ranges; however, the rate of stranding (# of fish per 1 $m^3 \cdot s^{-1}$ flow change) appears to be lower at flows above ~13 $m^3 \cdot s^{-1}$ than below; and 4) entrainment of kokanee from Carpenter Reservoir into the Lower Bridge River channel can occur during high flow releases but did not occur in every high flow year (i.e., *n*=83 observed in 2016, *n*=48 observed in 2017, and *n*=0 observed in 2018 and 2021).

Summary of BRGMON-1 Management Questions and Year 10 – 2021 Status

Primary Objectives	**Management Questions**	**Results up to and including Year 10 (2021)**
Core Components: To reduce uncertainty about the relationship between the magnitude of flow release from the dam and the relative productivity of the Lower Bridge River aquatic and riparian ecosystem. To provide comprehensive documentation of the response of key physical and biological indicators to alternative flow regimes to better inform decision on the long-term flow regime for the Lower Bridge River. The scope of this program is limited to monitoring the changes in key physical, chemical, and biological productivity indicators in reaches 2, 3, and 4 of the Lower Bridge River aquatic ecosystem.	How does the instream flow regime alter the physical conditions in aquatic and riparian habitats of the Lower Bridge River ecosystem?	• The biggest gains in wetted area were achieved by the wetting of Reach 4 and the augmentation of flows in Reach 3 by the Trial 1 and 2 treatments. The Post-high flow years (2019-2021) were characterized by a return to lower (i.e., similar to Trial 2) flows. Increases in wetted area from high flows (>15 $m^3 \cdot s^{-1}$) in 2016-2018 were proportionally less substantial and the additional discharge reduced the suitability of mid-channel habitats by increasing flow velocities above suitable thresholds for fish and benthic invertebrates. • Site-specific discharge estimates highlighted that flow conditions in reaches 3 and 4 were fairly similar, but that they differed from sites in reaches 2 and 1 due to the contribution of the Yalakom inflows at the top of Reach 2. During years with lower flow releases from Terzaghi Dam (including 2021), the Yalakom River inflow contributes a higher proportion of the total discharge in the lower reaches which dilutes or masks some of the physical and water chemistry characteristics of the release. The primary difference in the hydrographs among the various flow treatments were the magnitude of the release flows during the peak period (i.e., May to July). • Variation in dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) concentrations across monitoring years produced shifts in potential limitation of benthic algal growth based on molar N:P ratios at different times and places. The Reach 2 N:P during the High Flow years showed greatest potential phosphorus deficiency in Reach 2 on record in the Lower Bridge River. With the presence of pink salmon in 2019 and 2021, the Post-high flow molar N:P declined sharply, showing mostly N limitation caused by more P enrichment than N enrichment. Highest P deficiency was in the Yalakom River. • Water temperatures under all trial flows were cooler in the spring and warmer in fall relative to the Pre-flow (Trial 0) profile. During the High Flow and Post-High Flow treatments, water temperatures during the peak flow period were warmer than previous treatments, but still within optimal ranges for rearing (for fish that remained during/after the high flows). The cause of the warmer temperatures was attributed to the influence of reservoir operations, dam release discharges and ambient temperatures. There is a gradient of temperature associated with distance from the dam (e.g., Reach 1 temperatures were ~4°C cooler on average than Reach 4 in fall 2019-2021).
	How do differences in physical conditions in aquatic habitat resulting from instream flow regime influence community composition and productivity of primary and secondary producers in Lower Bridge River?	• Spring to early summer peak flows during the High Flow Years of 2016 – 2018 caused a 69% decline in the density of benthic invertebrates in the Lower Bridge River at baseflow in the fall. Recovery of the community was found a year later. Density and composition of the benthic invertebrate community in the Post-High Flow Years (2019 – 2021) was the same as that found prior to the High Flow Years. • Biomass accrual by periphyton that was measured in the fall after high spring-summer peak flows during the High Flow Years of 2016 – 2018 was not affected by those high peak flows. While scour of periphyton probably occurred during the high flows, recovery occurred within two to four months of disturbance, resulting in no measurable change. • Nutrient enrichment from Pink salmon spawning and carcass decomposition was found to double the density of fish food organisms via trophic upsurge wherein nutrient loading increases production of lower trophic levels. This

Primary Objectives	**Management Questions**	**Results up to and including Year 10 (2021)**
Core Components: To reduce uncertainty about the relationship between the magnitude of flow release from the dam and the relative productivity of the Lower Bridge River aquatic and riparian ecosystem. To provide comprehensive documentation of the response of key physical and biological indicators to alternative flow regimes to better inform decision on the long-term flow regime for the Lower Bridge River. The scope of this program is limited to monitoring the changes in key physical, chemical, and biological productivity indicators in reaches 2, 3, and 4 of the Lower Bridge River aquatic ecosystem.		process was particularly effective in Reach 2 that is typically most nutrient deficient among reaches and receives most spawners based on nutrient concentration proxies for spawner abundance. • Nutrient addition to Reach 2 may be considered a useful option for mitigating losses of fish food organisms in future years when very high flow releases exceeding 100 $m^3 \cdot s^{-1}$ from Terzaghi Dam may be unavoidable. Nutrient addition treatment is best applied near or at low base flow after a high flow event has occurred to optimize recovery of benthos and fish food production.
	How do changes in physical conditions and trophic productivity resulting from flow changes together influence the recruitment of fish populations in Lower Bridge River?	• Juvenile salmonid abundance was highest (overall) under the Trial 1 and 2 flow regimes (in general, production between them was near equivalent, but both impacted Chinook recruitment). Relative to Trial 2, the high flows in 2016-2018 reduced salmonid abundance by 76%. Reductions for mykiss and coho fry were by 76% and 89%, respectively. Mykiss parr abundance was 71% lower and Chinook fry abundance remained low (equivalent to Trial 2). • The return to lower flows (similar to the Trial 2 hydrograph) in 2019-2021 provided for modest recovery of mykiss and coho fry in Reach 3; however, the abundance of mykiss parr and Chinook fry declined relative to the High Flow period. Recovery of all species in reaches 2 and 4 was negligible. Abundances for each species and age class (except coho fry) increased in Reach 1 between 2019 and 2021. Reach 1 is the length of reaches 2, 3 and 4 combined and produced 1/3 of the total fish in the study area. • Mean biomass in the Post-high flow years was the lowest for any flow treatment to-date; approx. 188 kg lower than the High Flow period and 767 kg lower than Trial 2. • Based on stock-recruit analysis, production for Chinook and coho is characterized by a different curve and asymptote (i.e., carrying capacity) for each flow treatment. It appears that production was not stock-limited in most study years; however, conclusions are constrained for the High Flow and Post-High Flow treatments due to a limited number of datapoints (*n*= 3) for defining the initial slope and asymptote of those curves. • Genetic stock identity was analyzed for 106 Chinook fry sampled in 2020. Analysis results suggested that the majority were from pure form Bridge River parents and at least 13% were admixed and attributed to the successful spawning of mid and upper Fraser River Chinook salmon strays in the Bridge River in 2019.
	What is the appropriate 'shape' of the descending limb of the 6 cms hydrograph, particularly from 15 cms to 3 cms?	• 2021 flow rampdown monitoring between 15 and 3 $m^3 \cdot s^{-1}$ (i.e., the WUP Trial 2 range) further affirmed what has been learned about fish stranding risk and documented in past reports and the fish stranding protocol. • Modified Operations (2016–2021) results also reaffirmed that ~13 $m^3 \cdot s^{-1}$ is the approx. flow threshold below which stranding risk tends to increase. As such, slower (i.e., WUP) ramp down rates are likely warranted below that level. Above this threshold there is likely flexibility to implement faster ramp rates to reduce flows more quickly without increasing the incidence of stranding significantly.
Modified Operations: High Flow Monitoring	Do flow releases from Terzaghi Dam under the modified flow regime affect water	• The high flows in 2021 resulted in elevated water temperatures, turbidity and %TGP levels (relative to background) in reaches 3 and 4 and caused some erosion and substrate recruitment along the wetted edge at various sites that were monitored in reaches 2 – 4 during the peak release period.

Primary Objectives	**Management Questions**	**Results up to and including Year 10 (2021)**
	quality or cause erosion in the Lower Bridge River? If so, what are the potential effects on fish and what mitigation options are available?	• Summer temperature differences (based on the logger data) were within optimal ranges for juvenile salmonid rearing cited in the literature, the turbidity changes (based on the spot measurements) were lower in 2021 than in 2017 and 2018, the erosion tended to be at existing alluvial slides, and the %TDG levels were below the thresholds for triggering a mitigation response according to Table 1 of BC Hydro's Total Dissolved Gas Management Strategy (i.e., BC Hydro 2014). • As such, within the scope of information available in 2021, we don't have any data to suggest a direct adverse effect of the measured temperatures, turbidity levels, %TDG saturation or bank erosion across the high flow period on fish in 2021.
Modified Operations: High Flow Ramp Down Monitoring and Stranding Risk Assessment	How does the risk of fish stranding during LBR ramp downs vary with discharge?	• Above a threshold of ~13 $m^3 \cdot s^{-1}$, the fish stranding risk (per 1 $m^3 \cdot s^{-1}$ increment of flow change) was consistently low (or occasionally moderate). Conversely, below the 13 $m^3 \cdot s^{-1}$ threshold, the fish stranding risk was more consistently high. • This difference likely provides the opportunity to continue to implement (and monitor) faster ramp rates for higher flows (>13 $m^3 \cdot s^{-1}$). • * Important caveat: juvenile fish abundance was substantially reduced overall in 2016 – 2018 and 2021, which likely affected salvage results following high flows during those years.
	How does the risk of fish stranding during LBR ramp downs vary by reach?	• Under previous flow trials (≤15 $m^3 \cdot s^{-1}$), differences in the number of fish salvaged (per 100 m^2) among reaches was significant. Reach 4 densities were more than double Reach 3 densities. • Differences among reaches in the high flow range (>15 $m^3 \cdot s^{-1}$) were also apparent but they were smaller. Slightly higher densities were observed in Reach 4, followed by Reach 3, and then Reach 2 and Reach 1. Note: under the high flow range (>15 $m^3 \cdot s^{-1}$) in 2021, among all four reaches, fish were only salvaged at one location (i.e., near the fish fence site; Rkm 26.1) in Reach 3.
	How does the risk of fish stranding during LBR ramp downs vary with ramping rate and stage change?	• At higher ramp rates up to 5.0 cm/hr implemented in 2017-2019 and 2021, there was no appreciable difference in fish stranding risk relative to lower rates (≤2.5 cm/hr) across the high flow range tested: 102.0 to 15.0 $m^3 \cdot s^{-1}$). • These results suggest there is opportunity to continue testing higher rates (up to 5.0 cm/hr with fish salvage crews on the ground) across the high flow range going forward without unduly increasing the fish stranding risk. * Important caveat: the abundances of juvenile salmonids during the modified operations years were low overall, which could have influenced results.
	How does the risk of fish stranding during LBR ramp downs vary by river bank?	• At high flows, site distribution was nearly equal (45% river left; 55% river right), whereas at low flows, the distribution was more skewed (76% river left; 24% river right). We speculate that these differences at the lower flows are due to human-caused effects (e.g., river access, gold mining, gravel placements, etc.) on habitats at low elevations, rather than natural causes.

Primary Objectives	Management Questions	Results up to and including Year 10 (2021)
	Are there opportunities to minimize or mitigate the risk of fish stranding during ramp downs in the Lower Bridge River?	• The primary opportunity (or most conservative approach) for minimizing or mitigating the risk of fish stranding is by implementing the ramping rates referenced in the WUP (i.e., ≤2.5 cm/hr) and having fish salvage crews actively salvaging fish in each of the reaches downstream of the dam. • In some cases, such as during years with high flows, there can be additional rationale for ramping the flows down faster in order to reach more optimal summer rearing flows (i.e., ≤15 $m^3 \cdot s^{-1}$) more quickly following peak flows. With the data for high flows available from 2016-2018 and 2021, there is some evidence for when faster ramping rates can be applied without unduly increasing fish stranding risk. • Implementation of faster ramp rates should be accompanied by ramp monitoring and fish salvaging to improve certainty about effects on stranding risk.
Modified Operations: Juvenile Salmonid Habitat Availability and Displacement	How does juvenile salmonid habitat availability in the Lower Bridge River change with discharge under the modified flow regime?	• Based on the results of the pilot-level sampling in 2018 and 2021, there was insufficient information to answer this management question (i.e., based on the field data collected for this component alone). • Depths and velocities were highly variable across the different mainstem sites and tended to be more consistent in the off-channel sites • There were some sites (both high quality and low quality) that provided suitable depths and velocities for rearing (based on LBR HSI curves for coho, chinook and mykiss) across the full range of high flows. Since data collection for this component has not been collected since 2018, refer to the Year 7 report for more information (Sneep et al. 2019). • A better approach to addressing this question may be to use BC Hydro's Telemac2D model
	How does habitat use by juvenile salmonids change with discharge under the modified flow regime?	• Under both the Trial 2 peak flows in 2020, and the higher peak flows in 2021, there were no substantial changes in density in the off-channel sites that would point to significant immigration or emigration in response to main channel flows for any of the target species and age classes. • Analysis of the available data, as well as what we can learn from them, is limited since the sites were open, sampled in a single pass, and there was no means of determining any changes in capture efficiencies among sites or survey dates. • Even if differences or changes in lineal densities occur between years, or sessions within a year, there are a wide range of factors to be considered for discerning the potential causes (including life history events such as emergence or outmigration, or factors which affect capture efficiencies, to name a few) which are not possible based on the current methods.

Table of Contents

1. Introduction

1.1. Background

The context for the Lower Bridge River flow experiment and its associated aquatic monitoring program is only briefly summarized here. It has been more fully described in earlier manuscripts by Failing et al. (2004) and (2013), and Bradford et al. (2011).

The Lower Bridge River (LBR) is a large glacially fed river that has been developed and managed for hydroelectricity generation by BC Hydro and its predecessors since the 1940s. Prior to impoundment, the Bridge River had a mean annual discharge (MAD) of 100 cubic meters per second ($m^3 \cdot s^{-1}$) and maximum flow during spring freshets of up to 900 $m^3 \cdot s^{-1}$ (Hall et al. 2011). Following the completion of Terzaghi Dam in 1960 there was no continuous flow released into the LBR channel due to the complete diversion of water stored in Carpenter Reservoir (upstream of the dam) into Seton Lake in the adjacent valley to the south. This resulted in the dewatering of just over 3 kilometres (km) of Bridge River channel immediately downstream of the dam, other than during periodic mid-summer spills caused by high inflows (Higgins & Bradford 1996). On average, these spill events occurred approximately two to three times per decade (Figure 1.1).

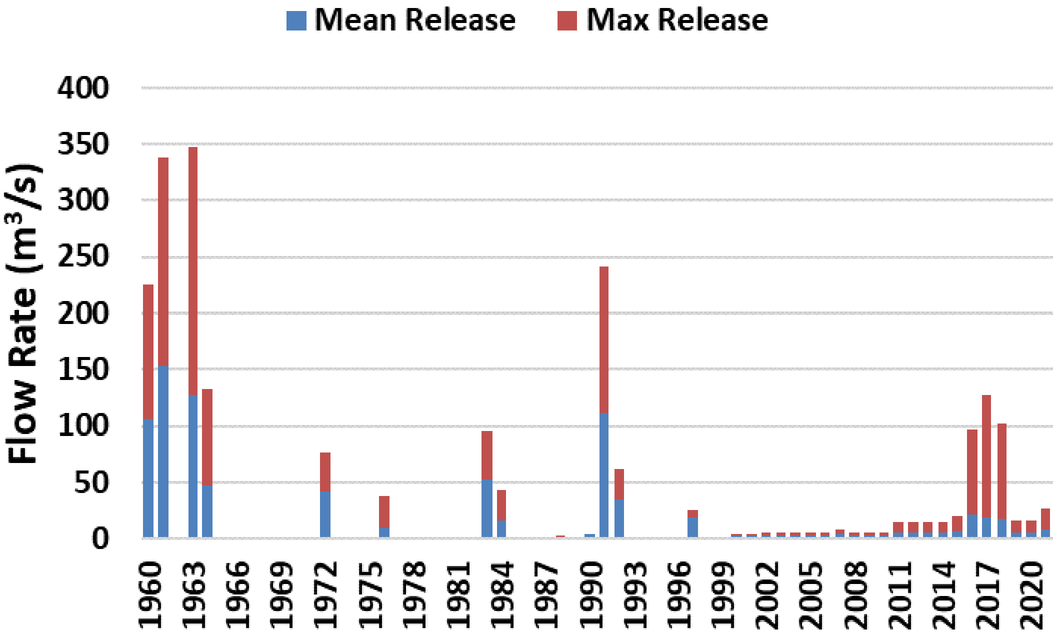


Figure 1.1 Frequency of spill and flow release events from Terzaghi Dam into the Lower Bridge River following impoundment in 1960.

Downstream of the dewatered reach, the river had a low but continuous and relatively stable streamflow, with groundwater and five small tributaries cumulatively providing a MAD of approximately 0.7 $m^3 \cdot s^{-1}$. Fifteen km downstream from the dam, the unregulated Yalakom River joins the Bridge River and supplies, on average, an additional 4.3 $m^3 \cdot s^{-1}$ (range = 1 to 43 $m^3 \cdot s^{-1}$) to the remaining 25 km of Lower Bridge River.

Starting in the 1980s and following spill events from Terzaghi Dam during the 1990s, concerns about impacts of dam operations (particularly the episodic spill events) and the lack of a continuous flow release on the aquatic ecosystem of the Lower Bridge River were raised by First Nations representatives, local stakeholders, and fisheries agencies. According to the magnitude and timing of the spill, the effects of these events likely included: flooding the river channel outside of the typical freshet period, scouring of the streambed, flushing gravels and other sediments, fish entrainment from the reservoir into the river, and fish stranding as the spill flows diminished. Beyond the information provided by fish salvage surveys, the scope of effects from past spills on the aquatic ecosystem were not well understood but were recognized to be significant and warranted mitigation.

In 1998, an agreement between BC Hydro and regulatory agencies (stemming from litigation pertaining to spills in 1991 and 1992) specified that an environmental flow be implemented with the goal of restoring a continuous flow to the dewatered section below the dam and optimizing productivity in the river. However, information was not available to determine what volume of flow and what hydrograph shape would provide optimal conditions for fish production and other ecosystem benefits. This was considered a key uncertainty which precluded the ability to make a flow decision at that time. Therefore, initiation of the continuous release was set up as a flow experiment with an associated monitoring program designed to assess ecosystem response to the introduction of flow from Carpenter Reservoir. The continuous flow release from Terzaghi Dam was initiated by BC Hydro in August 2000.

1.2. The Flow Experiment

The original flow experiment consisted of 2 flow trials: a 3 $m^3 \cdot s^{-1}$ mean annual release (Trial 1; August 2000 to March 2011) and a 6 $m^3 \cdot s^{-1}$ mean annual release (Trial 2; April 2011 to December 2015). The flows for each trial were released according to prescribed hydrographs (Figure 1.2) that were designed by an interagency technical working group. Monthly flows during Trial 1 ranged between a fall/winter low of 2 $m^3 \cdot s^{-1}$ (November to March) to a late spring peak of 5 $m^3 \cdot s^{-1}$ (in June). During Trial 2 the fall/winter low flow was 1.5 $m^3 \cdot s^{-1}$ (October to February) and peak flows were approximately 15 $m^3 \cdot s^{-1}$ for all of June and July.

Reduction of the flow release (ramping) for Trial 1 was conducted in small increments following the peak in mid June down to 3 $m^3 \cdot s^{-1}$ by the end of August, and then down to the fall/winter low in mid to late October. Ramping for the Trial 2 flows occurred ca. weekly during August from 15 to 3 $m^3 \cdot s^{-1}$, and the final ramp down from 3 to 1.5 $m^3 \cdot s^{-1}$ typically occurred in early October (Sneep and Hall 2012b; McHugh and Soverel 2017).

The main intent of this monitoring program was to assess the influence of each of the flow release trials (the flow experiment) on fish resources and the aquatic ecosystem of the Lower Bridge River. Monitoring was also conducted for four years during the Pre-flow period (dubbed “Trial 0”; May 1996 to July 2000) to document baseline conditions when the mean annual release from the dam was 0 $m^3 \cdot s^{-1}$. Since the wetted portion of the channel between the dam and the Yalakom River confluence was wetted by tributary and groundwater inflows during the pre-flow period, it was important to document existing productivity so the results of the flow trials could be understood in context.

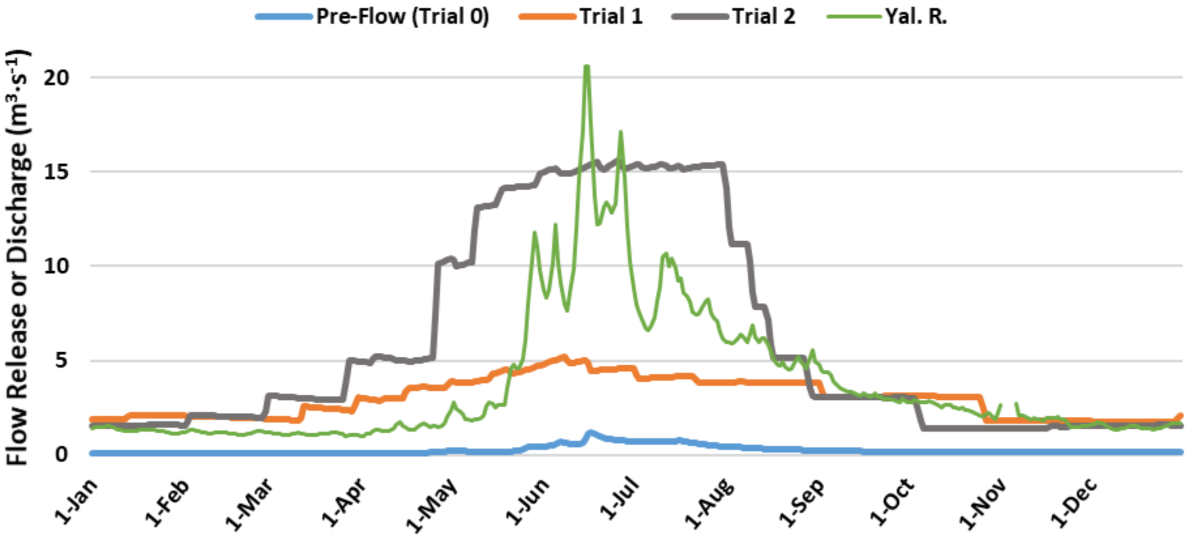


Figure 1.2 **Mean daily releases from Terzaghi Dam for Trial 1 and Trial 2 during the flow experiment. Typical hydrograph shapes during the Pre-flow period and for the unregulated Yalakom River discharges are included for reference.**

Decisions on the magnitude of peak flows for the flow trials were constrained by morphological characteristics of the channel below Terzaghi Dam. In several areas the channel is confined by narrow canyons and characterized by high gradients; conditions that are not conducive for maintaining spawning substrates or creating rearing habitats at high flows. Prior to impoundment, natural discharges were generally much higher in the Lower Bridge River: summer flows ranged between 100 and 900 $m^3 \cdot s^{-1}$ (mean peak flow was ~400 $m^3 \cdot s^{-1}$; Bradford et al. 2011). However, historical records indicate that most of the best fish habitat (including spawning areas for salmon) were located upstream of the dam site and are now flooded by Carpenter Reservoir. The river below the dam site was primarily used as a migratory corridor for anadromous species (O’Donnell 1988). After construction of Terzaghi Dam, reduced flows in the high-gradient migratory corridor provided spawning and rearing habitat, and habitats above the dam were no longer accessible. Due to this change in the location of habitat, pre-impoundment flows were not considered appropriate benchmarks for the flow trials.

Additionally, available data from the Pre-flow period indicated that the production of salmonids was very high in the groundwater-fed section above the Yalakom River confluence under low flow conditions. Discharge at the top of this section was generally ≤ 1 $m^3 \cdot s^{-1}$, yet spawners of all species were able to reach the upper extent of the inflow and juveniles were distributed throughout the wetted extent of the channel. Juvenile salmonid densities were among the highest in the province of BC and average biomass values (g/m^2) were more than double typical values for trout and salmon in western North America (Bradford et al. 2011). This remarkable pre-flow productivity also served as important context for designing the trial flows. The technical working group sought to ideally strike a balance between creating new habitat (by rewetting the previously dry section below the dam and enlarging the wetted area of the river in general) without unduly reducing the exceptional productivity in the wetted section above the Yalakom River confluence.

1.3. Modified Operations

During implementation of the Trial 2 flows, BC Hydro identified issues with some of their infrastructure associated with water storage and flow conveyance within the Bridge-Seton hydroelectric complex. As a result, the storage of water in Downton Reservoir and conveyance of flows from Carpenter Reservoir to Seton Lake (via the diversion tunnels and generating units at Bridge 1 and 2) had to be reduced to allow for the affected infrastructure to be rebuilt or replaced.

The reduction of water storage and flow diversion above Terzaghi Dam meant that additional flow needed to be passed into the Lower Bridge River above the amounts prescribed for the flow experiment (described in Section 1.2) at least in some years according to the timing and magnitude of inflows. In years with normal or below average inflows, the Trial 2 hydrograph would remain the target for flow conveyance. As such, flow magnitudes and hydrograph shapes have tended to be more variable during the period known as “Modified Operations” which began in 2016.

Delivery of higher flows occurred for three consecutive years from 2016 to 2018. For this period of Modified Operation years, mean annual flows from the dam were approximately 22, 19 and 18 $m^3 \cdot s^{-1}$ (peak flows = 97, 127 and 102 $m^3 \cdot s^{-1}$), respectively (Figure 1.3). These peak flows were much higher than the Trial 1 and Trial 2 hydrographs but were within the range of spill flows from past events since the completion of Terzaghi Dam in 1960 (Figure 1.1).

The ascending limb of the high flow hydrograph in 2016 started on 17 March, peaked in mid June, and returned to Trial 2 levels by 25 July (2016 high flow duration = 131 days). The high flows in 2017 had a higher peak, but a shorter duration relative to 2016: Flows increased above the Trial 2 hydrograph on 24 May, peaked across the month of June, and were ramped back down to Trial 2 levels on 21 July (2017 high flow duration = 59 days). High flows in 2018 began on 10 May, peaked in late June, and were ramped back down to Trial 2 levels on 1 August (2018 high flow duration = 83 days). Outside of the high flow period in 2016–2018, the flow release from mid summer through fall and winter was identical to the Trial 2 hydrograph shape. The years from 2016 to 2018 are collectively referred to as the “High Flow Years” in this report.

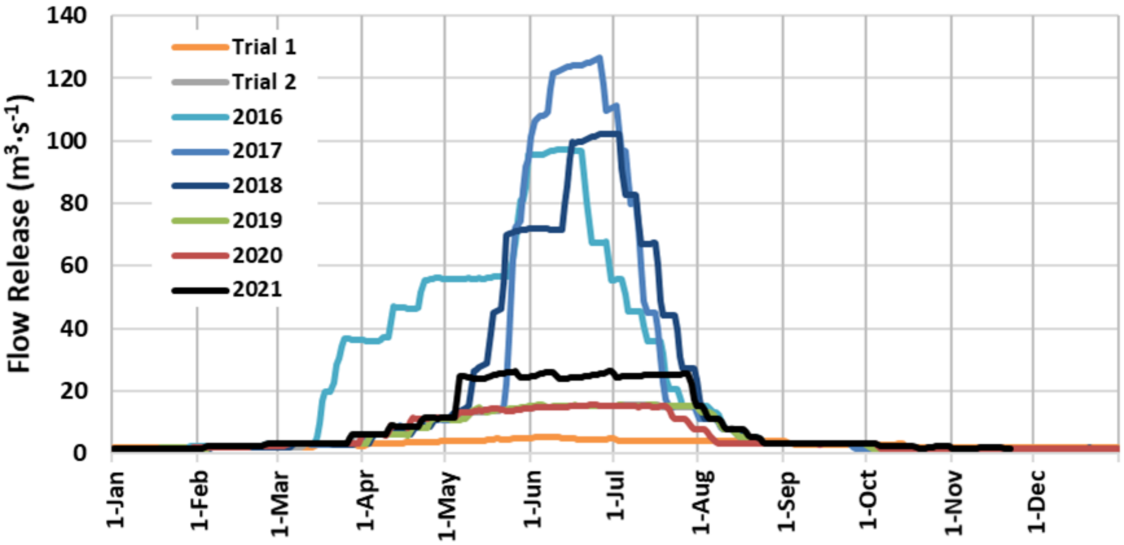


Figure 1.3 **Terzaghi Dam flow release hydrograph shapes for Modified Operations years (2016–2021). Mean daily releases for the Trial 1 and Trial 2 hydrographs are shown for reference. Peak flow releases in Year 10 (2021) plateaued at a higher level (~26 $m^3 \cdot s^{-1}$) than the Trial 2 peak (15 $m^3 \cdot s^{-1}$) but were lower than the High Flow peaks in 2016-2018.**

From 2019 to 2021, the combination of reduced inflows and reservoir management allowed for a return to lower peak flows from Terzaghi Dam (Figure 1.3). Peak releases in 2019 and 2020 were ~15 $m^3 \cdot s^{-1}$ (i.e., the same as Trial 2; high flow duration = 0 days), and peak releases in 2021 were ~25 $m^3 \cdot s^{-1}$ (i.e., higher than the Trial 2 peak but far lower than the high flow peaks from 2016 to 2018; high flow duration = 87 days). As in the Trial 2 years (2011 to 2015), flows ramped up from the winter lows from late March to early May, peaked across the months of May, June and July, and were ramped back down to 3 $m^3 \cdot s^{-1}$ between late July and mid August, and then to 1.5 $m^3 \cdot s^{-1}$ at the beginning of October. Despite their occurrence during the modified operations period (years ≥2016) and similar flow delivery characteristics to the Trial 2 releases, the flow releases in 2019-2021 were considered a new treatment (called the Post-High Flow years) in the analyses and results provided in this report since they represented the return to lower peak flows relative to the channel-altering high flows from 2016 to 2018.

Figure 1.4 shows mean trial flows on a logarithmic scale to compare differences in the shapes of the flow release hydrograph between trials. The High Flow years produced a pronounced bell-shaped hydrograph with steep ascending and descending limbs and highest peak release among treatments. The Trial 1 shape was a flattened bell-shaped hydrograph appearing more like a shallow dome with low slopes on the ascending and descending limbs and lowest peak release among trials. The Trial 2 hydrograph was in between, having a moderate bell shape, moderate rates of ascending and descending limbs and peak water releases in between Trials 1 and 3. For the reasons noted above, the line characterizing the 2019-2021 (Post-High Flow) releases was very similar to the Trial 2 mean throughout the year (Figure 1.4).

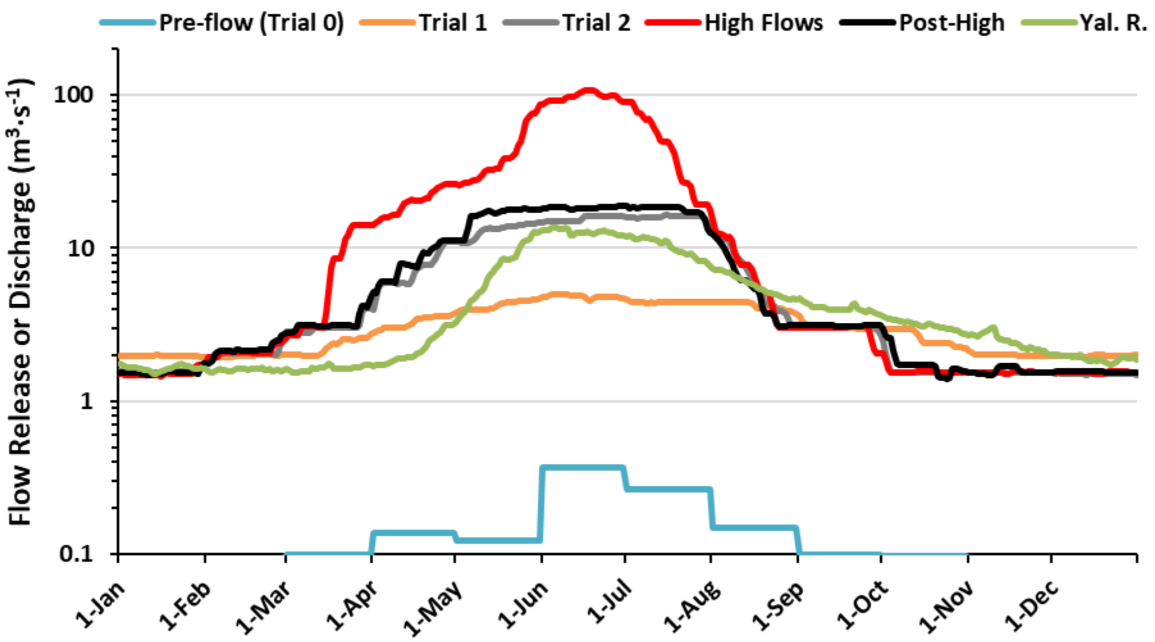


Figure 1.4 **Mean daily flow release from Terzaghi Dam among all years in each flow Trial. Mean daily flow among all years (1996 – 2021) in the Yalakom River is shown for reference. Note the log scale on the Y axis.**

The different magnitudes of flow by trial in the Bridge River are compared to those in the Yalakom River, where flow is not regulated (Table 1.1). Mean annual flow in the Yalakom River was 4.2 to 5.2 $m^3 \cdot s^{-1}$ among all trials, which was between the mean annual flow release to the Bridge River in Trials 1 and 2. The average minimum flows were approximately 1 to 2 $m^3 \cdot s^{-1}$ in both the water release to the Bridge River and in the Yalakom River. Average peak flow in the Yalakom River was 22 to 25 $m^3 \cdot s^{-1}$ among all blocks of Trial years, which was about 50% greater than the peak flow release under the Trial 2 hydrograph and comparable to the peak flow release in 2021.

Table 1.1 Flow statistics by Trial in the Bridge River and in the Yalakom River.

River	Trial number (years) [a] (sample size)	Flow statistic ± standard deviation		
		Mean annual water release or flow ($m^3 \cdot s^{-1}$)	Average minimum water release or flow ($m^3 \cdot s^{-1}$)	Average peak water release or flow ($m^3 \cdot s^{-1}$)
Lower Bridge	0 (1996–1999) (n=4)	0.6 ± 1.3	0	6.3 ± 12.5
	1 (2001–2010) (n=10)	3.1 ± 0.2	1.8 ± 0.1	5.4 ± 1.1
	2 (2012–2015) (n=4)	6.1 ± 0.3	1.1 ± 0.7	16.8 ± 2.6
	3 (2016–2018) (n=3)	19.5 ± 2.1	1.4 ± 0.1	108.7 ± 15.7
	4 (2019–2021) (n=3)	6.8 ± 1.6	1.3 ± 0.1	19.2 ± 6.2
Yalakom	0 (1996–1999) (n=4)	5.1 ± 1.2	1.2 ± 0.1	25.1 ± 12.7
	1 (2001–2010) (n=10)	4.2 ± 0.9	1.2 ± 0.2	22.5 ± 10.7
	2 (2012–2015) (n=4)	4.5 ± 0.5	0.7 ± 0.4	21.9 ± 5.1
	3 (2016–2018) (n=3)	5.2 ± 0.4	0.9 ± 0.3	25.1 ± 1.8
	4 (2019–2021) (n=3)	4.8 ± 0.2	0.8 ± 0.2	24.6 ± 5.0

[a] Years 2000 and 2011 were omitted because they were transition years between flow treatments and therefore represent incomplete years for calculations of flow statistics.

At least until the end of the Modified Operations period (i.e., until BC Hydro infrastructure is sufficiently rebuilt or replaced to allow a return to normal WUP operations – the timing of which is uncertain at this point), spring flows could continue to be more variable across years than they were under the flow experiment trials and may require discharges above the Trial 2 peak more frequently. Increases in the maximum Terzaghi Dam discharge may have short and long-term effects on the LBR and aquatic productivity. In the short-term, high discharges have caused increased entrainment at Terzaghi Dam, reduced juvenile salmonid rearing habitat area, erosion and sediment deposition throughout the river, and increased the total number of fish stranded during ramp downs from high flows. In both the short- and long-term, high flows may alter primary and secondary productivity, juvenile salmonid growth and abundance, and salmonid habitat suitability.

1.4. Objectives and Management Questions

The objective of the monitoring program was to reduce uncertainty about the long-term ecological response to the release of continuous flows from Terzaghi Dam into the Lower Bridge River channel. This lack of certainty was an impediment to decision-making on an optimal flow regime and centred around the unknown effects of different flows on aquatic ecosystem productivity. A decision about flow release volumes and hydrograph shape based on invalid judgements would have implications for both energy production and the highly valued ecological resources of the Lower Bridge River. Therefore, the goal of the monitoring program was to resolve the uncertainty by the collection and analysis of scientifically defensible data.

1.4.1. Original (WUP) Management Questions

To guide the program, a set of specifically linked "Management Questions" were developed during the Water Use Planning (WUP) process:

1) **How does the instream flow regime alter the physical conditions in aquatic and riparian habitats of the Lower Bridge River ecosystem?**

   Changes in the physical conditions regulate the quantity and quality of habitats for aquatic and riparian organisms. Documenting the functional relationships between river flow and physical conditions in the habitat is fundamental for identifying and developing hypotheses about how physical habitat factors regulate, limit or control trophic productivity and influence habitat conditions in the ecosystem.

2) **How do differences in physical conditions in aquatic habitat resulting from the instream flow regime influence community composition and productivity of primary and secondary producers in the Lower Bridge River?**

   Changes in the flow regime are expected to alter the composition and productivity of periphyton and invertebrate communities. Understanding how these physical changes influence aquatic community structure and productivity are important as they act as indicators to evaluate "ecosystem health" and the trophic status of the aquatic ecosystem in relation to provision of food resources for fish populations.

3) **How do changes in physical conditions and trophic productivity resulting from flow changes together influence the recruitment of fish populations in the Lower Bridge River?**

   Changes in the flow regime can have significant effects on the physical habitat and trophic productivity of the aquatic ecosystem and these two factors are critical determinants of the productive capacity of the aquatic ecosystem for fish. Understanding how the instream flow regime influences abundance, growth, physiological condition, behavior, and survival of stream fish populations helps to explain observations of changes in abundance and diversity of stream fish related to flow alteration.

4) **What is the appropriate 'shape' of the descending limb of the Trial 2 (6 $m^3 \cdot s^{-1}$ MAD) hydrograph, particularly from 15 $m^3 \cdot s^{-1}$ to 3 $m^3 \cdot s^{-1}$?**

   Inherent in the development of the Trial 2 hydrograph, was uncertainty regarding the risk of fish stranding given the relative magnitude of ramp-downs during the months when flows were reduced (i.e., August and October). Some information on the incidence of fish stranding between 8.5 and 2 $m^3 \cdot s^{-1}$ *had been documented during the Trial 1 period (Tisdale 2011a, 2011b). However, there was limited existing information on fish stranding in the discharge range from 15* $m^3 \cdot s^{-1}$ *to 8.5* $m^3 \cdot s^{-1}$ *and the types of habitats in this flow range. The collection of information on the risk of fish stranding at each stage of flow reduction is needed for assigning flow ramping rules during the descending limb of the annual hydrograph.*

These management questions were originally intended to improve understanding of LBR aquatic productivity under the Trial 1 and Trial 2 hydrographs. However, the management questions are still considered relevant for understanding the effects of modified operations from Terzaghi Dam (i.e., the high flow years from 2016 to 2018 and the post-high flow years from 2019 to 2021) even though these operations were not within the context of the flow experiment as originally conceived.

Changes to Chinook salmon emergence timing and life history have been observed in the LBR over the course of the flow trials, but these changes were not specifically addressed in the original BRGMON-1 WUP management questions. Two new management questions to address uncertainties about the observed changes were included in a BRGMON-1 Terms of Reference Revision 1 (BC Hydro 2018), as follows:

5) **Do increased water temperatures and early emergence associated with Terzaghi Dam flow releases affect the survival of juvenile Chinook salmon in the Lower Bridge River?**

   BRGMON-1 monitoring results have identified increased fall water temperatures associated with minimum flow releases under the Trial 1, Trial 2, High Flow and Post-High Flow hydrographs (relative to pre-flow conditions). Based on predicted emergence timing from temperature exposures during incubation under the release coupled with the collection of recently emerged fry during late fall and early winter sampling surveys, the flow release thermal regime has advanced the emergence timing of Chinook salmon fry in the LBR, and most notably in the upper portion of the study area. These changes have also coincided with reduced juvenile abundance for this species. However, there is uncertainty about the extent to which early emergence has affected the survival of Chinook salmon since the observed decline in juvenile Chinook salmon abundance under flow release conditions also coincided with reduced adult returns to the Lower Bridge River and other Mid-Fraser populations.

6) **What freshwater rearing habitats are used by Lower Bridge River juvenile Chinook salmon and is rearing habitat use influenced by Terzaghi Dam flow releases?**

   In addition to potential early emergence effects on Chinook salmon survival described for question #5, other explanations for reduced juvenile abundance in the fall may also include life history changes (e.g., timing of outmigration) or habitat use changes (e.g., rearing in the Fraser River rather than the LBR).

1.4.2. Modified Operations Management Questions

Due to the modified operations resulting from the La Joie Dam and Bridge River Generation issues, additional monitoring programs with new management questions were created in 2016 to guide the short-term high flow monitoring programs and inform the LBR impact assessment and mitigation planning. This information is intended to inform the BC Hydro-St'at'imc "Joint Planning Forum" (JPF) which plans and coordinates monitoring and mitigation activities for the duration of the modified operations. As indicated in the BC Hydro Scope of Services document, it is noted that the following management questions were developed to guide reporting but maybe updated according to the needs of the JPF.

Water Quality, Erosion and Entrainment Monitoring

High flow water releases from Terzaghi Dam can temporarily affect water quality and cause erosion in the Lower Bridge River. The management question to address these effects is:

1) Do flow releases from Terzaghi Dam under the modified flow regime affect water quality or cause erosion in the Lower Bridge River? If so, what are the potential effects on fish and what mitigation options are available?

Fish Salvage and Stranding Risk Assessment

Previously, fish stranding had only been monitored under the range of WUP flows (<20 $m^3 \cdot s^{-1}$) which were delivered from 2000 to 2015. As a result of the high flows in 2016 and 2017, stranding risk also needed to be assessed at discharges >15 $m^3 \cdot s^{-1}$. Management questions created to guide this monitoring were:

2) How does the risk of fish stranding during LBR ramp downs vary with discharge, reach, river bank, ramping rate, and stage change?
3) Are there opportunities to minimize or mitigate the risk of fish stranding during ramp downs in the Lower Bridge River?

Juvenile Salmonid Habitat Availability and Displacement

The high flows delivered from 2016 to 2018 have impacted juvenile salmonid rearing habitats by introducing higher velocities throughout more of the channel and mobilizing sediment resulting in additional areas of scour and deposition. The effects of these changes were expected to include potential changes to rearing habitat area, displacement of fish out of the study area, and/or life history changes in the longer term. In response to (or anticipation of) these potential changes, the following management questions were developed:

4) How does juvenile salmonid habitat availability in the Lower Bridge River change with discharge under the modified flow regime?
5) How does habitat use by juvenile salmonids change with discharge under the modified flow regime?

Substrate Mobilization, Deposition and Composition Monitoring

During the previous flow trials, the range of flow magnitudes delivered from the low-level outlet at Terzaghi Dam (1.5 to 15 $m^3 \cdot s^{-1}$) were below the threshold for mobilizing sediment materials within the LBR channel or recruiting new materials from the banks. High flows delivered in 2016, 2017 and 2018 were expected to exceed this threshold, which had not previously been described, requiring monitoring and assessment to define the threshold and characterize sediment transport for informing decisions on flow magnitudes and hydrograph shapes. The management question to the guide the work for this component were:

- To what extent does substrate movement under the modified flow regime affect the distribution, availability or suitability of juvenile rearing in the Lower Bridge River and what potential mitigation opportunities exist to minimize or mitigate any measured effects?

1.5. Study Area

The Bridge River drains a large, glaciated region of the Coast Range of British Columbia and flows eastward, joining the Fraser River near the town of Lillooet. The river has been impounded by the La Joie and Terzaghi dams which have segmented the river into three main sections: The Upper Bridge River and Downton Reservoir (above La Joie Dam); the Middle Bridge River and Carpenter Reservoir (above Terzaghi Dam); and the Lower Bridge River (downstream of Terzaghi Dam). The Lower Bridge River between Terzaghi Dam and the confluence with the Fraser River is approximately 41 km long and is currently the only section accessible to anadromous fish.

The Lower Bridge River was divided into four reaches by Matthew and Stewart (1985); their reach breaks are defined in Table 1.2. Monitoring for this program conformed to these reach break designations and from 1996 to 2018 focused on the section of river between Terzaghi Dam and the bridge crossing upstream of Camoo Creek (i.e., reaches 4, 3 and 2). Starting in 2018, measurement of periphyton and benthic invertebrate metrics was extended to include sites in the lower portion of the Yalakom River to allow comparison between the flow controlled Lower Bridge River and the unregulated Yalakom River. In 2019, monitoring was also extended to include Reach 1 (including 4 periphyton and benthic invertebrate monitoring sites and 12 juvenile stock assessment sites), such that all 4 reaches of the Lower Bridge River were covered with comparable monitoring effort. The overall study area is illustrated in Figure 1.5. UTM coordinates for the thirteen index monitoring locations and 61 juvenile stock assessment sites are provided in Appendix A.

Table 1.2 Reach designations and descriptions for the Bridge River below Terzaghi Dam.

Reach	Boundary (Rkm)		Length (km)	Description
	Downstream	Upstream		
1	0.0	19.0	19.0	Fraser River confluence to Camoo Creek
2	19.0	26.0	7.0	Camoo Creek to Yalakom River confluence
3	26.0	37.7	11.7	Yalakom R. confl. to upper extent of groundwater inflow
4	37.7	40.9	3.2	Upper extent of groundwater inflow to Terzaghi Dam

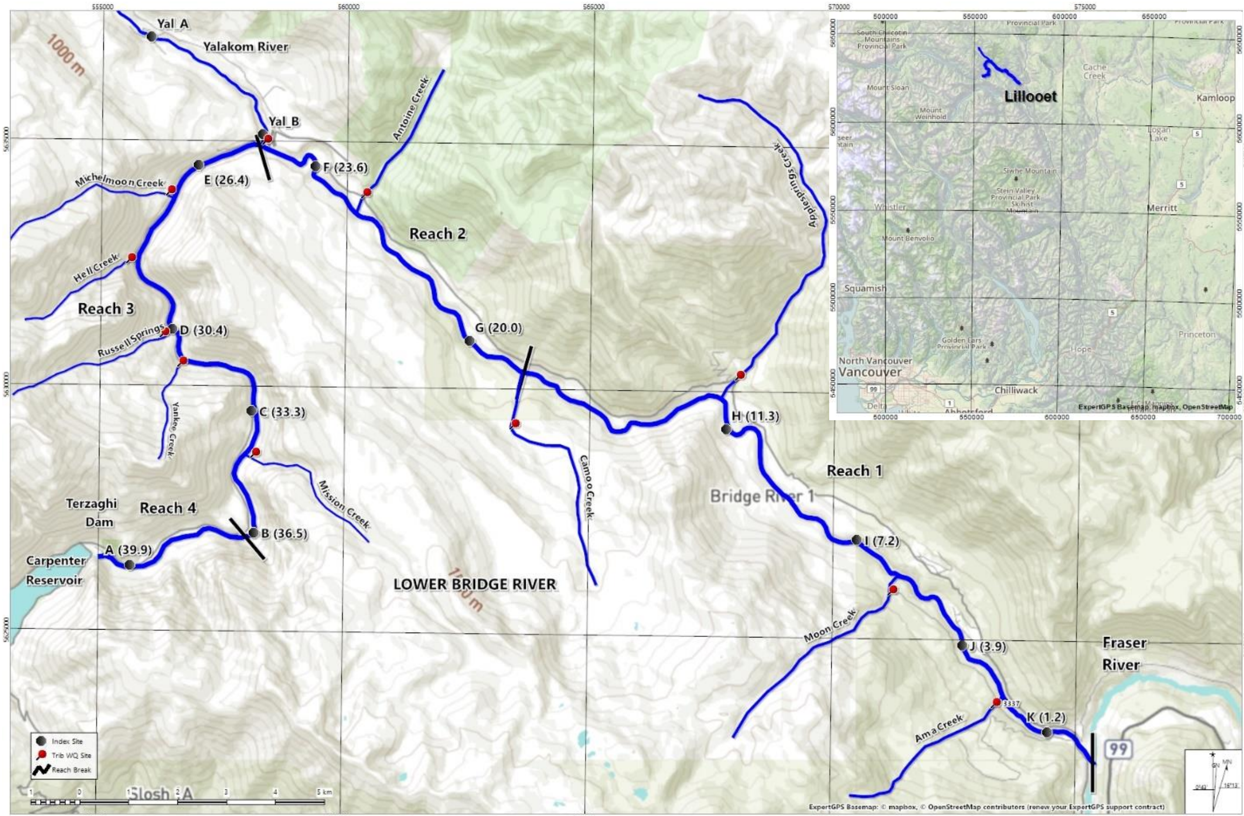


Figure 1.5 **The Lower Bridge River downstream of Terzaghi Dam near Lillooet, British Columbia. Reaches are labelled 4 (upstream) through 1 (downstream). Index sampling sites are labelled as distances upstream of the Fraser River and correspond to the following letters: 39.9 km (A), 36.5 km (B), 33.3 km (C), 30.4 km (D), 26.4 km (E), 23.6 km (F), 20.0 km (G), 11.3 km (H), 7.5 km (I), 3.9 km (J) and 1.2 km (K). Yal_A and Yal_B are the two index sites that were established in the Yalakom River in 2018. Tributary water sampling locations are shown as red pins. The inset map in the top-right corner frames the location of the sampling area within the context of southwestern British Columbia.**

Prior to initiation of the continuous flow release at the start of the flow experiment (i.e., August 2000), Reach 4 was the previously dry section immediately below the dam (length = 3.2 km). Tributary inflows to this reach were negligible, so flow was mostly from water released at Terzaghi Dam. Reach 3 was the groundwater- and tributary-fed reach extending down to the Yalakom confluence (length = 11.7 km). The tributaries to Reach 3 include: Mission Creek, Yankee Creek, Russell Springs, Hell Creek and Michelmoon Creek. These inflow sources were small, so discharges in this reach prior to the flow release were low (~1% of pre-regulation MAD) and release flows have dominated since the start of the flow trials. Flows in Reach 2 (length = 7.0 km) include the inflow from the Yalakom River, the largest tributary within the study area which contributes between approximately 1 and 45 $m^3 \cdot s^{-1}$ at the top of Reach 2 (mean discharge = 4.6 $m^3 \cdot s^{-1}$). One other small tributary, Antoine Creek, flows into Reach 2. Reach 1 (length = ~19.0 km) receives tributary inflow from Camoo Creek, Applesprings, Moon Creek and Ama

Creek. Relative to the differences among trials in the other reaches, discharge rates in Reach 1 were expected to be fairly similar to Reach 2.

1.6. Study Period

Data collection in 2021 was conducted between March and November according to each monitoring component listed in Table 1.3. Certain components that were measured by loggers (i.e., water temperature, river stage, and discharge from the dam) were recorded year-round. This report focusses on the data collected in 2021 (Year 10); however, comparisons with previous years and flow trials are included where applicable and relevant.

Table 1.3 Summary of data to be included in BRGMON-1 analysis and reporting for monitoring year 10 (2021). Components that have prior years of data are noted.

Task	Components	Year 10 (2021) Period	Prior Years of Data[1]
Physical Parameter Monitoring	Water temperature; river stage; discharge	Year-round	1996 to 2020
	Turbidity	10 Mar to 1 Dec	2019[2], 2020
Water Chemistry	Nutrients; alkalinity; pH	11 May; 8 Jun; 20,21 Jul; 26-28 Aug; 27 Sep; 25,26 Oct; 29,30 Nov	1996 to 2007, 2020 (Spring/Summer) 1996 to 2020 (Fall)
Primary & Secondary Productivity	Periphyton accrual; benthic invertebrate diversity & abundance	22 Sep to 24 Nov (LBR)	1996 to 2020
		24 Sep to 23 Nov (Yalakom)	2018 to 2020
Juvenile Salmonid Abundance	Annual standing stock assessment	30 Aug to 20 Sep	1996 to 2020
	Juvenile fish density in off-channel habitats	21 to 23 Sep	2018 to 2020
WUP Ramp Down Monitoring	Stage monitoring; fish salvage	3 to 24 Aug & 5-19 Oct	2011 to 2020
High Flow Monitoring	Kokanee entrainment; water quality sampling; sediment erosion & deposition; fish stranding site reconnaissance	10 Jun; 6 Jul; 27 Jul; 30 Aug	2016 to 2018
Juvenile Salmonid Habitat Availability & Displacement	Single-pass, open site electrofishing at pre-selected rearing sites	17,18 Jun 14,15 Jul	2018 to 2020
High Flow Ramp Down & Stranding Risk Assessment	Stage monitoring; fish salvage at flows >15 $m^3 \cdot s^{-1}$	29,30 Jul	2016 to 2018

[1] Results of analyses for prior years of monitoring will only be included in this annual report where relevant for providing context to the Year 10 (2021) results and where this could be supported by the project budget.

[2] Note: While turbidity loggers were deployed in 2019 and data were collected, the loggers were not adequately shielded from ambient light interference so these data were unusable for the analyses and were not included in this report.

2. Methods

2.1. Overview

The purpose of monitoring was to test effects of different flow releases from Terzaghi Dam on benthic assemblages and fish populations among reaches of the Lower Bridge River. Sampling Reaches 2 to 4 was done among all years since 1996 while the Yalakom River was added in 2018 and Reach 1 was added in 2019. Since a control site was not originally included, the study design has relied primarily on before-after comparisons among reaches and examining functional relationships between flow and biological metrics to examine the importance of flow driving biota abundance and diversity. When the flow experiment and associated monitoring program was conceived, the effects of the flow release trials on the aquatic ecosystem were expected to be most strongly observed in reaches 3 and 4. Due to the attenuation of inflows including the Yalakom River inputs, coupled with differences in channel morphology, the effects in reaches 1 and 2 were expected to be more muted. We expected that change in the value of biological metrics or assemblages between trials in reaches 1 and 2 would be less than in reaches 3 and 4 because of interactions from factors other than (or in addition to) flow release from the Terzaghi Dam.

Core sampling methods were the same during all flow treatments and again in Year 10 (2021). Tasks included: 1) continuous recording of flow release discharge, river stage, temperature and turbidity; 2) monthly sampling of water chemistry parameters (between May and November); 3) assessment of periphyton accrual, and aquatic invertebrate abundance and diversity during fall; 4) a fall standing stock assessment to estimate the relative abundance and distribution of juvenile salmonids in the study area; and 5) flow ramp down monitoring and fish salvage. As in 2018-2020, activity 4) also included assessment of fish use of two off-channel habitats during the summer rearing period (June & July) and the fall stock assessment period (September).

Tasks 1) and 2) were conducted at eleven index sites in the LBR and two sites in the Yalakom River in Year 10 (2021). They included the seven historical index sites located at approximately three-kilometer intervals in the LBR downstream of Terzaghi Dam (i.e., river kilometer (Rkm) 39.9 (Site A), 36.5 (B), 33.3 (C), 30.4 (D), 26.4 (E), 23.6 (F), and 20.0 (G)), as well as 4 new sites in Reach 1 (established in 2019) at Rkm 11.3 (H), 7.5 (I), 3.9 (J), and 1.2 (K). Site A is located in Reach 4; sites B to E are in Reach 3; sites F and G are in Reach 2; and sites H to K are in Reach 1 (see Figure 1.5 and Appendix A). The two sites established in the Yalakom River in 2018 (Yal_A and Yal_B) were sampled again in 2019, 2020 and 2021. Inclusion of these sites provided measurements of periphyton accrual and benthic invertebrate diversity and abundance and insight into invertebrate recruitment to lower reaches of the LBR from this important tributary. In addition to the mainstem sites described above, water chemistry sampling was conducted in the following tributaries to characterize nutrient inputs from the various inflow sources: Mission Creek, Yankee Creek, Russell Springs, Hell Creek, Michelmoon Creek (in Reach 3), Yalakom River, Antoine Creek, Camoo Creek (in Reach 2), and Applesprings Creek, Moon Creek and Ama Creek (in Reach 1). The fall standing stock assessment was conducted at 36 sites distributed among reaches 2 and 3 during the Pre-flow period (Trial 0); at 49-50 sites across reaches 2, 3 and 4 during flow trials 1, 2 and the High Flow years; and at 61 sites across reaches 1 to 4 during the Post-High Flow years (2019-2021).

Sample collection periods during each flow trial for the water chemistry, periphyton, and benthic invertebrate monitoring components are summarized in Table 2.1. There was a shift in the number of seasons sampled mid way through the flow experiment. Samples were collected during spring (April to June), summer (July to September), and fall (September to December) during the Pre-flow (Trial 0) years and the first half of the Trial 1 period (up to 2005). Starting in the second half of Trial 1 (i.e., 2006) and continuing through Trial 2 and the six Modified Operations years to-date (i.e., high flow years from 2016-2018; and post-high flow years from 2019-2021), samples were collected only in the fall to standardize sampling to a single time of year among all trials.

Table 2.1 Water chemistry, periphyton and benthic invertebrate sample collection by flow trial and season for the Lower Bridge River.

Trial		Years	Reaches	Seasons when samples were collected	Target mean annual flow release from Terzaghi Dam ($m^3 \cdot s^{-1}$ ± %)	Actual mean annual flow release from Terzaghi Dam ($m^3 \cdot s^{-1}$ ± SD)
Trial 0		1996 – July 2000	2, 3	Spring Summer Fall	0	0.5 ± 1.1
Trial 1		August 2000 – 2005	2, 3, 4	Spring Summer Fall	3 ± 5%	3.0 ± 0.3
		2006 – 2010	2, 3, 4	Fall		
Trial 2		2011 – 2015	2, 3, 4	Fall	6 ± 5%	6.2 ± 0.4
Modified Operations [a]	**High Flows**	2016 – 2018	2, 3, 4	Fall	No target [a]	19.5 ± 2.1
		2018	Yalakom		n/a	n/a
	Post-High Flows	2019 – 2021	1,2,3,4	Spring [b] Summer [b] Fall	6 ± 5%	6.8 ± 1.6
			Yalakom		n/a	n/a

[a] Modified operations were a variance from Trial 2 resulting from reduction of water storage in Downton Reservoir and issues limiting diversion of flow above Terzaghi Dam to the generating stations at Shalalth. Flow excursions above the Trial 2 hydrograph (in terms of magnitude and duration) depend on snowpack and inflows during each mod. ops. year.

[b] Spring and summer sampling in 2020 and 2021 applied to water chemistry samples only. As in previous years since 2006, periphyton and benthic invertebrate samples were collected only in fall.

Following on recommendations included in the Year 8 (2019) report, sampling of water chemistry parameters during multiple seasons was resumed in 2020 and 2021 to document whether the declining trend in some nutrients (e.g., soluble reactive phosphorus) noted in the annual fall samples were also apparent during the other seasons. However, periphyton and benthic invertebrate sampling continued in the fall only.

Field data collection in Year 10 (2021) was conducted by staff from Coldstream Nature-Based Solutions Inc. (CNBS; formerly Coldstream Ecology Ltd.), Xwísten and St'at'imc Eco-Resources. CNBS also managed the collection of data, reporting and analysis for most of the Trial 2 years (i.e., 2012 to 2015), and the first high flow year in 2016 (McHugh and Soverel 2013, 2014, 2015, 2017; McHugh et al. 2017).

2.2. Physical and Chemical Habitat Parameters

2.2.1. Discharge and Site-specific Flow

Discharge rates were either provided or estimated according to location in the study area. Flows in Reach 4 (after initiation of the flow release) were comprised entirely of dam discharge since tributary inputs to this reach are very minor and ephemeral. As such the discharge data for this reach were based on the flow release values alone, which were provided by BC Hydro Power Records (as hourly values). Flows at each index site in reaches 3, 2 and 1 were estimated using a plug-flow approach (described by equations 1 and 2, below) based on tributary drainage area coupled with known Yalakom River discharge data provided by Water Survey of Canada (Gauge 08ME025). The names of the tributaries to the LBR study area are provided in Section 0 and the locations are shown on Figure 1.5. Mean daily, site-specific discharge estimates were calculated for each index site according to the following formulas:

$$(1) \qquad Q_{trib,d} = \left(\frac{Q_{yal,d} \times A_{trib}}{A_{yal}}\right)$$

where

$Q_{trib,\,d}$ = discharge of Lower Bridge River tributary on day (*d*) – see list of tributaries provided in Section 0 and locations shown on Figure 1.5;
$Q_{yal,\,d}$ = discharge of the Yalakom River on day (*d*) -- data provided by Water Survey of Canada;
A_{trib} = drainage area of Lower Bridge River tributary (estimated from a 1:50,000 topographic map); and,
A_{yal} = drainage area of the Yalakom River above the WSC gauge (estimated from a 1:50,000 topographic map).

$$(2) \qquad Q_{i,d} = Q_{i-1,d} + \sum(Q_{trib1,d}, Q_{trib2,d}, \dots)$$

where

$Q_{i,\,d}$ = discharge at Lower Bridge River mainstem index site (*i*) on day (*d*);

$Q_{i-1,\,d}$ = discharge at the next upstream index site (*i-1*) on day (*d*); and,
$Q_{trib,\,d}$ = discharge of Lower Bridge River tributaries between index site (*i-1*) and index site (*i*) on day (*d*) as calculated by equation (1).

So the daily discharges at the index sites in reaches 3, 2 and 1 (Equation 2) were estimated as the discharge at the next upstream index site plus the sum of the discharge estimates for the tributaries between each index site (Equation 1) as follows:

Site A (km 39.9) = Terzaghi release discharge;
Site B (km 36.5) = Site A discharge + km 36.8 groundwater inflow estimate;
Site C (km 33.3) = Site B discharge + Mission Creek inflow;
Site D (km 30.4) = Site C discharge + Yankee Creek & Russell Springs inflow;
Site E (km 26.4) = Site D discharge + Hell Creek & Michelmoon Creek inflow;
Site F (km 23.6) = Site E discharge + Yalakom River inflow;
Site G (km 20.0) = Site F discharge + Antoine Creek inflow;
Site H (km 11.3) = Site G discharge + Camoo Creek & Applesprings Creek inflow;
Site I (km 7.5) = Site H discharge;
Site J (km 3.9) = Site I discharge + Moon Creek inflow;
Site K (km 1.2) = Site J discharge + Ama Creek inflow.

2.2.2. Wetted habitat area

Surveys of hydraulic conditions were conducted during water releases at the dam ranging from 0 $m^3 \cdot s^{-1}$ to 15 $m^3 \cdot s^{-1}$ among the flow trials. Wetted width and length were measured with a laser distance meter (Leica Geosystems, Model Disto X4) and a laser rangefinder (Bushnell Corporation, Model Legend 1200 ARC), respectively, for each habitat type (cascades, runs, riffles, pools, rapids, and side channels). Wetted widths were measured at 10-20 m intervals within each habitat unit from one wetted edge across the channel (perpendicular to the stream axis) to the wetted edge on the other side. A minimum of 2 width measurements, and a maximum of 15 width measurements were recorded for each unit, according to unit length. Habitat unit lengths were measured along the stream axis (in mid-channel) from the upstream to downstream extents of each unit by aiming the rangefinder at a target (white, reflective board held by a crew member) and recording the distance. One length measurement was sufficient for shorter habitat units, and multiple measurements were necessary for longer units or around bends in the channel.

Water depths and velocities (at 0.6 of depth) were measured using a top-set wading rod and Swoffer Instruments, Inc. Model 2100 velocity meter at a minimum of 2 and a maximum of 18 locations along the thalweg in each habitat unit, according to unit length. Wetted area for a whole reach was calculated as the sum of wetted areas among habitat units within a reach wherein wetted area was unit length multiplied by mean wetted width within that unit. A model was fitted to the data and used to show change in wetted area as a function of the mean site-specific flows for each reach. Mean site-specific flow for a reach was the arithmetic mean flow calculated among sampling sites within a given reach.

2.2.3. Sediment Particle Size Distribution

Sediment particle size distribution was measured in 2019 and 2021 by conducting pebble counts to measure the composition of the stone matrix at each of the periphyton and benthic invertebrate sampling sites (Figure 1.5; Sections 2.3 and 2.4). A 100-particle Wolman Pebble Count (Wolman 1954) was completed at each site, where the intermediate diameter of 100 randomly selected particles was measured using calipers or a ruler. A crew member entered the stream. Without looking, he or she picked up a stone and measured its intermediate diameter, moved two or three steps along a transect across the channel and repeated the measurement on another stone. This process was repeated until 100 stones were measured. Measurements were done at each index site (A to K in the LBR mainstem and Yal_A and Yal_B in the Yalakom River). Measurements at sites H and I were not completed in 2019 due to unsafe access conditions to those locations on the day of sampling that year. From sizing of the 100 stones, the 50th percentile (called D_{50}) was calculated as the median particle size among the 100 measurements (Eaton et al. 2019). Sediment particle size measurements were not completed in 2020 due to budget limitations.

2.2.4. River Stage

The relative stage of the river has been monitored at seven stations: Terzaghi Plunge Pool (km 40.9), Reach 3/4 Boundary (km 36.8), Bottom of Reach 3 (km 26.1), Top of Reach 2 (km 23.6), Bottom of Reach 2 (km 20.0), Top of Reach 1 (km 11.3), and Bottom of Reach 1 (km 3.9). BC Hydro maintained the river stage monitoring equipment at Rkm 36.8, which is considered the compliance point for measurement of stage changes associated with flow ramp down events. River stage data for this site was recorded every 5 minutes throughout the year and was provided by BC Hydro Generation Operations. The stage data for the remaining sites were recorded every half-hour by water level data loggers manufactured by Onset Computer Corporation (Model: U20-001-01), which were maintained by the BRGMON-1 field crews. The Onset loggers were deployed from 10 March to 1 December 2021 in reaches 2, 3 and 4, and from 30 March to 9 December 2021 in Reach 1. During the deployment period, these loggers were retrieved and downloaded on 26-28 August and 18-20 October 2021.

Note: in Year 10 (2021), the Onset loggers deployed at kms 40.9, 26.1, 23.6, 20.0, 11.3 and 1.2 recorded absolute pressure (psi) but not river stage levels at those locations. As such, river stage data were not available for those locations in 2021 (until or unless an accurate conversion from pressure readings to stage levels can be determined, which was not completed at the time of writing for this report). As such, the following only applies to years prior to 2021 when river stage level data were available: Since the depths where the loggers were placed varied at each site and among deployments, the resulting stage values needed to be standardized so that relative changes across the year could be compared among the monitored locations. The stage data for each site were calibrated to 0.5 m at low flows (i.e., on 1 November each year) using the following formula:

$$Stage(cal)_{i,d} = Stage(raw)_{i,d} + \left(0.5 - Stage(raw)_{i,Nov1}\right)$$

where $Stage(cal)_{i,d}$ is the calibrated stage value at site (*i*) on day (*d*), $Stage(raw)_{i,d}$ is the original stage value at site (*i*) on day (*d*), and $Stage(raw)_{i,Nov1}$ is the original stage value at site (*i*) on 1 November. This approach ensured that the absolute differences in stage value remained unchanged, but they were all anchored to the same y-axis base point at low flows.

2.2.5. Water Temperature

Water temperature was recorded hourly throughout 2021 using Onset Model UTBI-001 data loggers (Onset Computer Corporation (Cape Cod, Massachusetts)). The temperature loggers were deployed at each of the eleven index sites in the LBR (including the four new sites in Reach 1) and at one location in the Yalakom River, approx. 100 m upstream of its confluence with the Bridge River. The loggers were anchored to the river substrate so they remained continuously submerged and were checked and downloaded at ca. 3- to 4-month intervals to reduce the potential for data loss.

To evaluate the effects of flow releases on the timing of emergence of Chinook and coho salmon fry from spawning gravels we calculated accumulated thermal units (ATU), defined as the sum of daily temperatures above 0°C from the observed average date of peak spawning, using average surface water temperatures for each monitoring station. Median emergence was assumed to occur at 1000 ATU for Chinook salmon (Groves et al. 2008) and 500 ATU for coho salmon (Murray et al. 1990; based on development data for 2-5°C water). Peak of spawning was set at September 8 and November 15 for Chinook and coho salmon, respectively, based on LBR run-timing distributions from streamwalk survey data collected under BRGMON-3 (White et al. 2021).

2.2.6. Turbidity

Turbidity was continuously monitored from mid-March to late fall in 2021 at each of the following locations (in upstream to downstream order): 1) Rkm 40.9 (Terzaghi Plunge Pool); 2) Rkm 36.8 (Reach 3/4 Boundary); 3) Rkm 26.1 (downstream end of Reach 3); 4) Yalakom River (near confluence with the Bridge River; Site Yal_B); 5) Rkm 20.0 (downstream end of Reach 2; Site G); and 6) Rkm 1.2 (downstream end of Reach 1; Site K). We used RBRsolo Tu loggers equipped with a Seapoint sensor and sensor wiper (RBR Ltd. Ottawa, ON). Due to variation in timing of logger installation, the logging period was March 3 to November 26 at Rkm 40.9, 26.1, and 20.0; April 30 to November 26 at Yalakom; August 13 to November 26 at Rkm 1.2; and August 14 to November 26 at 36.8. The logging interval was 10 minutes, and the sensors were wiped once an hour. Each logger and wiper assembly was installed inside an ABS pipe housing that was bolted to a large boulder or other anchor point (Photo 2.1). A cut-out on the pipe exposed the sensor to flowing water and allowed free movement of the wiper arm. To avoid sunlight affecting saturation of the photodiode amplifier on the sensor that can produce negative readings (J. Mather, Seapoint Sensors Inc. Pers Comm. May 11, 2020), the sensor was shielded using a plastic cover fixed in position above the pipe cut-out as shown in Photo 2.1 (the green cover). Despite the sensor shading, some negative values remained. These occurrences were deleted from the data frame.

Photo 2.1 Image of the turbidity logger housing installed at Rkm 36.8. The green shield was installed over the sensor to limit exposure to sunlight that was found to produce faulty readings in previous years.

Numerous high values were found, which represented voltage spikes potentially caused by artifacts on the sensor or other interference not associated with the general time course change in turbidity. These anomalies were removed by calculating median daily values where we assumed that daily resolution of turbidity was acceptable for descriptions and for use of turbidity in any statistical modeling. One-day anomalies not part of a multiday trend were considered spurious artifacts and were excluded.

2.2.7. Water Chemistry

Water samples for analysis of nutrient concentrations were collected monthly between May and November 2021 using standard grab methods. Sample locations were each of the mainstem index sites (A to K) as well as tributaries to each reach (shown as red pins on Figure 1.5). Sampling dates in 2021 were: 11 May, 8 June; 20-21 July; 26-28 August; 27 September; 25-26 October; and 29-30 November. Sampling in May, June, July and August showed water chemistry during spring and summer at each station. Samples from September, October and November showed water chemistry during the periphyton and benthic invertebrate sampling time series in fall.

All samples were kept cool and shipped to ALS Environmental in Burnaby, B.C. for analysis of ammonium (NH_4-N), nitrate (NO_3-N), soluble reactive phosphorus (SRP), total dissolved phosphorus (TDP), total phosphorus (TP), total alkalinity, total dissolved solids (TDS) concentration, conductivity, and pH using standard methods (APHA 2011). The sum of NH_4-N and NO_3-N was called dissolved inorganic N (DIN) that can be taken up by biota and used in photosynthetic production.

The molar ratio of DIN:SRP showed potential change in N and P limitation of benthic algal growth (Bothwell 1989, Perrin and Richardson 1997, Biggs 2000, Nelson et al. 2013). Rhee (1978) showed that for a given species of algae there is a sharp transition between P-limited and N-limited growth. The particular N:P ratio at which the transition between N and P-limitation occurs is species dependent, varying from as low as 7:1 for some diatoms (Rhee and Gotham 1980) to as high as 45:1 for some blue-greens (Healey 1985). Below a molar N:P of 20, the growth of most algal species will be limited by N whereas P-deficient growth is prevalent at molar N:P ratios greater than 50 (Guildford and Hecky, 2000). Because an optimum N:P ratio (above which P limitation occurs and below which N limitation occurs) can vary widely among freshwater algae, the range between 20 and 50 was regarded as a transition range in a community where the growth of some species will be P-limited and the growth of others will be N-limited.

2.3. Periphyton Biomass and Composition

Periphyton was sampled during the fall (September – November) in 2021 from riffle or run habitats at each of stations A to K in the LBR and at both Yalakom River stations (see Figure 1.5 and Appendix A for locations). The two stations in the Yalakom (Yal_A and Yal_B) and the four stations in Reach 1 (Sites H – K) were added in 2018 and 2019, respectively. The sampling locations in reaches 2 to 4 were the same during the earlier flow trials as described by Bradford and Higgins (2001) and Decker et al. (2008). Periphyton sampling methods in 2021 were the same as those used in all previous years of BRGMON-1, summarized as follows.

Artificial substrata called “periphyton plates” were used to sample periphyton assemblages (Photo 2.2). Each plate was a 30 x 30 x 0.64 cm sheet of open-cell Styrofoam (Floracraft Corp., Pomona CA) attached to a plywood plate that was bolted to a concrete block. Styrofoam is a good substratum because its rough texture allows for rapid seeding by algal cells, and the adhered biomass is easily sampled (Perrin et al. 1987). Use of the plates standardized the substrate at all stations and removed variation in biomass accrual due to differences in substrata roughness, shape, and aspect.

Photo 2.2 Image of an installed periphyton plate.

Periphyton biomass was sampled weekly from each of three replicate plates at each location for 57 days between 22 September and 24 November 2021, which was generally the same time of incubation used in previous years. Each biomass sample was a 2 cm diameter core of the Styrofoam and adhered biomass that was removed as a punch from a random location on each plate using the open end of a 7-dram plastic vial. The samples were kept frozen from the end of each sampling day until they were analyzed at ALS Environmental (Burnaby, B.C.) for chlorophyll-a concentration using fluorometric methods reported by Holm-Hansen et al (1965) and Nusch (1980). Units were µg chlorophyll-a·cm^{-2}. Chlorophyll-a is a plant pigment commonly used as a measure of biomass of photosynthetic algae (e.g., Stevenson 1996). This measure is preferred over something like dry weight that can include non-biological material in the stream substratum or non-photosynthetic organic matter (e.g., bacteria, fungi, detritus of terrestrial origin). The highest chlorophyll-a concentration accruing on each plate during the incubation period was called peak biomass (PB), which is related to cellular growth rate (Bothwell 1989) and was used as a standard metric of periphyton production.

At the time of biomass sampling, depth and velocity was measured at each plate using a top-set wading rod and Swoffer Instruments velocity meter. On the final periphyton sampling day (November 24, 2021), an additional core was removed from each plate and preserved in Lugol's solution, for later cell counts and measurement of biovolume by taxon. In the laboratory, biomass was removed from the Styrofoam punch using a fine spray from a dental cleaning instrument within the sample vial. Contents were washed into a graduated and cone shaped centrifuge tube and water was added to make up a known volume. The tube was capped and shaken to thoroughly mix the algal cells. An aliquot of known volume was transferred to a Utermohl chamber using a pipette and allowed to settle for a minimum of 24 hours. Cells were counted along transects examined first at 300X magnification to count large cells and then at 600X magnification to count small cells under an Olympus CK-40 inverted microscope equipped with phase contrast objectives. Only intact cells containing cytoplasm were counted. A minimum of 100 cells of the most abundant species and a minimum of 300 cells were counted per sample. The biovolume of each taxon was determined as the cell count multiplied by the volume of a geometric shape corresponding most closely with the size and shape of the algal taxon. Data were expressed as number of cells and biovolume per unit area of the Styrofoam punch corrected for the proportion of total sample volume that was examined in the Utermohl chamber.

2.4. Benthic Invertebrate Abundance and Composition

Three replicate benthic invertebrate samples were collected from the same 11 sites in the LBR and two sites in the Yalakom River that were used for the periphyton sampling (Section 2.3). Each invertebrate sample was collected from 25–50 mm size gravel enclosed in a wire basket measuring 30 cm long x 14 cm wide x 14 cm deep (Photo 2.3), with 2 cm openings that was installed in the river for 61 days (between 22 September and 24 November 2021). The basket was similar to that shown by Merritt et al. (2008). The baskets were filled with clean material that was collected from the stream bed or bank and closed using cable ties.

Photo 2.3 Image of a basket sampler before installation in the Lower Bridge River. Baskets used in 2020 and 2021 were longer than those from previous years. To ensure consistency of data among years, a baffle was installed in each basket to contain the same volume of packed stones as in previous years.

At the start of each colonization period, the baskets with contained stones were placed among the natural river substrata in riffle or run habitat. The baskets remained undisturbed for the duration of the ca. eight-week colonization period. Water depth and velocity was measured weekly during the incubation period at the upstream end of each benthic invertebrate sampling basket using a top-set wading rod equipped with a velocity sensor manufactured by Swoffer Instruments.

Following incubation, the baskets were carefully removed from the streambed and placed into individual buckets. The basket was opened by clipping the cable ties, and invertebrates were brushed from the gravel using nylon brushes. Removed material was filtered through a 250 µm Nitex screen (to remove excess water), transferred to a sample jar, and preserved in 10% formalin. Following sample collection, the preserved invertebrates were submitted to Stamford Environmental for sorting, identification to Family, and enumeration.

In the laboratory, formalin was removed from the samples by washing with water through a 250 µm filter then neutralized with FORMEX (sodium metabisulfite) before discarding. Animals were picked from twigs, grasses, clumps of algae, and other large organic debris. These animals and remaining material were washed through 2 mm and 250 µm mesh sieves to separate the macro portion (>2 mm) from the micro portion (<2 mm and >250 µm). Animals in the macro portion were picked and placed in vials containing 70% ethanol. The micro portions were subsampled according to the following four steps:

1) Suspended specimens and substrate were decanted from the micro portions in preparation for subsampling. The remaining sandy heavy portion was then examined under a microscope and all specimens (e.g., stone-cased caddis fly larvae) were picked out and added to the decanted volume.
2) Suspended micro portions were each homogenized by stirring then subsampled using a four-chambered Folsom-type plankton splitter: an apparatus designed to collect random proportions from volumes of suspended invertebrates. Approximately 300 specimens (minimum 200) were used for guiding subsample sizes. Simulations suggest random subsamples containing >200 specimens encompass the diversity present in a sample and provide accurate estimates of abundance (Vinson and Hawkins 1996; Barbour and Gerritsen 1996; Walsh 1997; King and Richardson 2001). Micro portions were split into half portions repeatedly until the resultant splits contained about 300 specimens.
3) A random selection of three samples (10%) were sorted twice to ensure picking efficiency was consistently maintained at 95%.
4) Counts from the micro portions were multiplied by the inverse of the split proportion to obtain estimates of abundance in the micro portions. These values were added to the direct counts from the macro portion to obtain the estimated abundance in the whole sample. All animals were preserved and retained in 70% ethanol.

For taxonomic identification and enumeration, the animals were identified to family except *Acari* (mites), *Clitellata/Oligochaeta* (earthworms), *Nematomorpha* (horsehair worms), *Platyhelminthes/Turbellaria* (flatworms), and *Ostracoda* (ostracods). Enumeration at the family level was based on findings by Reynoldson et al. (2001), Bailey et al. (2001), Arscott et al. (2006), and Chessman et al. (2007) that family assemblage data are equally sensitive to lower taxonomic levels for evaluating invertebrate response to change in habitat condition in resource management applications. Higher level taxonomy (e.g., class, order) was applied for non-insect aquatic invertebrates and terrestrial taxa. Taxonomy was based on keys in Merritt and Cummins (1996) and Thorpe and Coviche (2001).

Benthic invertebrates were quantified as counts of animals in aquatic life stages (mainly larval forms of aquatic insects). Individuals from terrestrial habitats or adults of aquatic insects were not included in the animal counts. All invertebrate data were expressed as number of individuals per basket sampler or per unit area where the planar areal dimension of the basket lying on the stream substratum was the area of sample. Biomass of the benthic invertebrates was not measured.

2.5. Juvenile Fish Production: Size, Abundance, Biomass and Habitat Use

For fish sampling, the focus of the program has been on the juvenile lifestage (i.e., fry and parr) of Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*) and steelhead (*O. mykiss*), because it was expected that instream flows and associated freshwater productivity could have a measurable influence on the recruitment and survival of these species. It is understood that both resident rainbow trout and anadromous steelhead reside in the Lower Bridge River. Based on the results of otolith microchemistry analysis in 2015, a higher proportion of the recruited juveniles are steelhead (King and Clarke 2015); however, potential changes in the relative proportions were not routinely assessed across each of the flow trials. As such, juvenile steelhead/rainbow trout are referred to collectively as “mykiss” in the text and represented by the abbreviation “RB” in tables and figures throughout this report.

Juvenile Fish Size

Monthly growth sampling was discontinued in 2018 – refer to the Year 7 report for the rationale (Sneep et al. 2019). As a result, the analysis of fish size among flow treatments included in this report is based solely on fish sampled during the annual stock assessment (see description of this method under “Abundance and Biomass”, below). During this task, a sufficient sample size ($n \geq 30$) of each target species and age class has been more consistently acquired for each reach, and the sample timing has been very consistent across years. Forklength (mm) and weight (g) has been recorded for captured fish since 1996.

Abundance and Biomass

The abundance and biomass contributions of juvenile salmonids were estimated by conducting an annual closed-site, depletion-type electrofishing survey. For 1996 to 1998, sampling was conducted between late September and mid October, but for the remainder of the experiment, sampling generally occurred between early and late September (Table 2.2). The selection of sampling sites for the flow trials was based on habitat surveys that were conducted in reaches 2 and 3 in 1993, and in Reach 4 in 2000 (after initiation of the flow release re-wetted that reach) that inventoried all major meso-habitat types (e.g., runs, riffles, pools).

At the start of the program, 18 sampling units were randomly selected for each of reaches 2 and 3 from the inventory of habitat units in proportion to their occurrence in the inventory. Although the original intent was to use these sites throughout the entire flow experiment, some sites had to be relocated slightly owing to changes in the channel morphology resulting from debris flows and spills from the dam. New sites were chosen to have the same characteristics as the altered sites to maintain the same distribution of habitat types being sampled. Two additional sites were added to the upper region of Reach 3 in 1998, bringing the total number of sites for Reach 3 to 20. In 2000, an additional 12 sites were selected in the rewetted Reach 4 by the same procedure that was used for reaches 2 and 3, bringing the total number of sites for all three study reaches to 50. Starting in Trial 2, 1 – 2 sites in Reach 2 were dropped, reducing the number for that reach to 16 or 17 and the total for reaches 2 – 4 to 48 or 49 since 2012. Starting in 2019, 12 new sites were established in Reach 1 bringing the total number of sites to 61 and extending the spatial scope of monitoring to the confluence with the Fraser River (Figure 2.1 and Appendix A).

Table 2.2 Years of juvenile stock assessment sampling used to compute average abundance and biomass for each flow regime in the Lower Bridge River for Chinook, coho, and mykiss fry (Age-0+) and mykiss parr (Age-1).

Year	Flow Treatment (MAD)		Sampled Reaches	Total # of Sites	Sampling Dates
1996	Trial 0 – Pre-Flow ($0\ m^3 \cdot s^{-1}$)		2-3	36	8 – 16 Oct
1997				36	2 – 13 Oct
1998				38	29 Sep – 9 Oct
1999				38	3 – 10 Sep
2000	Transition Year [a]		2-3	50	30 Aug – 10 Sep
2001	Trial 1 ($3\ m^3 \cdot s^{-1}$)		2-4	50	27 Aug – 10 Sep
2002				50	28 Aug – 5 Sep
2003				50	2 – 11 Sep
2004				50	7 – 15 Sep
2005				50	6 – 16 Sep
2006				50	5 – 14 Sep
2007				50	5 – 19 Sep
2008				50	3 – 18 Sep
2009				49	8 – 24 Sep
2010				50	7 Sep – 19 Oct [b]
2011	Trial 2 ($6\ m^3 \cdot s^{-1}$)		2-4	50	6 – 22 Sep
2012				45	5 – 27 Sep
2013				47	4 – 26 Sep
2014				48	2 – 24 Sep
2015				48	1 – 28 Sep
2016	Modified Operations	High Flows ($>18\ m^3 \cdot s^{-1}$)	2-4	48	1 – 21 Sep
2017				49	5 – 20 Sep
2018				49	4 – 15 Sep
2019		Post-High flow ($6\text{-}9\ m^3 \cdot s^{-1}$)	1-4	61	29 Aug – 18 Sep
2020				61	25 Aug – 11 Sep
2021				61	30 Aug – 20 Sep

[a] The year 2000 was considered a transition year because the flow release started on 1 Aug that year, only one month before the annual stock assessment timing. As such, this year was not included in any trial averages.

[b] In 2010, 4 sites were completed in mid-October (3 in Reach 2; 1 in Reach 4); The other 46 sites were completed by 19 September.

At each site, the area to be sampled was enclosed with block nets constructed of 6 mm mesh. The average size of a sampled area in 2021 was 72 m^2 (range: 20 to 118 m^2 among sites, based on the amount of suitable habitat at each location). Total catches were derived using a depletion method based on three or four passes of backpack electrofishing. A minimum of 30 minutes elapsed between passes. After each pass, captured fish were identified and forklength (nearest mm) and weight (0.1 g) of all salmonids were recorded before being released outside the enclosure. Ages (i.e., Age-0+, Age-1, etc.) were assigned to all captured fish according to identifiable size ranges based on analysis of length-frequency histograms for each reach.

During the Pre-flow period, nets were used to block off the full width of the stream in Reach 3; therefore, the sampled areas included the entire channel. This was not possible in reaches 1 and 2 during any monitoring year, or in reaches 3 and 4 after the flow release because of the greater depths and velocities associated with the increased flows. In these cases, sampling was conducted in three-sided enclosures along shore instead. These enclosures averaged 5.1 m in width. Flows from the dam during the depletion sampling period in September were the same (i.e., 3 $m^3 \cdot s^{-1}$) for all of the flow release monitoring years (2000 to 2021; see September period on Figure 1.3).

Figure 2.1 Distribution of Juvenile Stock Assessment Sites in the Lower Bridge River study area in 2021.

For the locations where three-sided sites were used, there was potential for some fish (e.g., parr) to be located further offshore and inaccessible to the gear. Therefore, the proportion of the population that was vulnerable to this sampling method was estimated using data that was collected as part of a separate Lower Bridge River microhabitat use study. In that study, divers located the position of juvenile salmonids during the day relative to the shoreline at two sites in Reach 2 and two sites in Reach 3 during August 1999, October 1999 and July 2000, prior to the flow release, and in August 2000 after the flow release.

For Reach 2, where the flow release from the dam had little impact on habitat conditions, observations from the August 1999 and August 2000 surveys were combined for estimating the distribution of fish from shore. The data collected in Reach 3 in late August 2000, approx. one month after the start of the flow release, was used to estimate the post-flow release distribution for reaches 3 and 4. The location of fish concealed in the substrate could not be determined by the daytime surveys, so the assumption was made that the distribution of fish observed during the microhabitat study would be a reasonable approximation of the location of all fish in the channel (either concealed in the substrate or swimming in the water column).

2.6. Adult Escapement

Adult spawner count data for the Lower Bridge River (up to 2020) were provided by Instream Fisheries Research (IFR) who are conducting the Lower Bridge River Adult Salmon and Steelhead Enumeration program (ref. BRGMON-3). As a part of their work, IFR have compiled and analyzed historical data to supplement their own data collection which began in 2012.

Visual counts for Chinook and coho were conducted annually by helicopter overflights or streamwalks during the flow experiment period (i.e., Pre-flow (Trial 0), Trial 1 and Trial 2 years), as well as the high flow years from 2016 to 2018, and the post-high flow years from 2019 to 2021. Counts by helicopter overflight were conducted in all reaches from 1997 to 2004 (missing 2000, 2002 and 2003 for Chinook Salmon, 2000 and 2002 for Coho Salmon). Visual streamside counts have been used since 2005 to enumerate both Chinook and Coho Salmon in Reach 3 and 4 (missing 2007 for Coho Salmon). Count data obtained from DFO was used to reconstruct AUC estimates for Chinook and coho salmon from the Yalakom confluence to Terzaghi Dam (Reach 3 and 4) since the start of monitoring. However, IFR has noted a caveat that the historical estimates are highly uncertain given changes in methods over time, the low number of surveys in some years, and the lack of measurements for observer efficiency and survey life prior to 2012 (White et al. 2021).

Visual surveys (streamwalks) conducted under the BRGMON-3 program (2012 to the present) followed methods used in previous years, where two observers walked in a downstream direction on the riverbank, counted spawners and recorded species and location. Viewing conditions, cloud cover, and secchi depth were also recorded (White et al. 2021). Visual counts occurred weekly for Chinook and coho salmon in Reaches 3 and 4. Starting in 2018, the streamwalks for spawner enumeration were expanded to include sections of Reach 2 and spot counts at the downstream end of Reach 1 (White et al. 2021). However, for consistency with BRGMON-1 reporting from previous years, only counts for reaches 3 and 4 were used in the stock-recruitment analyses (note: low numbers of Chinook (number unreported) and one coho salmon were observed during streamwalks in Reach 2 during 2019, but spawning behaviour was not confirmed; no spawners were noted in reaches 1 or 2 in 2020).

In 2020, the visual survey period for Chinook was from 23 July to 1 October, and the survey period for coho was from 1 October to 17 December. **However, as noted in the BRGMON-3 report, there was an important factor requiring a change in how the total spawner estimate was derived for Chinook in 2019 and 2020: A channel-spanning fish fence (for broodstock collection)**

was installed at the downstream end of Reach 3 on 20 August in 2019 and 10 August in 2020. As such, the visual-based estimate was combined with the numbers counted at the broodstock collection fence in order to get a complete estimate for those years. Another important factor for salmon runs in 2019 and 2020 was the Big Bar Slide on the Fraser River. The slide reportedly occurred in November 2018 and may have increased the proportion of stray spawners from other systems entering the Bridge River (refer to the Year 9 (2020) report for DNA-based stock identification results for adult Chinook captured at the broodstock collection fence; Sneep et al. 2021). Surveys for steelhead were deemed ineffective in past years due to high turbidity and flow volumes in the LBR during their migration and spawning period; thus, visual surveys have not been completed for steelhead.

Escapement estimates from these visual surveys were generated using area under the curve (AUC) estimation which relied on observer efficiencies and residence times determined by radio telemetry and visual surveys, including marked fish, which have been conducted since 2011 (White et al. 2021). However, as noted by the authors, generating accurate and precise AUC estimates from the historic data was hampered by inconsistent sampling methodology and survey area across flow treatments, and a lack of historic observer efficiency data. A key assumption in AUC estimates is that the mean observer efficiency documented by the BRGMON-3 program reflects conditions both before and after the flow release. It is likely that observer efficiency prior to the flow release was higher owing to lower and clearer flows. Thus, escapements prior to the flow release are likely overestimated due to this assumption.

A fish enumeration facility (resistivity counter) was constructed by IFR in October 2013 near the downstream end of Reach 3 to obtain more precise escapement estimates for coho, Chinook and steelhead above the Yalakom confluence going forward. Based on results in other systems, resistivity counters can provide accurate estimates (with confidence limits +/- 10% of true abundance). In future, these counter-based estimates can be compared to the estimates based on visual methods as a means of calibrating the historic estimates (though such a comparison would only apply to post-flow release counting conditions and would not address the bias described in the preceding paragraph). However, at the time of this report, the time series of data available from the resistivity counter were insufficient for incorporation in the stock-recruitment analyses.

For more detailed information on the collection of the adult salmon and steelhead escapement data and the associated analyses for generating the annual abundance estimates, refer to the IFR BRGMON-3 report (White et al. 2021).

2.7. Modified Operations Monitoring

Each of the sampling components described above have been a part of the long-term WUP monitoring to assess the effects of releases from Terzaghi Dam on the Lower Bridge River ecosystem across broad time scales and the range of flow treatments. However, increases in the maximum discharges from Terzaghi Dam above the trial flow levels were expected to have additional impacts as well. To address some of the identified effects in the immediate-term, additional high flow monitoring was incorporated to supplement the BRGMON-1 program. This work was implemented under three new high flow monitoring components for modified operations from Terzaghi Dam: 1) High flow monitoring of water quality parameters, kokanee entrainment, bank erosion and deposition, and potential fish stranding locations; 2) Surveys to document juvenile salmonid habitat use in enhanced off-channel habitats under high flows (mitigation effectiveness monitoring); and 3) High flow ramp down monitoring and stranding risk assessment (to supplement the annual ramp down monitoring within the WUP flow range).

2.7.1. High Flow Monitoring

Water Quality

The high flow period in 2021, defined as the period when flows exceeded the Trial 2 peak (15 $m^3 \cdot s^{-1}$), was between 5 May and 30 July 2021 (87 days). Water quality monitoring of modified operations in 2021 was conducted on 8 dates between 11 May and 30 August 2021. Monitoring prior to the start of high flows to document baseline conditions did not occur in 2021, and monitoring on dates in August 2021 was intended to document conditions following the return to Trial 2 flow levels. As in previous years with high flows (i.e., 2016-2018), data were collected at three selected locations in the LBR channel spread between the dam and the Yalakom River confluence (i.e., reaches 3 and 4). These locations were: the Terzaghi Dam plunge pool (river km 40.5), Russell Springs (Rkm 30.4), and just upstream of the Yalakom River confluence (Rkm 25.0). Results were also documented for the Yalakom River as reference data from this significant tributary. High flow monitoring included spot measurement of air temperatures, water temperatures, total dissolved gases (TDG), and turbidity (NTUs). Conductivity (µS/cm) and Alkalinity (mg/L as $CaCO_3$) were also determined from water samples collected on or near the same dates (and at or near the same locations) which were analyzed by ALS Environmental Labs (Burnaby, BC).

Spot measurements were recorded on 10 June, 6 and 27 July, and 30 August 2021, and samples for laboratory analysis were collected on 11 May, 8 June, 20 July, and 26 August 2021, which spanned the peak flow period and descending limb of the hydrograph during summer. Spot water temperature measurements were taken at >60 cm depth in the water column, and TDG and turbidity monitoring were conducted according to BC Hydro protocols, as specified in the Lower Bridge River High Flow Monitoring Scope of Services. To facilitate comparability of results, the same time of day was targeted on each sampling day for measurements. A Point Four Tracker portable total gas pressure meter was used to measure TDG and water temperature. Barometric pressure was measured on each sampling day for calibrating the instrument. Turbidity samples were collected by plunging a clean sample bottle (that had been pre-rinsed 3x with river water) below the surface until full. Turbidity (as Nephelometric Turbidity Units, or NTUs) of each sample was measured by a LaMotte 2020we portable turbidity meter.

Kokanee Entrainment

To assess the incidence of kokanee entrainment from Carpenter Reservoir into the Lower Bridge River channel during the period of high flows (>15 $m^3 \cdot s^{-1}$), visual streamwalks were conducted to observe and enumerate kokanee (live and mortalities). The surveys were conducted by two technicians from CNBS and Xwísten on 10 June, 6 and 27 July 2021. The survey area extended from the dam to approximately 1.5 km downstream on the river left side of the channel only (river right was not accessible due to the high flows). Total numbers of observed kokanee were recorded for each survey.

Fish condition (e.g., injured, uninjured, etc.), age, and maturity were qualitatively assessed or estimated for all observed kokanee. When it was possible to safely collect any mortalities, they were measured for fork length (mm) and weight (g), and scale samples were collected to support the Carpenter Reservoir Fish Habitat and Population Monitoring program (Ref. BRGMON-4). Other documented information included: GPS coordinates at fish observation locations, and the presence of eagles in the area.

The data from these surveys were intended to: a) document whether kokanee entrainment occurred during the 2021 high flow event; b) establish an index of the number observed on the survey dates; and c) record some observations about the fish that were found. Based on these data, it is not possible to estimate total numbers of entrained fish (since observer efficiency was not assessed—which would require a much larger and more complex monitoring approach), determine the proportion of entrained fish that were live or mortalities, or determine the specific cause of the observed mortalities.

Bank Erosion and Sediment Recruitment

Crews also conducted surveys to identify and assess bank erosion and sediment recruitment sites associated with the high flows. A set of fourteen bank erosion and deposition sites in reaches 2, 3 and 4 that had been identified by surveys conducted in 2017 and 2018 by Kerr Wood Leidal Associates Ltd. (KWL; Ellis et al. 2018) and CNBS (Sneep et al. 2019), respectively, were re-visited for repeat observations. In addition to the pre-selected sites, surveys also included documentation of any new erosion areas that were observed. It was not possible to monitor sediment *deposition* within the LBR channel due to the depths, velocities and turbidity under high flow conditions, as was the case in 2018 (O'Farrell and McHugh 2018). The identified locations were surveyed four times by vehicle and shore-based streamwalks, on 10 June, 9 and 27 July, and 30 August 2021.

For each identified erosion area, recorded parameters included: site name or location description, GPS coordinate, river bank, reach, whether the erosion was active (Y/N), dimensions (approx. length and width in meters; area in m^2), substrate composition (% by size class), and whether there were localized water quality or habitat effects observed. All sites, with the exception of the Fraser Lake rip rap site (approx. Rkm 33.2) were assessed using binoculars at distances of 30 m to 300 m. Photographs were taken at each location for reference (only a few are included in this report; the remainder can be provided upon request).

Fish Stranding Site Reconnaissance

Targeted field reconnaissance during the high flow (>15 $m^3{\cdot}s^{-1}$) period was also intended to identify potential new fish stranding sites in reaches 1, 2, 3, and 4 under the range of high flows in 2021, and for proactively guiding fish salvage efforts during the high flow rampdown events. Surveys were conducted on three dates in 2021: 10 June, 6 July and 27 July. Sites that were considered accessible (e.g., didn't require crossing the river at high flows) were subsequently revisited during ramp down to document how these locations dewatered and salvage fish, as required.

The following parameters were recorded for each potential stranding location that was flooded by the 2021 high flows: GPS location, reach, river bank, habitat type, site description, dimensions (approx. length and width in meters; area in m^2), potential flow release discharge ($m^3{\cdot}s^{-1}$) when site may dewater, fish observations, and any other factors deemed relevant for the subsequent fish salvage surveys (e.g., access). Photos were also taken and are available upon request.

2.7.2. Mitigation Effectiveness Monitoring

Following from pilot efforts during high flow releases in 2018, fish sampling was undertaken in a couple of enhanced off-channel sites in spring (i.e., June) and summer (i.e., July or August) to characterize fish use of these habitats during the peak period and descending limb of the LBR hydrograph in 2019-2021. There is interest in understanding the relative importance of these habitats for providing refuge when high discharges limit the suitability of fish habitat in the mainstem. Data collection in non-high flow years (i.e., 2019 and 2020) was intended to provide context and background information against which the high flow results can be compared. By repeat-sampling the same set of sites on two dates in the same year, the intent was to assess changes in use of those sites by juvenile salmonids at different flow release discharges and seasonal periods (i.e., spring vs. summer).

The mitigation effectiveness monitoring surveys in 2021 were conducted on 17 and 18 June, and 14 and 15 July. The surveys targeted peak flows (i.e., 24-25 $m^3{\cdot}s^{-1}$ on both June and July dates). Two sites were selected in the off-channel habitat at Bluenose (Bluenose Outflow Channel and Bluenose Pond & Upper Intake Channel) and four sites were in the off-channel habitat at Applesprings (Applesprings Outflow Channel, Applesprings Upper Sidechannel, Middle Sidechannel and Lower Sidechannel) (see Appendix A for locations).

Each site was sampled in an upstream direction and spanned the full width of the channel. Fish sampling was conducted by open-site electrofishing. As such, catch results for these surveys represented a minimum estimate of fish presence at the time of each survey. Juvenile (Age-0+

and Age-1) coho, Chinook, and *O. mykiss* were the target species and age classes. Site length and electrofishing effort varied depending on the amount of habitat available at the selected sites.

All fish collected during sampling were identified to species and age class (estimated), measured for length and weight, and a sub-set were photographed. Documentation at each site also included: electrofishing effort (seconds), crew member initials, GPS locations at the upper and lower extent of each site, the length of shoreline sampled, and the general characteristics of the site (habitat type, % composition of substrate classes, water visibility).

Enhanced Off-channel Sites – Juvenile Salmonid Stock Assessment

Enhanced side channel sites were also sampled for juvenile salmonids in fall of 2018-2021. During 2021, three sites in riffle habitat and one site in pool habitat were sampled at the Applesprings enhanced side channel located in Reach 1, and one site in riffle habitat and one site in pool habitat were sampled at the enhanced Bluenose side channel in Reach 4 (refer to Appendix A for maps and UTM coordinates) for a total of four sites at Applesprings and 2 sites at Bluenose. Fish sampling methods applied at these locations were the same as those used during the mainstem fall standing stock assessment (described in the Abundance and Biomass sub-section of Section 2.5, above.

Sample timing in 2021 was 21 and 23 September for the enhanced off-channel sites, which was just after the mainstem depletion sample timing. A multi-pass electrofishing depletion approach was used to estimate density, biomass and abundance. Estimates of density and biomass were compared to averages from mainstem sites. The area of each enhanced side channel site by habitat type was multiplied by the estimated densities and summed to determine the total abundance at each site. We compared the abundance from these enhanced side channels relative to the total abundance in the mainstem to determine the extent to which they potentially mitigate impacts of high flows. Note: the abundance estimates for the off-channel sites were not included in the total estimates for the mainstem reaches presented in this report in order to maintain consistency with the results and analysis from previous years and flow treatments.

2.7.3. Fish Salvage and Stranding Risk Assessment

Flow ramping and fish salvage data were collected as part of High Flow Ramp Down Monitoring at LBR discharges >15 $m^3{\cdot}s^{-1}$, and across the range of WUP Trial 2 flows (15 to 1.5 $m^3{\cdot}s^{-1}$) in Year 10 (2021). The methods described in this section are based on communications and documentation provided by the field staff at CNBS. 2021 discharge data for Terzaghi Dam and river stage data for Rkm 36.8 (~4 km downstream from the dam; a.k.a. the compliance location) were provided by BC Hydro Power Records. These data were available at hourly intervals and 5-minute intervals, respectively.

On each ramping date before any flow changes were initiated, field reconnaissance of the survey area was completed at an overview level to identify and rank specific locations with potential fish stranding risk or confirm stranding risk at sites identified during the high flow reconnaissance. Once the flow changes from the dam began, fish salvage crews were dispatched to the areas deemed to have the most immediate risk first, and then moved as the degree of risk shifted from location to location. Site and habitat information was recorded for each identified stranding location on each ramping day, which included: Date, flow release rate at the dam, approximate river kilometre (upstream of the confluence with the Fraser River), GPS coordinates, bank, area (in m^2 based on length and width measurements), habitat type, substrate composition, and weather.

For fish salvaging, backpack electrofishing (EF) and hand collection were the methods employed. Parameters recorded for the fish salvaging included: Sampling effort (EF seconds), number of passes, species and age class (i.e., fry or parr), and number salvaged. Forklengths (in mm) and weights (g) were measured for a subset of salvaged fish and DNA was collected from a sample of Chinook juveniles for stock identification. Fish salvage efforts focussed on fish that were already isolated, stranded or mortalities. As per the direction of BC Hydro's Scope of Services (BC Hydro 2020), fish in habitats that were not yet isolated or stranded (i.e., incidental catches) were not to be sampled. This was to ensure that salvage totals reflected the actual numbers of fish that were stranded from the main channel flow by the ramp down event.

Analyses of the flow ramp down and fish salvage results were based on the risk assessment approach outlined in BC Hydro's Lower Bridge River Adaptive Stranding Protocol to determine risk ratings for the identified stranding sites at each river stage change. Where possible, fish stranding data from 2021 were compared with the results from previous study years to better inform the risk of fish stranding across the flow ramp down range. As in previous high flow years (2016–2018), ramp down and fish salvage data from the high flow range (>15 $m^3 \cdot s^{-1}$) were included with the data from within the WUP Trial 2 range (≤15 $m^3 \cdot s^{-1}$) for the stranding risk assessment analyses.

2.8. Data Analysis

2.8.1. Trial, Reach, and Pink Effects on Benthic Communities

We tested Trial, Reach, and Pink salmon effects on several response variables using three-way analysis of variance (ANOVA). The response variables were algal PB and the count per sample or per m^2 of all invertebrates, Chironomidae, EPT (sum of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Tricoptera (caddisflies)), and family richness (number of invertebrate families in a sample). The chironomids and EPT represented most individuals among the invertebrate assemblages found in previous analyses (Sneep et al. 2021) and are sensitive indicators of habitat disturbance (Holt et al. 2015, Kennedy et al. 2016). Both the chironomid and EPT larvae are fish food organisms (Quinn 2018). "Trial" corresponded to each of the time blocks of flow plus the group of years 2019, 2020, and 2021 occurring after the High Flow years and collectively called "Post-High Flow Years". Final Trial assignments were called Trial 0, Trial 1, Trial 2, High Flow Years, and Post-High Flow Years. "Reach" blocked the response variables by physical reaches in the river (Reaches 1, 2, 3, 4) plus the Yalakom River (coded Reach 5 in data analysis). "Pink" was a surrogate for nutrient pulsing caused by salmon spawning and carcass decomposition (e.g., Harding and Reynolds 2014, Albers and Petticrew 2012). In the Bridge River, Pink salmon spawn in odd years and are absent in even years. This bi-annual sequence may produce greater benthic production in odd years (the on years for Pink spawning) and lower production in even years according to nutrient releases from the decomposing salmon. The overall layout allowed for testing effects of Trial, Reach, and Pinks on each of the response variables using the compiled data of response variables dating back to 1996 when benthos monitoring started in the Bridge River.

Each observation was the mean value of a response variable among all samplers and stations, by reach, in a given year. Years were replicates. This approach overcame pseudo-replication within a station and stations within a reach. Normality was tested using the Shapiro-Wilks test. If sample distributions were non-normal, the data were $\log_{10}$ transformed to achieve normality. The significant probability level was set at 0.05 (an effect was considered present if the probability value was <0.05). If a significant interaction between any combination of Trial, Reach, and Pink was found, graphs were used to show the interactions. If no significant interaction was found for a given factor with another, that given factor (Trial or Reach or Pink) was examined independently. If any one of the factors was found to have a significant effect on a response variable ($p<0.05$), the Tukey test was run to determine what level or levels of that factor differed significantly from the others. Significant probability was $p<0.05$.

Periphyton and benthic invertebrates were not measured in Reach 4 during Trial 0 due to no flow release, which meant that either Trial 0 or Reach 4 had to be omitted to allow complete testing of Trial, Reach, and Pink effects on the endpoint variables. We elected to omit Trial 0 on the premise that some flow release rather than no flow release was more important for management decisions than omitting Reach 4 because it is unlikely that a future flow scenario would consider no flow release from the dam. In Reach 1, periphyton and benthic invertebrates were only measured in the post-high flow years (2019, 2020, and 2021). Without observations from Trials 1 to 3, Reach 1 could not be included in the ANOVA's. Based on these criteria, Trial had 4 levels (Trials 1 to 4), Reach had 3 levels (Reaches 2 to 4), and Pink had two levels (0 for off years and 1 for on years).

2.8.2. Testing a Benthos Response to Nutrient Addition

If a Pink effect was found among any of the benthos ANOVA's, a second analysis was run to examine effects of nutrient enrichment on total density of benthic invertebrates. A Before-After-Control-Impact design (Stewart-Oaten et al. 1986, Johnston et al 1990, Rosario and Resh 2000, Smokorowski and Randall 2017) was applied using existing data describing total benthos density in years before and during nutrient addition caused by Pink spawners (the treatment) in a control and treatment reach. The control reach was where few or no Pink salmon spawners were present at any time based on low NH_4-N and SRP concentrations that were sensitive to and indicated the presence of spawning Pink salmon. The treatment reach was where high concentrations of NH_4-N and SRP from Pink spawners in odd years were found. Each "Before" year was an even year preceding an odd year when anomalous nutrient concentrations showed presence of Pink spawners in the treatment reach. The "After" years were those having anomalous and high NH_4-N and SRP concentrations showing the presence of Pink spawners in the treatment reach. A one-way ANOVA was run to determine if the mean difference of benthos densities between the control and treatment reach in the "Before" years was different from the mean difference in the "After" years. Years were replicates in this layout. If the ANOVA showed a statistically significant difference, a conclusion was that nutrient addition from Pink spawners affected total benthos densities. Means and standard errors were displayed graphically to show the Before-After-Control-Impact contrasts. Normality was tested using the Shapiro-Wilk test. If significant, the data were $\log_{10}$ transformed to improve normality and homogeneity of variance.

2.8.3. Juvenile Fish Production: Size and Condition

We evaluated effects of flow on juvenile salmonid growth based on weight samples taken during the annual fall stock assessment. As in the analysis for periphyton and benthic communities described above, 2019-2021 were considered a new modified operations flow treatment which we refer to as the "Post-High Flow years". The results as presented for the previous flow treatments did not change. Mean weight is also the metric used to transform abundance estimates to biomass estimates for each flow treatment, so understanding changes in mean weights in addition to changes in abundance is important for explaining why abundance and biomass trends may differ.

Using weight as a surrogate for growth assumes that the interval between emergence date and sampling date are relatively consistent among years, or at least among flow treatments. There was some variation in sampling dates for stock assessment among years, particularly between the first three years of the Pre-flow period (early to mid October from 1996 to 1998) and the subsequent flow treatments (late August to late September from 1999 to 2019; see Table 2.2, above). Generally, the variation within the flow trial years was low. Owing to changes in water temperatures due to differences in flow treatments, emergence timing was likely different, especially for Chinook where water temperature differences over the incubation period between the pre-treatment and later flow treatments have been large (Section 3.1.5). Thus, using weight data to make inferences about growth is problematic, especially for Chinook. Nevertheless, we computed average weight (and standard deviation) by reach and flow treatment for Age-0+ mykiss, coho, and Chinook, and also for Age-1 mykiss.

We did not use formal tests to determine whether average weights in a particular reach were statistically different among flow treatments for two reasons. First, this would involve a large number of comparisons. There are 10 potential flow treatment comparisons (Pre-flow to Trial 1, Pre-flow to Trial 2, Pre-flow to High flow period, Pre-flow to the Post-high flow period, Trial 1 to Trial 2, Trial 1 to High flow period, Trial 1 to the Post-high flow period, Trial 2 to High flow period, Trial 2 to the Post-high flow period, and High flow period to Post-high flow period) for both reaches 2 and 3, and 6 flow comparisons for Reach 4. This results in 26 different flow treatment comparisons for each of four species-age classes for a total of 104 statistical comparisons. Second, statistical tests provide no information on whether a statistically significant result is biologically meaningful. For example, mean weight across two treatments could be significantly different but their means may be very close if the amount of variation in mean weight within each treatment is small.

Thus, our assessment of differences in mean weight across flow treatments is based on an examination of differences in the mean values for each treatment, and the extent to which the error bars at one standard deviation overlap. When these standard deviation error bars do not overlap, it's likely that the difference may be statistically significant. Given uncertainty about the criteria used to define biologically relevant difference in mean weights, and errors associated with whether those differences are related to growth or habitat (as opposed to differences in sample timing or emergence), we did not test for statistical significance in these cases. The graphical comparison of mean weights and their errors provides an efficient way to identify trends or any major differences in treatment effects.

Differences in mean weight (or length) of fish among flow treatments can also be influenced by factors that select for size (e.g., under high flows the incidence of displacement out of the study area may be higher for smaller fish than larger fish, introducing a bias in the estimate of mean size based on fork lengths or weights). This effect was noted as one of the possible explanations for the higher mean weights observed during the High Flow years (2016-2018) in the Year 7 monitoring report (Sneep et al. 2019). To overcome this potential bias and assess the relative fitness of individuals under the different flows, we calculated Fulton's Condition Factor (K) to characterize the body condition of each juvenile salmonid measured for length and weight according to the following equation (Anderson and Neumann 1996):

$$K = \frac{W \times 10^N}{L^3}$$

Where:

W is weight in grams;
L is forklength in millimeters; and
N is an integer that scales the condition factor close to a value of 1 (N=5 for LBR).

We then calculated the mean condition factor by species and age class for each flow trial and reach, as well as the standard deviations.

2.8.4. Juvenile Fish Production: Abundance & Biomass

The abundance and biomass of juvenile salmon in each reach was estimated with a hierarchical Bayesian model (HBM) described in Bradford et al. (2011) and Appendix B. Note that minor modifications to priors used in Bradford et al. (2011) were made to account for sparse catches which began in 2015. These modifications are summarized in Appendix B. The HBM provided annual estimates of abundance for Chinook, coho, and mykiss fry (Age-0+) as well as for mykiss parr (Age-1). We also computed means under five flow regimes which included the original annual average flow release treatments of 0 (Trial 0 - Pre-flow), 3 (Trial 1), and 6 $m^3 \cdot s^{-1}$ (Trial 2), as well as the unplanned high flows from 2016 – 2018 (High Flows) and the three years (2019, 2020 and 2021) of Post-high Flows, to-date.

As described in detail in Appendix B, the effect of each flow treatment was determined based on mean abundance and biomass by reach for each regime. The years used to calculate average abundance and biomass for each treatment are provided in Table 2.3.

Table 2.3 Range of years used to compute average abundance and biomass for each flow treatment in the Lower Bridge River for Chinook, coho, and mykiss fry (Age-0+) and mykiss parr (Age-1).

Treatment	Mean Release	Age-0+	Age-1
Trial 0 – Pre-Flow	0 $m^3 \cdot s^{-1}$	1996-1999	1996-1999
Trial 1	3 $m^3 \cdot s^{-1}$	2001-2010	2002-2010
Trial 2	6 $m^3 \cdot s^{-1}$	2011-2015	2012-2015
High Flows	>18 $m^3 \cdot s^{-1}$	2016-2018	2017-2018
Post-High Flows	~7 $m^3 \cdot s^{-1}$	2019-2021	2020-2021

Note that data from 2000 was not used in the average for the Pre-flow or Trial 1 treatments because the change in flow occurred midway through the growing season and it is unclear how juvenile fish (both fry and parr) would have been affected in that year. There was no need to skip a year during the transition from the Trial 1 to Trial 2 treatments because flow changes occurred at the start of the growing season and prior to the emergence of mykiss fry in that year (2011). Despite a higher peak flow in 2015 (i.e., 20 $m^3 \cdot s^{-1}$ instead of 15 $m^3 \cdot s^{-1}$) owing to particular conditions and reservoir management decisions in that year, 2015 was included in the Trial 2 treatment because the yearly average (i.e., 6.6 $m^3 \cdot s^{-1}$) was still very close to the average for other years in this treatment (i.e., 5.3 to 6.1 $m^3 \cdot s^{-1}$). Age-0+ abundances from 2016 – 2018 were used to compute the average abundance and biomass for the High flow regime, and 2019 – 2021 results were used to characterize the Post-high flow period.

For Age-1 mykiss we did not use data from 2000 or 2001 in the average abundance and biomass for the Trial 1 treatment period. Same as for the fry, the effects of the transition from base flows to the Trial 1 release in August 2000 on that year class of Age-1 fish was unknown. The Age-1 fish in 2001 would have experienced baseline flows during their first 2-3 months after emergence from spawning gravels (as Age-0+ fish in spring 2000), which may have affected survival during this important early life stage. Due to this off-set year effect for Age-1 fish, the first year of transition from Trial 1 to Trial 2 (i.e., 2011), Trial 2 to High flow (i.e., 2016), and High flow to Post-high flows (i.e., 2019) were also not included in the treatment averages for mykiss parr.

2.8.5. Mixed Effects Model for Predicting Juv. Fish Density and Biomass

We used the following mixed effects modes to predict the density (fish/m) and biomass (g/m) for each species and age class:

$$1) \qquad \hat{d}_{r,y} = \rho_r + \phi_{f,r} + \gamma_y + \pi \cdot X_{r,y} + \varepsilon_{r,y}$$

where $\widehat{d}_{r,y}$ is the predicted density (or biomass) in reach *r* in year *y* in log space, ρ_r is a fixed effect of reach (log density when no flow was released from the dam under Trial 0), $\phi_{f,r}$ is a fixed reach-specific effect of flow treatment *f*, γ_y is a random effect of year, $\pi_{r,y}$ is a fixed effect for covariate X (e.g., soluble reactive phosphorous, invertebrate basket counts, etc.), and $\varepsilon_{r,y}$ is the remaining process error. Random year and process error effects are drawn from a zero-centered normal distribution with estimated standard deviations σ_y and σ_p, respectively ($\gamma_y \sim \mathrm{dnorm}(0, \sigma_y)$, $\varepsilon_{r,y} \sim \mathrm{dnorm}(0, \sigma_p)$).

The Age-0+ (fry) data consists of 71 density or biomass estimates across all year-reach combinations over 25 separate years. Data from 2000 for reaches 2 and 3 were excluded because the year was influenced by both trials 1 and 2. The rainbow trout Age-1 (parr) data consists of 70 density or biomass estimates over 25 separate years. Data from 2001 for reaches 2, 3, and 4 were excluded because Age-0+ fish were influenced by both trials 1 and 2. A total of 112 and 111 parameters are estimated when fitting the model to Age-0+ and Age-1 data, respectively. There are more parameters estimated than observations, but the parameters are identifiable because the random year and process error effects are constrained by their prior distributions. Owing to the penalty associated with these priors, the model will attempt to explain as much of the variation as possible using the fixed effects.

The likelihood of the data given the predictions was calculated by comparing the observed ($d_{r,y}$) and predicted log density or biomass (eqn. 1) using,

$$2) \qquad d_{r,y} \sim dnorm(\hat{d}_{r,y}, \sigma_{o_{r,y}}^{-2})$$

where $\sigma_{o_{r,y}}^{-2}$ is the inverse of the observation error variance (i.e., precision) in the log density or biomass for each reach and year. These values were estimated by the hierarchical Bayesian model (Bradford et al. 2011) and treated as data in the mixed effects model. As some deviation between predictions and observations are expected due to observation error, predictions of log density or biomass are not expected to match the observed values perfectly. In essence, the process variance (σ_p^2) only needs to be large enough to explain the predictions after observation error has been accounted for via eqn. 2. The model was fit in winBUGS using uninformative uniform prior distributions for all fixed effects and variance terms. To achieve adequate model convergence, as assessed by the Gelman-Rubin convergence statistic (Gelman et. al. 2004), we saved every 10th MCMC sample from a total of 25,000 for each of 3 chains initialized with random starting values, after discarding the first 10,000 samples to eliminate initialization effects.

Multi-level R^2 statistics (Gelman and Pardoe 2006, Recchia 2010) were used to describe fit and quantify the explanatory ability of fixed effects. At the data level, the familiar square of the Pearson correlation coefficient indicates how much of the total variation in observed log density or biomass is explained by the model, as determined by both fixed and random effects (e.g., via eqn. 1). We refer to this coefficient as the data r^2. At the next level, the ratio of the variation in predicted log density or biomass across reach and year strata explained by fixed effects, relative to its total variation resulting from both fixed and random effects, is used to quantify the explanatory power of fixed effects. We refer to this coefficient as the fixed effects r^2 (fixed effects r^2). Unstratified and stratified flow effect models were compared using the Deviance Information Criteria (DIC) which is a Bayesian analog of the Akaike Information Criteria (Spiegelhalter et al. 2002). The source code for the model is provided in Appendix C.

The mixed effects model includes a fixed covariate effect to represent factors other than reach and flow-treatment effects on juvenile densities and biomass. Covariate effects were included because a change in covariate value during one or more flow treatments could lead to a bias in estimated flow treatment effects if not accounted for. Furthermore, covariates may be useful for explaining variation in results among years or reaches that is not fully explained by the flow treatments alone. Covariates were selected based on *a priori* assumptions about the influence or effect of a given parameter on the metrics of concern for fish production in the river (i.e., juvenile salmonid abundance and biomass). The selection was also limited to parameters that were consistently measured across the same time series as the juvenile fish data. The covariates selected for the model based on this approach are defined as follows:

Benthic Invertebrates – Mean number of aquatic invertebrates per basket in each reach (covariate name: muBugs) was used as a measure of prey availability in the fall. Prey availability in the summer would have been preferred since this season reflects the growing period for juvenile salmonids prior to sampling in the fall; however, summer prey availability (benthic invertebrate samples) was not available in summer for all study years. Therefore, we assumed a linkage between prey availability in summer and fall (both of these periods occurred after the peak/disturbance flows each year).

SRP – Mean soluble reactive phosphorous concentration (covariate name: muSRP; µg/L) by reach was used as a proxy for prey availability during summer which potentially influences survival and hence abundance in the fall. As mentioned above, direct measures of prey availability from colonization baskets or drift would have been preferred, but invertebrate data were not available for the summer period during all study years.

Summer Water Temperature – Growth of juvenile salmon and trout is expected to increase with summer water temperature because fish can forage more effectively and convert prey into energy. Survival is expected to be higher when growth is higher because fish will spend less time foraging and therefore be less vulnerable to predation. We used mean water temperature (°C) for the period 21 June to 20 September (covariate name: muSummerTemp) in each reach.

Incubation Time – An effect of the flow release has been increased water temperatures in the study area (particularly reaches 3 and 4) during fall, which results in a reduced incubation period for Chinook eggs and earlier emergence from spawning beds. Early emergence can affect survival or movement out of the study area prior to the fall stock assessment sampling. Mean incubation time summarized by reach (covariate name: muCH_IncubDays), or the number of days from peak spawning to predicted emergence, was applied to the Chinook results only since this was the species affected by the thermal regime shift (see Section 3.1.5).

Spawner Escapements – Annual spawner escapements (# of spawners per km in each reach) were used to evaluate the potential effect of stock sizes on juvenile recruitment for Chinook (covariate name: CH_Sp.km), coho (covariate name: CO_Sp.km) and steelhead (covariate name: Chil_SH_Sp.km and Thom_SH_Sp.km). The AUC estimates (provided by BRGMON-3) were used for the time series of escapements for Chinook and coho. Since an equivalent time series of escapement data for steelhead was not available for the LBR, we used escapement estimates for the Chilcotin and Thompson populations (provided by Rob Bison, MFLNRORD) as a surrogate index, which assumes the LBR population has followed a similar trend in abundance across the study period. Obviously, a portion of the mykiss production in the LBR is from resident rainbow trout. However, there were no spawner abundance data for rainbow trout so we could only test for steelhead escapement effects in this case. Given that there was some uncertainty about the distribution of spawners among reaches (particularly the use of Reach 2 for some species) we summarized the escapement data for this covariate in two ways: 1) an equal distribution of spawners/km among reaches; and 2) an unequal distribution of spawners/km based on the redd distribution data provided by BRGMON-3 (for all three species).

2.8.6. Stock-Recruitment Analysis

Estimates of juvenile salmonid abundance and biomass reflect the productive capacity of reaches in the LBR if they are adequately ‘seeded’. That is, if the escapement to these reaches is sufficient so that fry and parr numbers are not limited by the number of fertilized eggs deposited in the gravel. If escapement is not sufficient to fully seed the habitat, fry and parr abundance and biomass will not reflect habitat conditions in the LBR (as affected by flow and other factors). The effect of escapement on fry production can be examined using a stock-recruitment (S-R) analysis, where the escapement in one calendar year is related to the fry produced from that escapement which is measured in the following calendar year.

Currently, escapement estimates for Chinook, coho and steelhead are generated by the BRGMON-3 Lower Bridge River Adult Salmon and Steelhead program (conducted by Instream Fisheries Research). However, a historical time series of escapement estimates (i.e., covering an equivalent time frame as the juvenile abundance data) are only available for Chinook and coho. As such, we were able to conduct stock-recruitment analysis for coho and Chinook salmon using annual estimates of escapement to evaluate the assumption of full seeding. However, the time series of escapement data for steelhead is too sparse to support stock-recruit analysis for this species at this point.

Escapement estimates for Chinook and coho in the mainstem LBR upstream of the confluence with the Yalakom River were derived from a modified area-under-the-curve (AUC) method (White et al. 2021). Escapement estimates for these species represent abundance in reaches 3 and 4 only as this is where the longest time series of stream walks have been conducted. Counts were expanded to estimates of the number present based on estimates of observer efficiency, which were determined from mark-resight data. A normal distribution was fitted to the expanded count data from each year, and the total escapement was determined by dividing the area under the normal curve by the survey life. The escapement estimates for each calendar year were plotted against fry abundance the following calendar year (e.g., Chinook spawning in September of 2020 produced fry that were sampled in the fall of 2021). We then fit the following Beverton-Holt model to these data,

$$F_{y+1} = \frac{\alpha \cdot E_y}{1 + \frac{\alpha}{\beta} \cdot E_y} \cdot e^{\lambda_j}$$

where F is fry abundance in year *y*+1, E is escapement in year *y*, α is the maximum productivity (fecundity/female * proportion of females * maximum egg-fry survival rate) which occurs when escapement is very low, β is the carrying capacity for fry, and λ is a parameter reflecting the effect of flow treatment *j* on the stock-recruitment relationship. For Trial 0 (0 $m^3 \cdot s^{-1}$ pre-flow period), $\lambda_{j=1}$ was fixed at 0. As e^0=1, α and β therefore represent the stock-recruitment curve under the pre-treatment conditions. Estimates of $e^{\lambda j}$ for j=2, 3, 4 and 5 represent how much the stock-recruitment curve shifts under the 3 and 6 $m^3 \cdot s^{-1}$ treatments, and under the high flow (2016-2018) and post-high flow conditions (2019 to 2021), respectively. This approach for modelling habitat effects on freshwater stock-recruitment relationships is the same as used by Bradford et al. (2005) in their power analysis of evaluating the response of salmon populations to experimental habitat alterations.

Parameters of the stock-recruitment model were estimated in R using the optim non-linear search routine (R Core Development Team 2009) by maximizing the log-likelihood returned from a normal distribution comparing predicted and observed log-transformed fry abundances (i.e., recruitments). Chinook and coho escapements used in the analysis represent the number of fish spawning in the LBR upstream of the Yalakom River confluence. Fry abundance estimates used in the analysis represent the total abundance across reaches 2 and 3 (pre-treatment condition) and 2, 3, and 4 (other treatments and high flows). Thus, we assume that: 1) there is minimal spawning in the LBR downstream of the Yalakom River confluence (which is supported by information collected under BRGMON-3); and, 2) that fry in Reach 2 are produced from fish that spawned upstream of the Yalakom River confluence.

Owing to the pattern in escapement-fry data, the estimated initial slope (α) of the unconstrained stock-recruitment model was unrealistically large. This occurred because observations of escapement near the origin still produced relatively high fry numbers. The initial slope of the escapement-fry stock-recruitment curve is the product of fecundity-sex ratio, and the maximum egg-fry survival rate at low density (from fertilization until the fall standing stock assessment). We constrained the initial slope based on assumed fecundity (5000 eggs/female for Chinook, 1500 eggs/female for coho), sex ratio (0.5), and maximum egg-fry survival rates (0.5 to 0.05). These estimates cover the wide range of values reported in Bradford (1995). We compared the fit of these alternate stock-recruitment models based on the difference in their log-likelihood values.

3. Results

3.1. Physical and Chemical Habitat Parameters

3.1.1. Discharge and Site-specific Flow

Mean annual flows matched the intended targets in Trials 1 (3 $m^3 \cdot s^{-1}$ released from the dam) and 2 (6 $m^3 \cdot s^{-1}$ released) from no flow releases in Trial 0 (Figure 3.1). Groundwater and tributary inflows including the Yalakom River added to releases from the dam. During the high flow years, mean annual flow in Reach 4 was 19.5 $m^3 \cdot s^{-1}$ and it was 28 $m^3 \cdot s^{-1}$ in Reach 1. The added flow with distance downstream was from tributaries. Mean annual flows in the post-high flow years were similar to those during Trial 2. Among all trials, mean annual flow in the Yalakom River was 4.3 – 5.2 $m^3 \cdot s^{-1}$, which increased flow in the Lower Bridge River by 2.2 times during Trial 1, 1.7 times during Trial 2, 1.3 times during the high flow years, and 1.7 times during the post-high flow years.

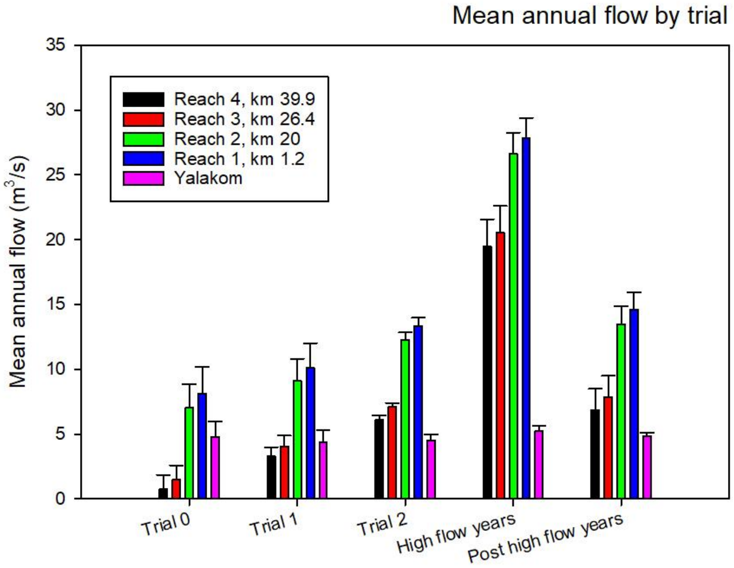


Figure 3.1 Mean annual flow by trial and reach (± standard deviation). Trial 0 was 1996 – July 31, 2000, Trial 1 was August 1, 2000 – May 2, 2011, Trial 2 was May 3, 2011 – December 31, 2015, High flow years were 2016 - 2018, and Post-high flow years were 2019 – 2021.

In 2021, the flow release from Terzaghi Dam generally conformed to the Trial 2 hydrograph for the entire year, but with a higher peak flow than in 2019, 2020 and the Trial 2 years (2011 to 2015) (see Figure 1.3 in Section 1.3). Minimum flows were 1.5 – 2.2 $m^3 \cdot s^{-1}$ from 1 January to 24 February, and 6 October to 31 December 2021. Flows were ramped up from the winter lows in a series of steps between 25 February and 5 May and reached a stable peak of ~25 $m^3 \cdot s^{-1}$ from 6 May to 28 July. Flows were ramped down from 25 to 3 $m^3 \cdot s^{-1}$ in a series of ten steps between 29 July and 24 August, and then down to the minimum release (i.e., 1.5 $m^3 \cdot s^{-1}$) in four steps on 5, 6, 19 and 20 October. Mean annual discharge from the dam in 2021 was 8.7 $m^3 \cdot s^{-1}$. 2021 was the third year of lower flows following the three consecutive years of high flows from 2016 to 2018 (MAD >18 $m^3 \cdot s^{-1}$; see Section 1.3).

Within the study area, estimated flow volumes across the index monitoring sites in reaches 3 and 4 (i.e., sites A to E) were the most consistent across the year and followed the shape of the release hydrograph quite closely, reflecting the small amount of tributary inflow to this section (Figure 3.2). The maximum difference in flow volume between Site A (top of Reach 4) and Site E (bottom of Reach 3) was 2.3 $m^3 \cdot s^{-1}$ in 2021, an increase of 9% at the peak of freshet. Between the bottom of Reach 3 (i.e., Site E) and the top of Reach 2 (i.e., Site F), differences in flow volumes were greater, particularly during the freshet period between early May and mid July. This difference reflected the influence of the Yalakom River inflows at the top of Reach 2. The maximum difference between Site E (bottom of Reach 3) and Site F (top of Reach 2) was 19.2 $m^3 \cdot s^{-1}$, an increase of 67%. The difference in flows across sites in reaches 2 and 1 was moderate (i.e., 2021 max. = 4.4 $m^3 \cdot s^{-1}$, or an increase of 9%,).

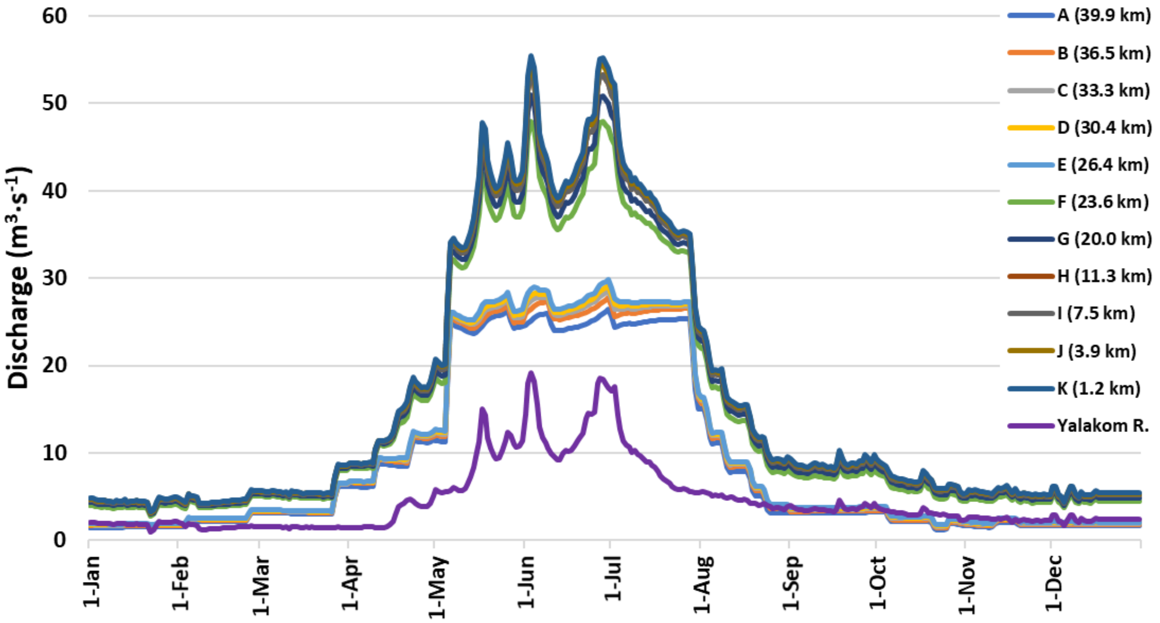


Figure 3.2 Site-specific discharge estimates (shown as mean daily values) in the Lower Bridge River during 2021. Site A is in Reach 4, sites B to E are in Reach 3, sites F and G are in Reach 2, and sites H to K are in Reach 1. 2021 Yalakom River discharge is also shown.

The Yalakom hydrograph in 2021 had three main peaks: an initial peak in mid May (up to 15.0 $m^3 \cdot s^{-1}$), followed by a higher second peak in early June (up to 19.2 $m^3 \cdot s^{-1}$), and then a third peak up to 18.6 $m^3 \cdot s^{-1}$ in late June (Figure 3.3). Each peak was at, or just over, the 90th percentile discharge for their respective period, although there have been many peaks at or beyond those levels in previous years (i.e., up to ~43 $m^3 \cdot s^{-1}$). The duration of each peak on the Yalakom River tends to be relatively brief. MAD for the Yalakom River in 2021 was 4.6 $m^3 \cdot s^{-1}$, which was the same as the median value for the 26-year monitoring period. The peak release flows from

Terzaghi Dam (i.e., from 6 May to 28 July 2021) overlapped the timing of the peak period on the Yalakom River (i.e., from approx. 12 May to 10 July 2021), although the LBR flows started ramping up about a month earlier and were ramped down about two weeks later (Figure 3.2).

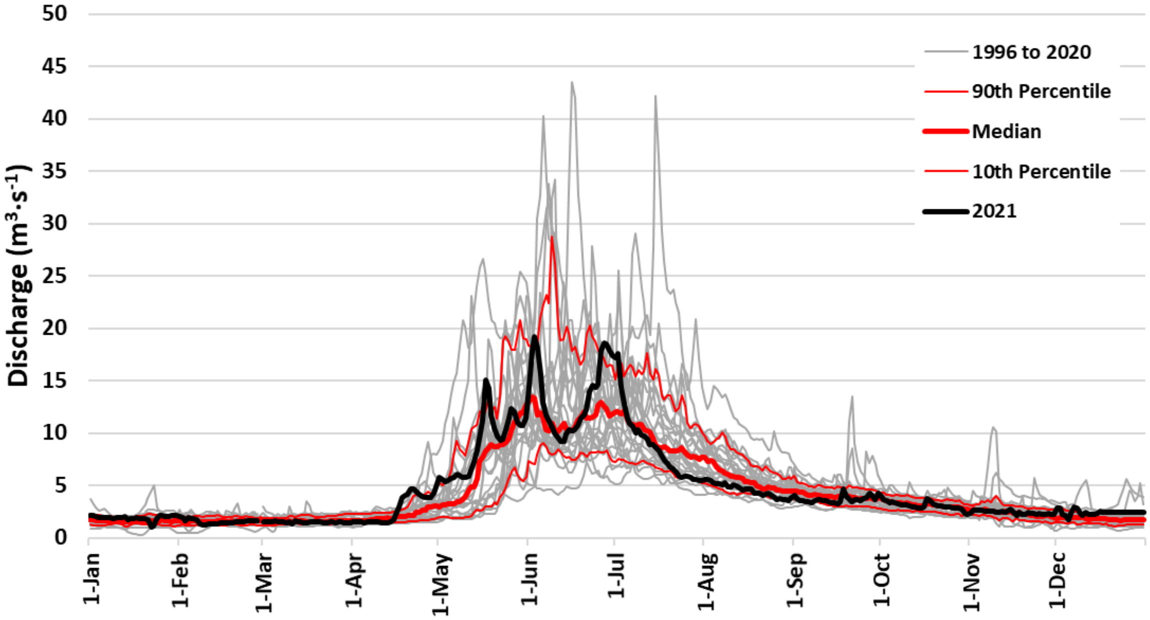


Figure 3.3 Mean daily discharge in the Yalakom River for the period 1996 to 2021. The 10th and 90th percentiles are represented by the thin red lines (lower and upper, respectively) and the median is the bolded red line. The 2021 hydrograph is shown as the bolded black line.

Reach-specific peak flow (highest daily flow among years within a trial) increased from Trial 0 to Trial 2 in association with flow releases from the dam (Figure 3.4). It increased several times further during the High Flow period, reaching 109 $m^3 \cdot s^{-1}$ in Reach 4 and 136 $m^3 \cdot s^{-1}$ in Reach 1. In the Post-high flow years (2019-2021), the spring-summer peak flow preceding the fall biological sampling was similar to that during Trial 2. The variation among peak flows by trial contrasted with relatively consistent peak flows in the Yalakom River (i.e., ranging from 22 – 25 $m^3 \cdot s^{-1}$ among the flow treatment periods).

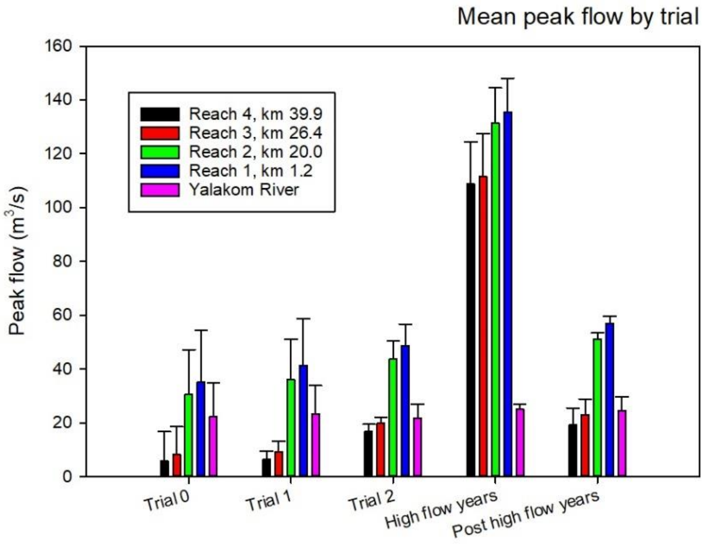


Figure 3.4 Peak flow by trial and reach (± standard deviation). Peak flow was the highest daily flow among years within a trial. Trial 0 was 1996 – July 31, 2000, Trial 1 was August 1, 2000 – May 2, 2011, Trial 2 was May 3, 2011 – Dec 31, 2015, High flow years were 2016 - 2018, and Post-high flow years were 2019 – 2021.

3.1.2. Wetted habitat area

Wetted habitat area was a logarithmic function of site-specific flow in all reaches (Figure 3.5). Without correction for reach length, the curves show wetted areas in Reach 3 > Reach 2 > Reach 4 (Figure 3.5 A). With correction for reach length, the curves show smaller wetted areas per km in Reach 3 than in Reaches 2 and 4 for a given flow (Figure 3.5 B). This difference was due to greater confinement of the channel in the Reach 3 canyon than in the other reaches. Given that wetted area was measured over different ranges of flows between reaches, change in wetted area among flows was only comparable between reaches where flows were the same: 5 – 15 $m^3 \cdot s^{-1}$. Application of the models in Figure 3.5 showed that this 300% increase in flow (5 – 15 $m^3 \cdot s^{-1}$) produced an increase in wetted area of 9.3% in Reach 2, 14.5% in Reach 3, and 17.2% in Reach 4.

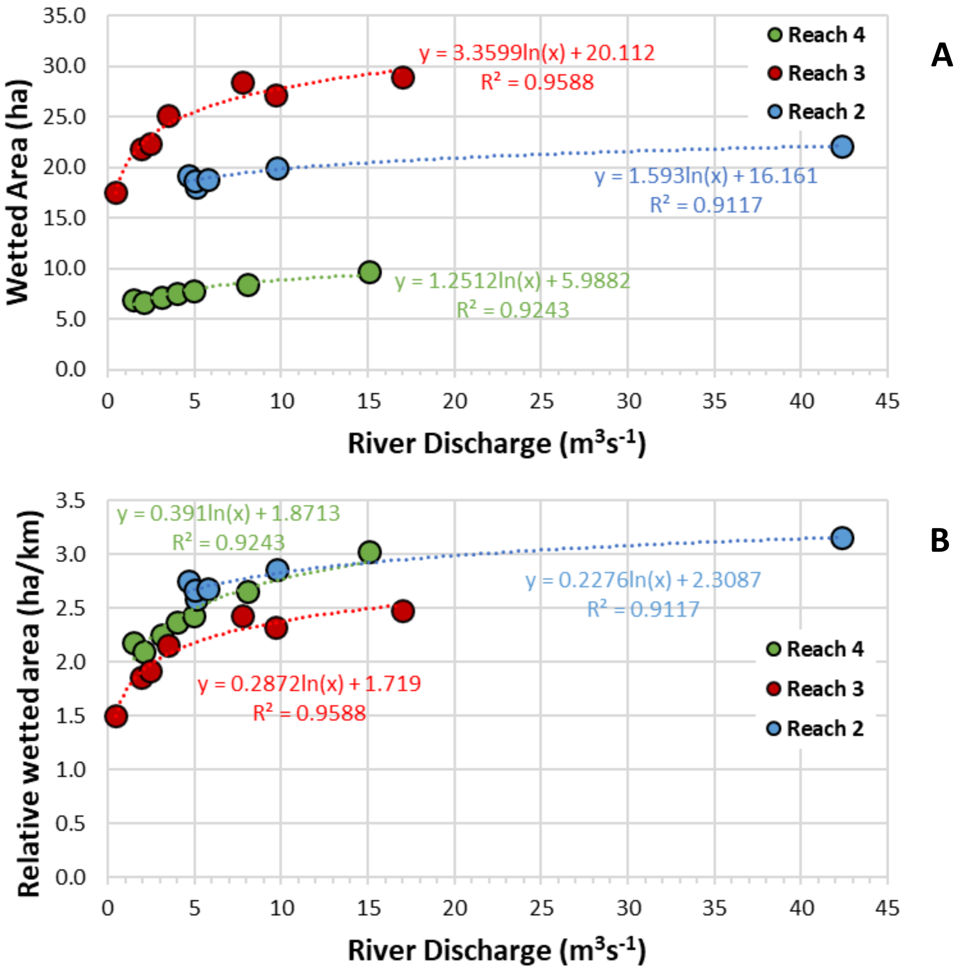


Figure 3.5 **Relationships between river discharge and wetted habitat area, by reach, (A) and between discharge and relative wetted area (wetted area divided by reach length) (B) in the lower Bridge River.**

3.1.3. Sediment Particle Size Distribution

The median particle size of river substrata (D_{50}) measured in 2019 followed a generally diminishing gradient of size from the upstream reach to the downstream reach (Table 3.1). The 2019 D_{50} values by reach were: 11.7 cm in Reach 4, 7.6 ± 1.1 cm in Reach 3, 8.9 ± 0.3 cm in Reach 2, and 7.3 ± 2.5 cm in Reach 1. These values were all larger than the 3.4 ± 0.9 cm D_{50} measured in the Yalakom River.

In 2021, a nearly opposite pattern of median sizes was measured among the reaches (i.e., the D_{50} generally increased from Reach 4 to Reach 1) although the median size range was generally similar. The 2021 D_{50} values by reach were: 7.6 cm in Reach 4, 7.3 ± 0.6 cm in Reach 3, 12.5 ± 10.1 cm in Reach 2, and 13.2 ± 5.0 cm in Reach 1, although the standard error was much larger in reaches 1 and 2 in 2021, where the biggest differences were noted. The 2021 D_{50} in the Yalakom River was again smaller than any of the LBR reaches at 5.2 ± 0.7 cm. The differences in D_{50}

between these two years is more likely a reflection of measurement error or methodological differences than actual changes in substrate composition, particularly given that peak flows were not very high between these sampling events and not likely to move substantial volumes of sediment (relative to the high flow years).

Table 3.1 Sediment particle size summarized as D_{50} among sites within the Lower Bridge River reaches and the Yalakom River (measured in 2019 and 2021).

Reach number	D_{50} mean ± standard error of arithmetic mean (cm)	
	2019	2021
Reach 4	11.7 ± 0	7.6 ± 0
Reach 3	7.6 ± 1.1	7.3 ± 0.6
Reach 2	8.9 ± 0.3	12.5 ± 10.1
Reach 1	7.3 ± 2.5	13.2 ± 5.0
Yalakom River	3.4 ± 0.9	5.2 ± 0.7

3.1.4. River Stage

The pattern of relative stage levels across the year at the top of Reach 3 (i.e., Rkm 36.8) mirrored the flow release hydrograph very closely (Figure 3.6). Unfortunately, river stage level data for the other monitored locations (included in the legend on Figure 3.6) were not available from CNBS at the time of writing this report. As such we are not able to comment on the relative differences in stage levels, etc. among locations in 2021.

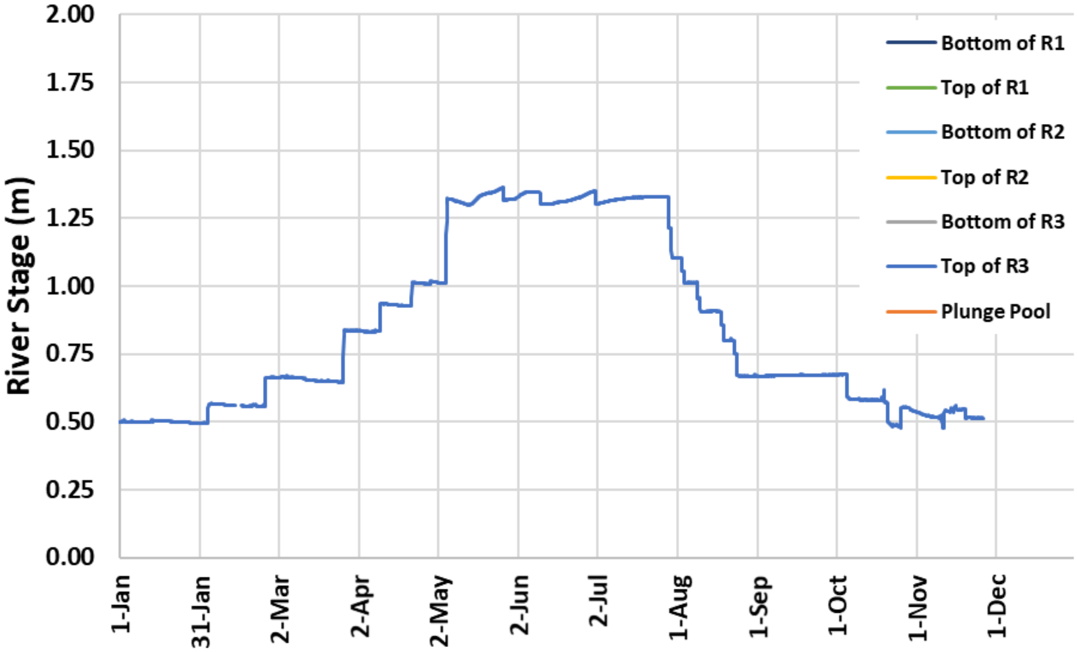


Figure 3.6 **Relative river stage levels (m) according to location in the Lower Bridge River in 2021.**

At the top of Reach 3 (i.e., the compliance location for monitoring flow ramp downs), the stage elevation changes tracked with the timing and shape of the flow release hydrograph quite closely. The river stage changed from a relative minimum value of 0.48 m at base flows to 1.36 m at peak flows (a total difference of 0.88 m) under the 2021 hydrograph at this location (Table 3.2). The mean stage change was 3.3 cm for every 1 $m^3 \cdot s^{-1}$ flow change. In previous years, the Plunge Pool site was the most sensitive to flow changes from the dam (i.e., 7.3 cm for every 1 $m^3 \cdot s^{-1}$ flow change in 2020), and the stage elevation changes at the bottom of Reach 3 were lower and very similar to the top of the reach (i.e., 3.5 cm per 1 $m^3 \cdot s^{-1}$ flow change in 2020). The river stage profiles for monitoring locations in reaches 1 and 2 closely followed the discharge profile of the Yalakom River, reflecting the dominant influence of this major inflow source in the lower reaches under the Trial 2 flow release hydrograph. In 2020, the mean stage change was <3 cm per 1 $m^3 \cdot s^{-1}$ flow change at all monitoring locations below the Yalakom confluence (i.e., 1.0, 1.2, 2.0, and 2.9 cm per 1 $m^3 \cdot s^{-1}$ flow change at the top and bottom of Reach 2 and Reach 1, respectively). The slightly higher values in Reach 1 (relative to Reach 2) were likely due to greater channel confinement at the monitoring locations in this reach (Sneep et al. 2021).

Table 3.2 Summary of flow changes, stage changes and rate of stage change per flow increment in 2021 at various locations in the Lower Bridge River under Modified Operations, and in the Yalakom River.

Location	River km	Site-specific Discharge ($m^3 \cdot s^{-1}$)			Site-specific Stage (m) [a]			Mean Stage Change per Flow Increment ($cm/m^3 \cdot s^{-1}$)
		Min.	Max.	Diff.	Min.	Max.	Diff.	
Plunge Pool	40.9	1.2	26.4	25.2	NA	NA	NA	NA
Top of Reach 3	36.8	1.4	27.7	26.3	0.48	1.36	0.88	3.3
Bottom of Reach 3	26.1	1.8	29.8	28.0	NA	NA	NA	NA
Top of Reach 2	23.6	2.8	48.0	45.2	NA	NA	NA	NA
Bottom of Reach 2	20.0	3.0	51.0	48.0	NA	NA	NA	NA
Top of Reach 1	11.3	3.1	53.5	50.4	NA	NA	NA	NA
Bottom of Reach 1	3.9	3.2	54.7	51.5	NA	NA	NA	NA
Yalakom River [b]	NA	1.0	19.2	18.2	NA	NA	NA	NA

[a] River stage level data were only available for the site at the top of Reach 3 (provided by BC Hydro Power Records) in 2021.
[b] River stage levels were not monitored in the Yalakom River.

Specific stage changes associated with flow ramp down events at Terzaghi Dam are discussed further in Section 3.5.3.

3.1.5. Water Temperature

Relative to the Pre-flow period (Trial 0), dam releases have caused water temperatures to be cooler in the early spring period (Mar-Apr), and warmer throughout the fall during each treatment (Figure 3.7). These effects were most evident in reaches 4 and 3, with a gradient of effect among stations associated with proximity to the dam. In addition to continuation of these effects, the High Flow years from 2016 to 2018 were also characterized by some warmer temperatures in January and February, and during the period of the year when the high flows were delivered, typically from May to July. The flow releases during the Post-high Flow years (2019-2021) generally followed the Trial 2 hydrograph (except for higher peak flows in 2021) and the temperature profiles by reach generally followed the patterns observed for the Trial 2 years. However, there were a few exceptions: the mean temperatures in winter (i.e., January to March) for the Post-high treatment were among the lowest of the range (driven primarily by the low temperatures during this seasonal period in 2019), and mean temperatures in summer (i.e., June to September) for the Post-high treatment were higher than the mean temperatures in Trials 1 and 2 (driven primarily by warmer temperatures in September 2020 and June to August 2021), but lower than the High Flow treatment for this seasonal period. Refer to Appendix D for plots showing all the individual years in each treatment period).

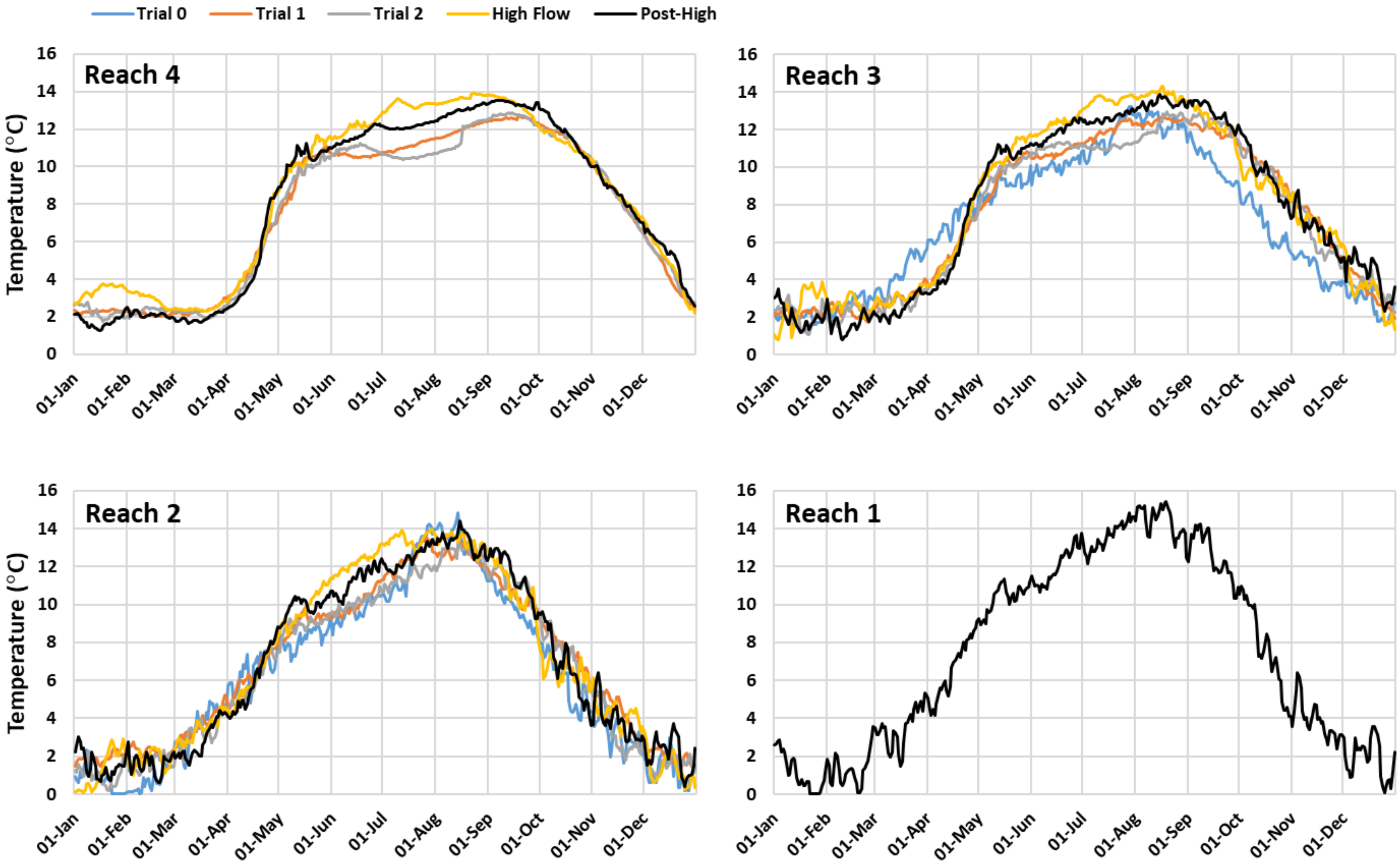


Figure 3.7 Mean daily water temperatures during Trial 0 (pre-flow), Trial 1 (3 $m^3 \cdot s^{-1}$), Trial 2 (6 $m^3 \cdot s^{-1}$), High Flow years (2016-2018), and the Post-High Flow years (2019-2021) for Reach 4 (top left), Reach 3 (top right), Reach 2 (bottom left) and Reach 1 (bottom right). Note the temperature loggers in Reach 1 were installed on 16 October 2019.

Due to the influence of tributary inflows (particularly the Yalakom River at the top of Reach 2), water temperatures tended to be more widely variable on a day-to-day basis in the lower reaches.

The cause of the cooler water temperatures in February 2019 and the warmer temperatures in September 2020, and June to August 2021 were likely due to ambient temperature influence since these effects were apparent in most of the reaches and the Yalakom River (Appendix D). Mean monthly air temperatures were colder in February 2019 and warmer in September 2020 and June-July 2021 than they were in those months during the previous flow treatment periods (Table 3.3; Data provided by Environment Canada). Some ambient temperatures in June and July 2021 were at or near record high levels due to a "heat dome" that developed over the Province of BC, which elevated the mean temperatures for those months.

Table 3.3 Mean monthly air temperatures for Lillooet, BC summarized by LBR flow treatment. The individual Post-High flow years are also included. Data provided by Environment Canada.

Flow Period	Mean Monthly Air Temperatures (± SD)												Period Average
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	
Trial 0	-1.8 (±3.0)	3.4 (±0.2)	5.8 (±1.1)	10.4 (±0.6)	15.4 (±3.0)	19.2 (±1.7)	22.4 (±3.2)	23.0 (±1.1)	17.9 (±1.6)	9.5 (±0.5)	4.6 (±0.5)	0.7 (±1.4)	10.9 (±0.8)
Trial 1	-1.6 (±2.0)	1.3 (±1.3)	5.3 (±2.2)	10.3 (±1.3)	15.2 (±1.2)	19.4 (±1.5)	23.4 (±1.2)	22.0 (±0.9)	16.9 (±1.2)	9.7 (±0.9)	2.8 (±2.9)	-2.3 (±2.9)	10.2 (±0.4)
Trial 2	-1.1 (±1.5)	1.3 (±3.4)	5.8 (±1.9)	10.0 (±1.0)	15.6 (±1.7)	19.5 (±1.9)	23.5 (±1.9)	22.9 (±0.6)	17.6 (±1.5)	10.5 (±1.3)	2.1 (±0.8)	-0.4 (±0.6)	10.6 (±0.7)
High Flows	-2.2 (±1.6)	0.6 (±2.9)	6.3 (±1.6)	11.3 (±2.4)	17.3 (±1.6)	19.9 (±0.7)	23.2 (±1.4)	22.7 (±1.3)	16.2 (±1.9)	8.9 (±0.7)	4.8 (±2.0)	-2.4 (±3.1)	10.6 (±0.5)
Post-High	-0.1 (±1.1)	-1.6 (±4.7)	5.3 (±0.6)	10.4 (±0.6)	16.6 (±1.3)	20.1 (±2.0)	23.0 (±2.3)	22.7 (±0.3)	16.8 (±2.4)	9.2 (±0.6)	4.0 (±0.7)	-1.5 (±3.9)	10.4 (±0.4)
2019	0.6	-6.4	5.2	10.2	18.0	19.6	21.4	22.7	16.9	8.7	3.4	0.1	10.0
2020	-1.4	2.9	4.8	10.0	16.3	18.3	22.0	22.4	19.2	9.9	3.8	1.4	10.8
2021	0.5	-1.4	5.9	11.1	15.4	22.3	25.6	23.0	14.4	9.1	4.8	-5.9	10.4

The warmer water temperatures during the June to August period in the Post-high treatment (relative to Trial 2) may also be influenced by upstream operations as the effect was also observed during the other modified operations years (2016-2018). In these cases, the warmer summer water temperatures may also have been caused by: 1) higher conveyance of water through Terzaghi Dam, which may affect draw from the various thermal layers in Carpenter Reservoir (high flow years only); 2) deeper drawdown of Carpenter Reservoir in the spring, which reduces the depth of water above the intake for the low-level outlet (observed in 2017 to 2020); or 3) some combination of 1) and 2). The CE-QUAL temperature model developed for Carpenter Reservoir under BRGMON-10 could be consulted to better understand the relationship between reservoir operation and release temperatures. However, the analyses required to determine the cause and mechanism of the observed effects were beyond the scope of this report.

Mean water temperature during the periphyton and benthic invertebrate sampling periods (typically September to November) declined upstream to downstream (Table 3.4). Relatively high temperatures in Reach 4 during fall were due to the release of warm hypolimnetic water from Carpenter Reservoir caused by mixing of epilimnetic water that had temperatures exceeding 20^{o}C and cooler hypolimnetic water in the fall (Limnotek 2019). The water cooled with downstream flow. Water temperature was relatively consistent among trials in a given reach except in reaches 2 and 3 during Trial 2 and the High Flow years. Mean water temperatures for the sampler colonization period were lower due to later installation of the plates and baskets in those years, which affected these reaches more than Reach 4 because the tributary inflows are cooling rapidly across this seasonal period (and there are no year-round tributary inflows to Reach 4).

Table 3.4 Mean temperature values ± standard error by reach and trial for the periphyton and benthic invertebrate sampling periods.

Reach	Mean temperature ± standard error of arithmetic mean				
	Trial 0 (May 1996 – Jul 2000)	Trial 1 (Aug 2000 – Mar 2011)	Trial 2 (Apr 2011 – Dec 2015)	High Flow Years (2016 – 2018)	Post-High Flow Years (2019 – 2021)
Reach 4	No flow	10.8 ± 0.2	10.1 ± 0.2	10.2 ± 0.4	10.8 ± 0.2
Reach 3	8.5 ± 1.3	9.7 ± 0.3	8.1 ± 0.2	8.3 ± 0.4	9.2 ± 0.4
Reach 2	7.5 ± 1.4	7.2 ± 0.3	5.6 ± 0.4	5.6 ± 0.5	5.9 ± 0.3
Reach 1	Not measured				6.2 ± 0.3
Yalakom R.	4.7 ± 0.3	3.7 ± 0.3	4.6 ± 0.6	4.1 ± 0.0	4.7 ± 0.3

Water temperatures during the early part of the salmon incubation period in fall (i.e., Sep to Dec) have been elevated during all flow treatment years (2000 to 2021) by up to 4°C at the top of Reach 3 (relative to the pre-flow period - Trial 0). Differences among the flow treatments during that seasonal period were small, as were differences among years within trials (refer to Reach 4 figure in Appendix D). Release flows among all flow trial years have been very similar across the fall period (3.0 $m^3 \cdot s^{-1}$ in Sep; and between 1.5 and 2.0 $m^3 \cdot s^{-1}$ from Oct to Dec in all cases – Figure 1.3 in Section 1.3). Changes to the thermal regime have caused large differences in the timing of juvenile salmon emergence from the spawning beds (based on modelled ATU data and sampling observations). Prior to the flow release the predicted median date of both coho and Chinook salmon fry emergence was late April or early May, with a trend to slightly later timing at downstream sites due to the cooling of water as it flows downstream in the fall months when air temperatures are falling (Figure 3.8). The estimated peak spawning dates for Chinook and coho salmon were September 8 and November 15, respectively, based on observations made during streamwalk surveys conducted as part of BRGMON-3 (White et al. 2021).

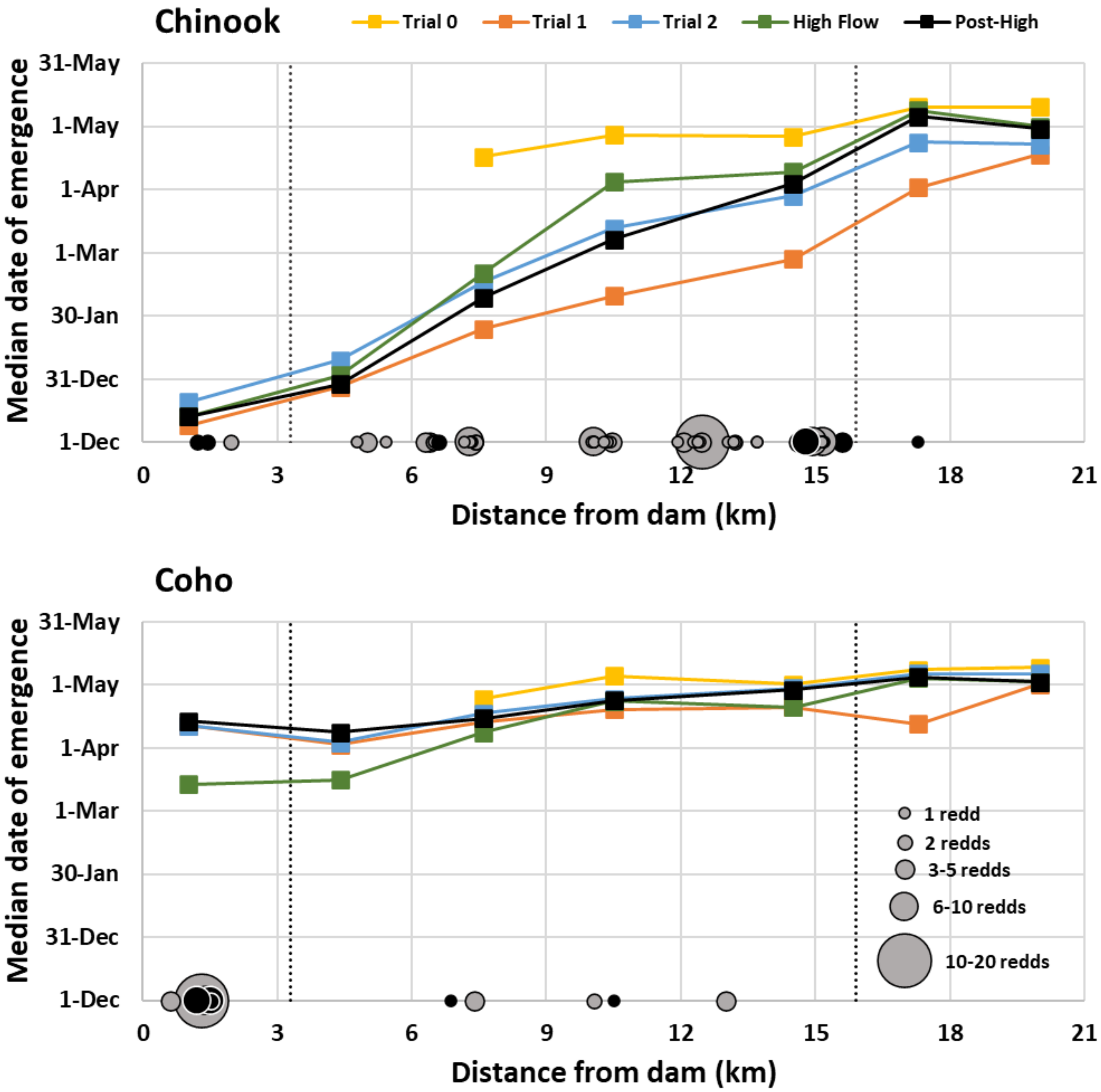


Figure 3.8 **Modelled median emergence dates for Chinook (top) and coho (bottom) fry at varying distances below Terzaghi Dam based on observed daily mean temperatures for each flow treatment. The vertical dotted lines indicate the location of reach breaks. The locations of redds observed for each species during the available period (2014 - 2021) are represented by the dots along the x-axis on each plot (data provided by BRGMON-3). The colour of dot reflects the period (black = 2019-2021 Post-high flow years; grey = 2014-2018) and the size of dot indicates the relative number of redds at each location.**

As noted above, there is a gradient of temperature across the length of the study area during fall, and this gradient influences the predicted timing of emergence according to distance from the dam (Figure 3.8), particularly for Chinook (due to their earlier spawn timing in the fall). For both Chinook and coho, the predicted emergence timing at each monitoring location for the Post-high treatment was very similar to the average estimates for Trial 2 (black and blue points in Figure 3.8). Despite some variation among years, the differences in the mean emergence timing predictions for Chinook between Trial 1 and the subsequent flow treatments starting at locations ~7.5 kms below the dam and continuing downstream are likely due to the difference in fall/winter flow volumes. Fall/winter discharges during the Trial 2, High Flow and Post-high Flow years were 1.5 $m^3 \cdot s^{-1}$, which were 25% lower than the fall/winter flows during Trial 1 (i.e., 2.0 $m^3 \cdot s^{-1}$). Lower release flow volumes are cooled more quickly by the tributary inflows and declining ambient temperatures, which is particularly important during the early part of the incubation period when release flows are warmer. However, predicted emergence timing for Chinook was consistently earlier than the Pre-flow period, particularly in Reach 4 and the upper half of Reach 3, during all of the flow treatments.

Since location-specific spawning information became available starting in Trial 2 (collected under BRGMON-3), Chinook and coho spawners have utilized spawning areas in both reaches 3 and 4, but the distribution of redds among those reaches has been different for the two species (Figure 3.8 and Table 3.5). The total number of Chinook redds observed from 2014 to 2021 was 142. Seven percent of the total (*n*= 10) were observed in Reach 4, and the remaining 93% (*n*= 132) were distributed across Reach 3. Based on the observed distribution, and assuming equivalent survival among locations, approx. 7% of the spawned eggs would have been associated with a predicted median emergence (PME) timing of mid-December (near temperature monitoring site A), and a further 24% would have had a PME timing between early January and mid February (near sites B and C). The remaining 69% would have had a PME of mid- March to early May in the bottom portion of Reach 3 and upper portion of Reach 2.

During the three Post-high Flow years (2019-2021), a total of eight redds, five redds and four redds were observed, respectively, for Chinook although spawning distributions and collection of these data were affected by the operation of a broodstock collection fence installed early in the spawning period at the bottom of Reach 3 (White et al. 2021). Of the observed redds in these three years, 29% (*n*= 5) were observed in Reach 4, 12% (*n*= 2) were observed in the upper portion of Reach 3, 53% (*n*= 9) were observed at the bottom of Reach 3 at or below the fence, and 6% (*n*= 1) were in the horseshoe bend (Reach 2). It is also important to note that, due to the effects of the Big Bar slide on Chinook migration in the Fraser River (upstream of the Bridge River confluence), there was likely an increased incidence of straying into the Bridge River by Chinook from other populations, particularly in 2019 and 2020 (i.e., before passage improvements at the Big Bar site were substantively completed).

Observations for coho redd locations were available for 2018 to 2021, and the total number of coho redds observed in those years was 53 (2018 *n*= 31; 2019 *n*= 6; 2020 *n*= 7; 2021 *n*= 9). Unlike Chinook, a much higher proportion of coho spawning was observed in Reach 4 (77%), with an associated PME timing of 10 April for this species. The remaining 23% of redds were observed in

Reach 3 with corresponding PME timing between 8 and 29 April, according to location (Table 3.5).

Table 3.5 Proportion of Chinook and coho spawning, according to observed redd locations (2014 to 2021), by distance from dam and predicted median emergence timing in reaches 2, 3 and 4.

Species	Reach	Station (Rkm)	Dist. From Dam (km)	Predicted Median Emergence Date	*n*	Percentage of observed redds[a,b]
Chinook	4	A (39.9)	1.0	14-Dec	10	7% (7%)
	3	B (36.5)	4.4	1-Jan	17	12% (19%)
		C (33.3)	7.6	10-Feb	17	12% (31%)
		D (30.4)	10.5	15-Mar	54	38% (69%)
		E (26.4)	14.5	1-Apr	43	30% (99%)
	2	F (23.6)	17.3	5-May	1	1% (100%)
Coho	4	A (39.9)	1.0	10-Apr	41	77% (77%)
	3	B (36.5)	4.4	8-Apr	0	0% (77%)
		C (33.3)	7.6	17-Apr	5	9% (87%)
		D (30.4)	10.5	27-Apr	3	6% (92%)
		E (26.4)	14.5	29-Apr	4	8% (100%)

[a] Values in brackets represent the cumulative percentage of redds observed at, and upstream of, each station.

[b] Values for coho are based on data collected in 2018-2021 only, as these were the only years of redd count data available for this species.

3.1.6. Turbidity

Turbidity in Reaches 4 and 3 during 2021 showed a tri-modal pattern driven by time course change in turbidity in water released from Carpenter Reservoir (Figure 3.9). The first mode was in spring when the release from Carpenter had median daily turbidity up to 80 NTU. That peak declined with distance downstream, reaching 60 NTU at the downstream end of Reach 3. A second mode was in early August having lower median daily turbidity than in the spring. Turbidity then declined to 20 – 30 NTU in Reaches 4 and 3 in late August before a final mode in September and October that was similar in magnitude to the second one. The timing of modes at all sites in Reaches 4 and 3 were similar to that in water released from Carpenter, which showed the pattern was due to in-reservoir processes in 2021.

These three modes were not apparent in Reaches 2 and 1 due to attenuation by inflow from the Yalakom River (Figure 3.10). Only spring-time high and variable turbidity up to 80 NTU was found downstream of the Yalakom River. After July, turbidity declined to eventually reach <10 NTU in November.

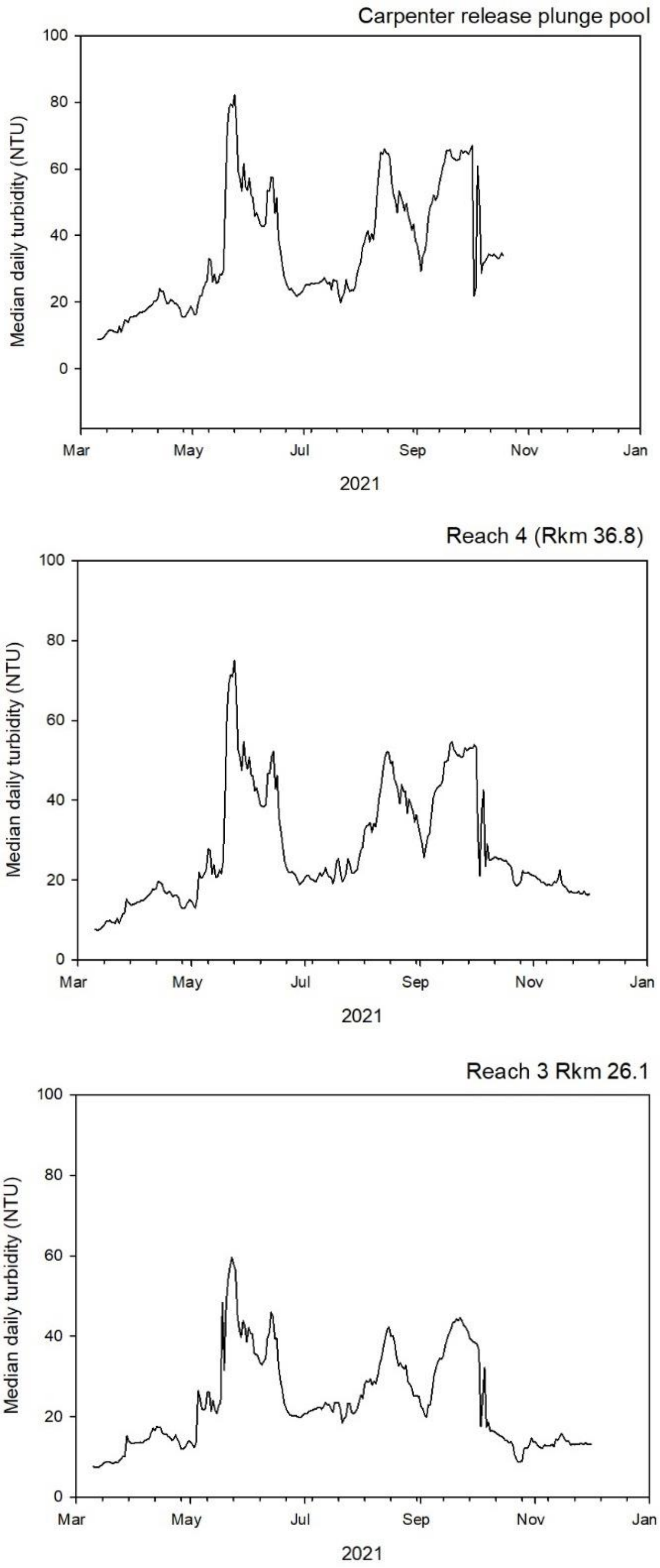


Figure 3.9 Turbidity at monitoring locations in Reaches 4 and 3, March to November 2021.

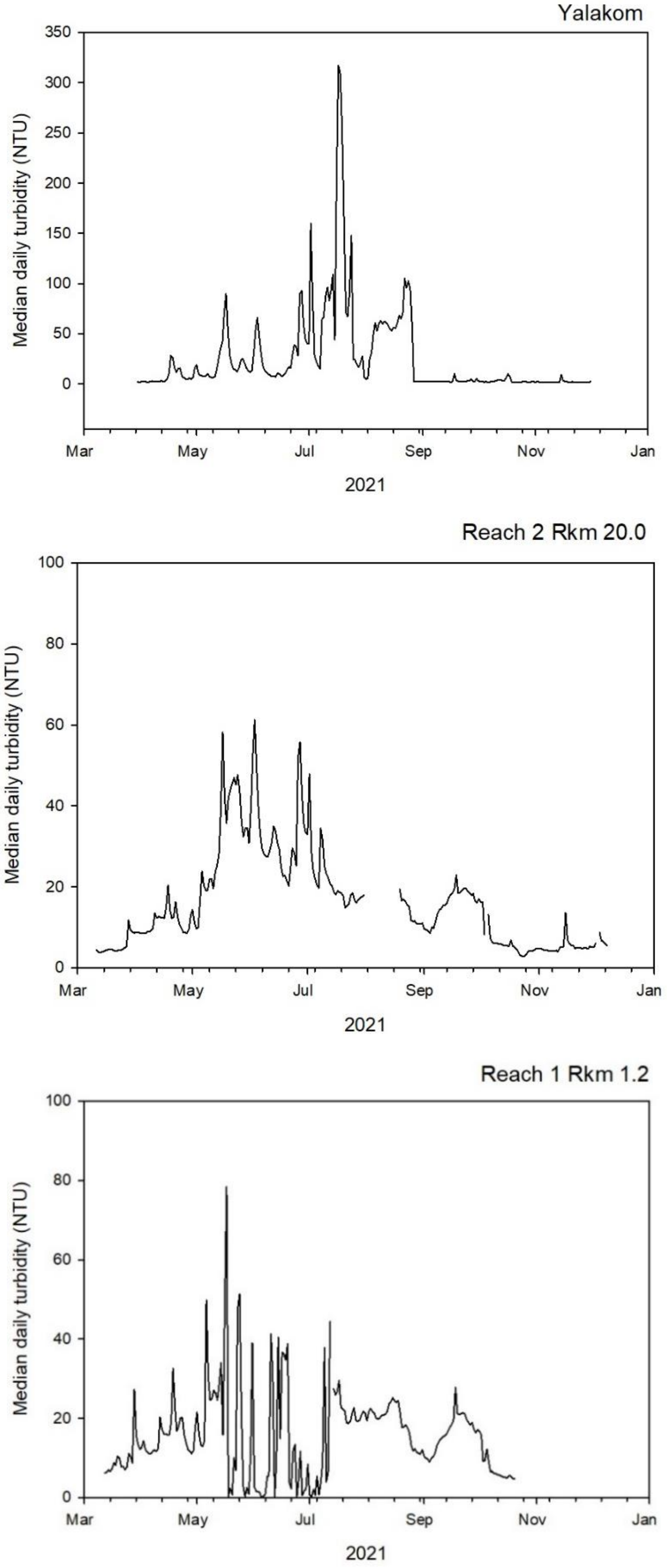


Figure 3.10 Turbidity in the Yalakom River and Reaches 2 and 1, March to November 2021.

3.1.7. Water Chemistry

Total alkalinity and pH increased upstream to downstream (**Table** 3.6). This pattern showed inorganic carbon was added to the Bridge River with distance from the Terzaghi Dam. Tributaries were a source of that alkalinity (Table 3.7). At the pH's of 7.5 to 8.3 found among tributaries, mainstem reaches, and trials, bicarbonate (HCO_3) was the expected dominant form of inorganic carbon (Stumm and Morgan 1996). Separate measurements of bicarbonate, carbonate, and hydroxide alkalinity (data not shown) confirmed bicarbonate dominance.

Total dissolved solids (TDS) concentration increased upstream to downstream, which is consistent with addition of bicarbonate as inferred from alkalinity and pH (**Table** 3.6). Addition of base cations (Na, K, Ca and Mg) may have contributed to this pattern.

With one exception, DIN concentration (sum of NO_3-N and NH_4-N) increased upstream to downstream (**Table** 3.6). The exception was during Trial 0 when Bridge River flow upstream of the Yalakom confluence was limited to seepage. DIN was mainly comprised of NO_3-N in Trial 0 and during the High Flow Years but during Trials 1 and 2, NH_4-N and NO_3-N concentrations were about the same, mainly in Reaches 3 and 2. Tributaries flowing into Reach 3 were a source of NO_3-N but Antoine Creek, a small tributary in Reach 2, had low concentrations of NO_3-N, indicating unusual demand for or loss of N in that stream. The loading of NO_3-N from tributaries in Reach 3 combined with that from the Yalakom River were the primary sources of NO_3-N in Reach 3 and Reach 2.

Relatively high NH_4-N concentrations in Reaches 2 and 3 during Trials 1 and 2 and during the Post-high flow years showed addition of reduced forms of nitrogen. Absence of disturbance and no forest silvicultural activities in the drainage of Reaches 2 and 3 eliminated an inorganic source, which was corroborated with concentrations of NH_4-N less than 7.5 $\mu g \cdot L^{-1}$ in tributaries (Table 3.7). High NH_4-N concentrations in the mainstem during Trials 1 and 2 and during the Post-high flow years must have come from enrichment by organic matter containing nitrogen that did not occur or was less apparent in Trial 0 and during the High Flow Years.

Table 3.6 Bridge River mainstem and Yalakom River mean chemical concentrations or values ± standard error of the arithmetic mean by Reach and Trial in the fall during periphyton and benthic invertebrate sampling.

Chemical variable and units	Reach	Lower Bridge River and Yalakom River mean value ± standard error of arithmetic mean				
		Trial 0 (May 1996 – Jul 2000)	Trial 1 (Aug 2000 – Mar 2011)	Trial 2 (Apr 2011–Dec 2015)	High Flow Years (2016 – 2018)	Post-High Flow Years (2019 – 2021)
Total alkalinity (mg·L^{-1} as CaCO$_3$)	Reach 4	No flow	57 ± 1	42 ± 8	28 ± 1	29 ± 2
	Reach 3	168 ± 5	72 ± 2	56 ± 5	41 ± 2	38 ± 1
	Reach 2	198 ± 10	141 ± 2	111 ± 15	86 ± 1	83 ± 5
	Reach 1	Not measured	Not measured	Not measured	Not measured	85 ± 2
	Yalakom River	Not measured	Not measured	Not measured	113 ± 0	104 ± 3
pH	Reach 4	No flow	7.5 ± 0.1	7.6 ± 0.1	7.7 ± 0	7.5 ± 0.1
	Reach 3	8.2 ± 0.1	7.5 ± 0	7.7 ± 0	7.8 ± 0.0	7.7 ± 0
	Reach 2	8.1 ± 0.1	7.8 ± 0.1	8.0 ± 0	8.1 ± 0.0	8.0 ± 0
	Reach 1	Not measured	Not measured	Not measured	Not measured	8.1 ± 0
	Yalakom River	Not measured	Not measured	Not measured	8.2 ± 0	8.1 ± 0
Total dissolved solids (mg·L^{-1})	Reach 4	Not measured	64 (no replicates)	Not measured	Not measured	58 ± 4
	Reach 3	105 ± 3	55 ± 3	Not measured	Not measured	65 ± 2
	Reach 2	107 ± 1	89 ± 3	Not measured	Not measured	112 ± 14
	Reach 1	Not measured	Not measured	Not measured	Not measured	119 ± 3
	Yalakom River	Not measured	Not measured	Not measured	Not measured	122 ± 3
NH$_4$-N (µg·L^{-1})	Reach 4	No flow	5.4 ± 1.0	11.0 ± 4.4	5.5 ± 0.5	11.0 ± 3.6
	Reach 3	6.1 ± 0.7	28.2 ± 7.5	56.0 ± 17.1	5.8 ± 0.3	30.6 ± 9.9
	Reach 2	9.4 ± 1.6	50.8 ± 14.2	70.2 ± 28.9	5.4 ± 0.4	38.3 ± 13.4
	Reach 1	Not measured	Not measured	Not measured	Not measured	22.9 ± 4.4
	Yalakom River	Not measured	Not measured	Not measured	<5.0	9.0 ± 2.0
NO$_3$-N (µg·L^{-1})	Reach 4	No flow	19.2 ± 2.4	17.6 ± 4.2	10.2 ± 1.3	16.1 ± 3.5
	Reach 3	41.0 ± 7.9	30.5 ± 3.2	67.5 ± 12.9	20.9 ± 2.3	41.1 ± 7.3
	Reach 2	34.8 ± 3.3	46.9 ± 6.2	81.0 ± 13.4	49.6 ± 1.6	65.0 ± 6.0
	Reach 1	Not measured	Not measured	Not measured	Not measured	56.0 ± 5.8
	Yalakom River	Not measured	Not measured	Not measured	76.5 ± 0	62.1 ± 3.2
DIN (µg·L^{-1})	Reach 4	No flow	24.7 ± 2.9	28.6 ± 7.0	15.7 ± 1.5	27.1 ± 7.1
	Reach 3	47.1 ± 7.9	58.7 ± 10.1	123.4 ± 29.3	26.7 ± 23	71.7 ± 16.7
	Reach 2	34.8 ± 3.3	97.6 ± 19.2	151.2 ± 40.6	55.0 ± 1.3	103.3 ± 16.6
	Reach 1	Not measured	Not measured	Not measured	Not measured	78.9 ± 9.6
	Yalakom River	Not measured	Not measured	Not measured	81.5 ± 0	71.1 ± 3.0
SRP (µg·L^{-1})	Reach 4	No flow	3.1 ± 0.3	2.7 ± 0.4	1.5 ± 0.3	1.8 ± 0.4
	Reach 3	3.0 ± 0.2	6.6 ± 1.5	10.2 ± 2.4	1.7 ±0.2	7.3 ± 2.0
	Reach 2	3.7 ± 0.9	9.5 ± 3.0	9.2 ± 4.0	1.1 ± 0.1	5.0 ± 1.7
	Reach 1	Not measured	Not measured	Not measured	Not measured	4.2 ± 0.7
	Yalakom River	Not measured	Not measured	Not measured	1.0 ± 0	1.1 ± 0

Table 3.7 **Mean chemical concentrations or values ± standard error in Bridge River tributaries partitioned within mainstem reaches by reach and trial in the fall during periphyton and benthic invertebrate sampling. Reach 4 is omitted because no tributaries flow into that reach.**

Chemical variable and units	Reach receiving flow from sampled tributaries	Mean concentration or value ± standard error of arithmetic mean among tributaries				
		Trial 0 (May 1996 – Jul 2000)	Trial 1 (Aug 2000 – Mar 2011)	Trial 2 (Apr 2011–Dec 2015)	High Flow Years (2016 – 2018)	Post-High Flow Years (2019 – 2020)
Total alkalinity (mg·L^{-1} as $CaCO_3$)	Reach 3	217 ± 12	210 ± 7	155 ± 13	117 ± 7	117 ± 7
	Reach 2	269 ± 40	285 ± 18	206 ± 27	158 ± 22	155 ± 21
	Reach 1	Not measured	Not measured	Not measured	Not measured	171 ± 23
pH	Reach 3	8.1 ± 0	8.0 ± 0	8.2 ± 0	8.2 ± 0	8.1 ± 0
	Reach 2	8.1 ± 0	8.1 ± 0	8.2 ± 0	8.3 ± 0	8.2 ± 0
	Reach 1	Not measured	Not measured	Not measured	Not measured	8.3 ± 0.1
Total dissolved solids (mg·L^{-1})	Reach 3	119 ± 8	126 ± 18	Not measured	Not measured	163 ± 14
	Reach 2	156 ± 26	176 ± 44	Not measured	Not measured	215 ± 39
	Reach 1	Not measured	Not measured	Not measured	Not measured	233 ± 22
Turbidity (NTU)	Reach 3	Not measured	Not measured	0.7 ± 0.1	0.5 ± 0.1	0.5 ± 0.1
	Reach 2	Not measured	Not measured	0.8 ± 0.1	0.6 ± 0.1	0.8 ± 0.3
	Reach 1	Not measured	Not measured	Not measured	Not measured	7.0 ± 3.3
NH_4-N (µg·L^{-1})	Reach 3	5.1 ± 0.5	3.8 ± 0.5	3.6 ± 0.3	5.0 ± 0	7.1 ± 0.6
	Reach 2	7.5 ± 1.6	7.5 ± 1.1	5.8 ± 1.4	5.7 ± 0.5	7.4 ± 1.0
	Reach 1	Not measured	Not measured	Not measured	Not measured	6.6 ± 0.8
NO_3-N (µg·L^{-1})	Reach 3	102 ± 32	104 ± 17	204 ± 67	180 ± 52	139 ± 44
	Reach 2	17 ± 5	18 ± 4	41 ± 16	29 ± 11	25 ± 9
	Reach 1	Not measured	Not measured	Not measured	Not measured	21 ± 6
SRP (µg·L^{-1})	Reach 3	4.6 ± 0.6	2.4 ± 0.2	3.3 ± 0.4	3.0 ± 0.6	2.9 ± 0.8
	Reach 2	4.0 ± 0.7	2.1 ± 0.2	2.2 ± 0.3	1.2 ± 0.1	1.1 ± 0.0
	Reach 1	Not measured	Not measured	Not measured	Not measured	2.8 ± 0.9

SRP concentration differed by reach and trial (**Table** 3.6). Mean SRP concentration in Reach 4 declined over time from a peak of 3.1 µg·L^{-1} in Trial 1 to 1.8 µg·L^{-1} in the Post-high flow years due to declining SRP concentration in outflow from Carpenter Reservoir over time (Figure 3.11). In contrast, SRP concentration in Reach 3 was greater than in Reach 4 between Trial 0 and Trial 2 indicating addition of phosphorus in Reach 3. The tributaries flowing into Reach 3 carried low and consistent concentrations of SRP, eliminating them as a major source of phosphorus in Reach 3 at any time. The same pattern occurred in Reach 2, which shows the phosphorus source was in the river mainstem downstream of Reach 4. Then during High Flow Years, mean SRP concentrations in both of reaches 3 and 2 declined to <2 µg·L^{-1}, showing less phosphorus enrichment compared to years during Trials 1 and 2. In the Post-high flow years, SRP concentration in Reach 3 increased by more than three times than in High Flow Years while in Reach 4 the SRP concentrations were the same between Post-high flow years and the High Flow

Years. This increase in SRP concentration in the Post-high flow years compared to the High Flow Years did not come from tributaries because they carried relatively unchanged SRP concentrations over the same time periods. Reach 3 SRP concentrations were diluted by the Yalakom inflow that had some of the lowest SRP concentrations in the whole study area at any time. This mixing produced lower SRP concentrations in Reach 2 compared to Reach 3 during the High flow years and the Post-high flow years. SRP concentrations in Reach 1 were about the same as those in Reach 2 during the Post-high flow years, the only years of chemical measurements in Reach 1.

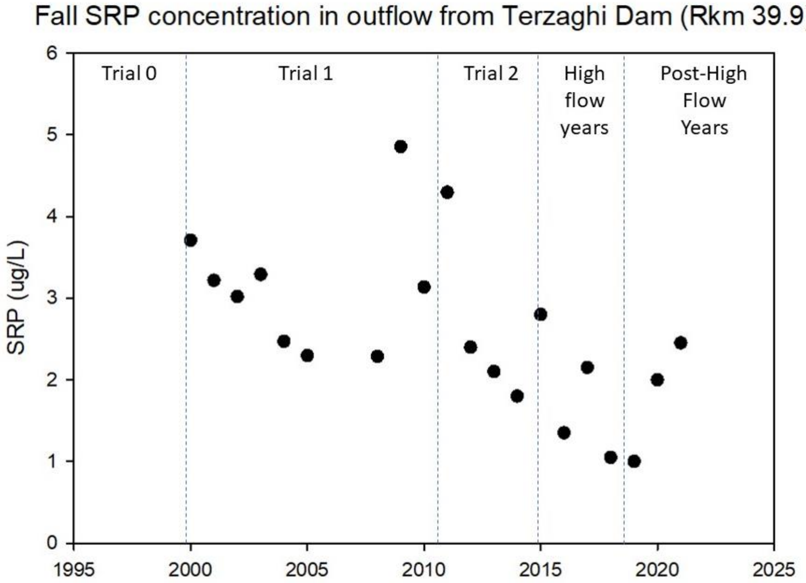


Figure 3.11 Fall SRP concentration in water released from Terzaghi Dam (Site A; Rkm 39.9) among years of all flow trials.

Relatively high mean SRP and NH_4-N concentrations in Reaches 3 and 2 during Trials 1 and 2 resulted from high concentrations in many of the odd years but low concentrations in even years (Figure 3.12). These interannual differences coincided with the timing of the pink salmon spawning runs that were dominant in odd years and absent in even years (Grant et al. 2014). The coincidences were pronounced in 2003, 2005, 2009, 2011, 2013, and 2021. Unfortunately, there are no data describing the size of pink runs in this part of the Fraser River system (D. Sneep, Fisheries and Oceans Canada, Pers. Comm. April 12, 2022), which prevents us from looking at associations between pink run size and nutrient concentrations in the lower Bridge River.

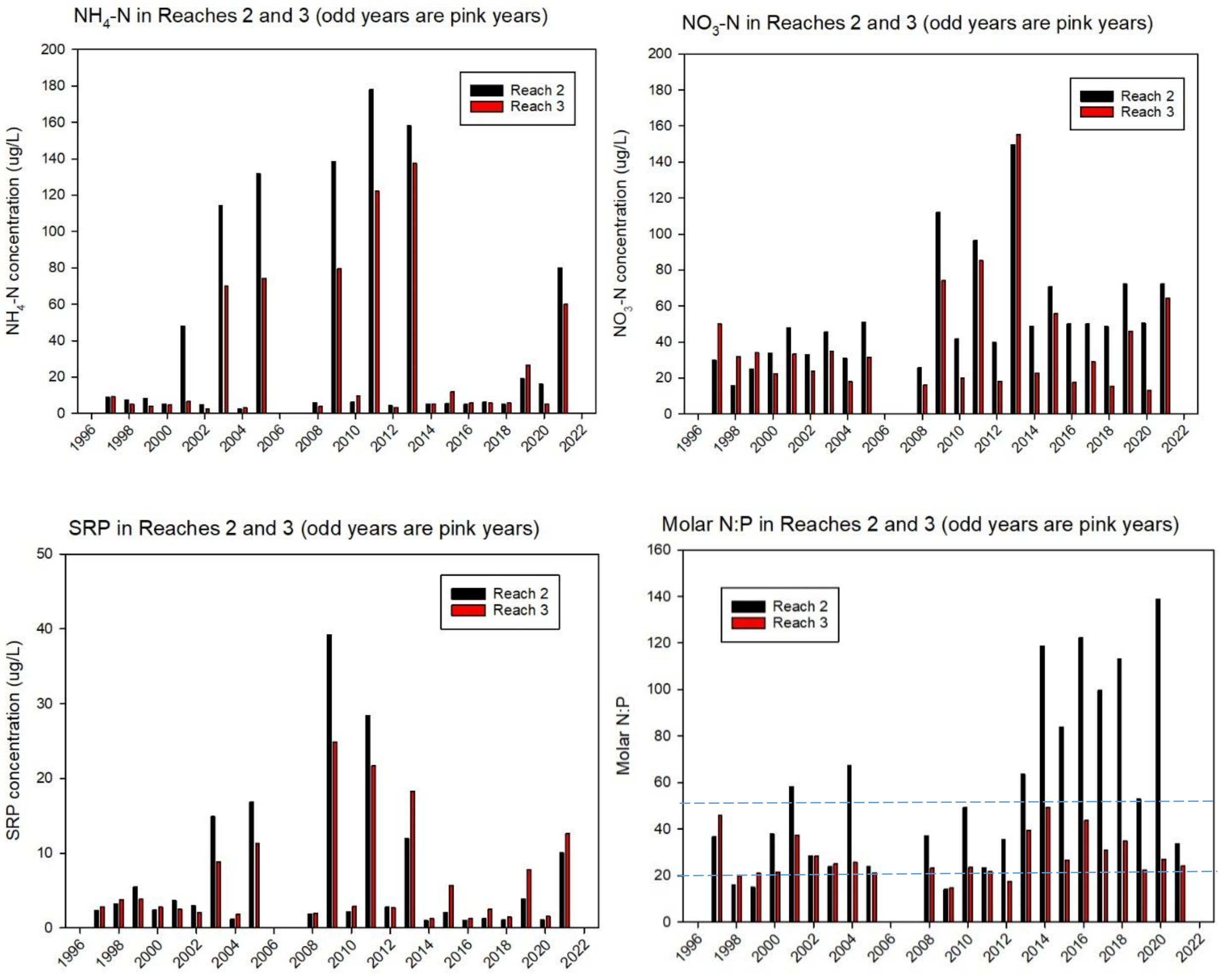


Figure 3.12 **Concentration of NH₄-N (top left), NO₃-N (top right), and SRP (bottom left) with molar N:P (bottom right) among years in Reaches 2 and 3 with reference to pink salmon spawning in odd years. In the N:P figure, values above the top dashed line show potential P-limitation of algal growth; values below the bottom dashed line show potential N-limitation; values in between the dashed lines indicate potential co-limitation by N and P of algal growth.**

The variation in DIN and SRP concentration produced shifts in potential limitation of benthic algal growth based on molar N:P ratios at different times and places (Figure 3.13). During Trial 0, the N:P was 20 - 30 in Reaches 3 and 2, the only reaches monitored at that time, showing co-limitation by N and P. During and after Trial 1, molar N:P remained about the same in Reach 3, again showing co-limitation by N and P. In contrast, N:P in Reach 2 increased during and after Trial 2, which showed a time course increase in potential phosphorus deficiency for algal growth. With the presence of pink salmon in 2021, the molar N:P declined sharply, showing mostly N limitation caused by more phosphorus enrichment than N enrichment. Exceptionally high molar N:P in the Yalakom River showed greater phosphorus deficiency for algal growth than in the Lower Bridge River (Figure 3.13).

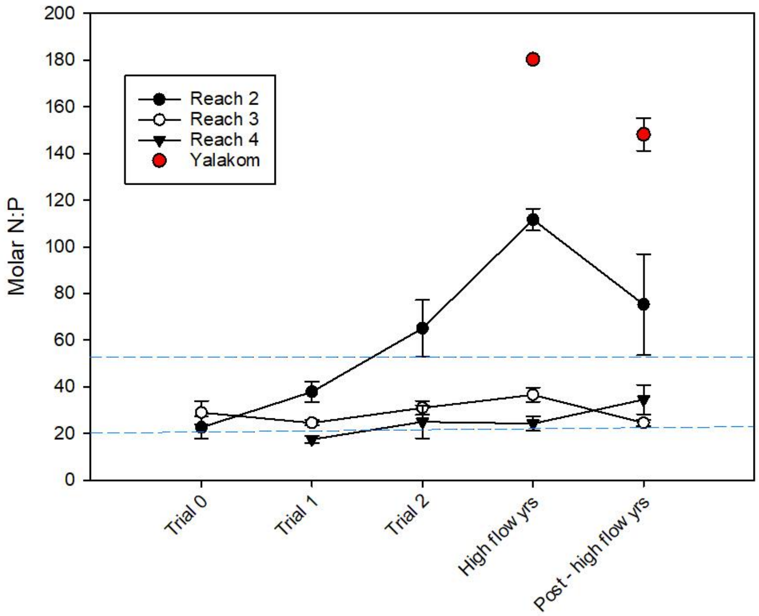


Figure 3.13 Mean molar N:P (±standard error) by Reach among trials. N:P values above the top dashed line show potential P-limitation of algal growth. Values below the bottom dashed line show potential N-limitation. Values in between the dashed lines indicate potential co-limitation by N and P of algal growth.

3.2. Periphyton

Periphyton taxa were almost entirely diatoms with low numbers of cyanobacteria (commonly known as blue green algae), Chlorophyta (green algae), and chryso-cryptophytes (flagellates) (Figure 3.14). Small counts of Euglenozoa (Euglena sp.) (Deuteromycota (fungi), Miozoa (dinoflaggelates), and Ochrophyta (golden brown algae) were combined into an “Other” category in Figure 3.14. The flagellates (Euglena sp. and chryso-cryptophytes) were likely entrained in water released from Carpenter Reservoir. Most abundant diatom genera included *Achnanthidium, Diatoma, Encyonema, Eunotia, Gomphonema, Nitzschia, Fragilaria, Melosira, Cymbella, Cocconeis, Synedra, Tabellaria, Staurosira, and Rossithidium.* There were 26 less common genera. The cyanobacteria genera included *Pseudanabaena, Oscillatoria, Anabaena, Aphanizomenon, Aphanocapsa, Arthrospira, Gloeocapsa, Merismopedia, Rhabdoderma*, and *Lyngbya*.

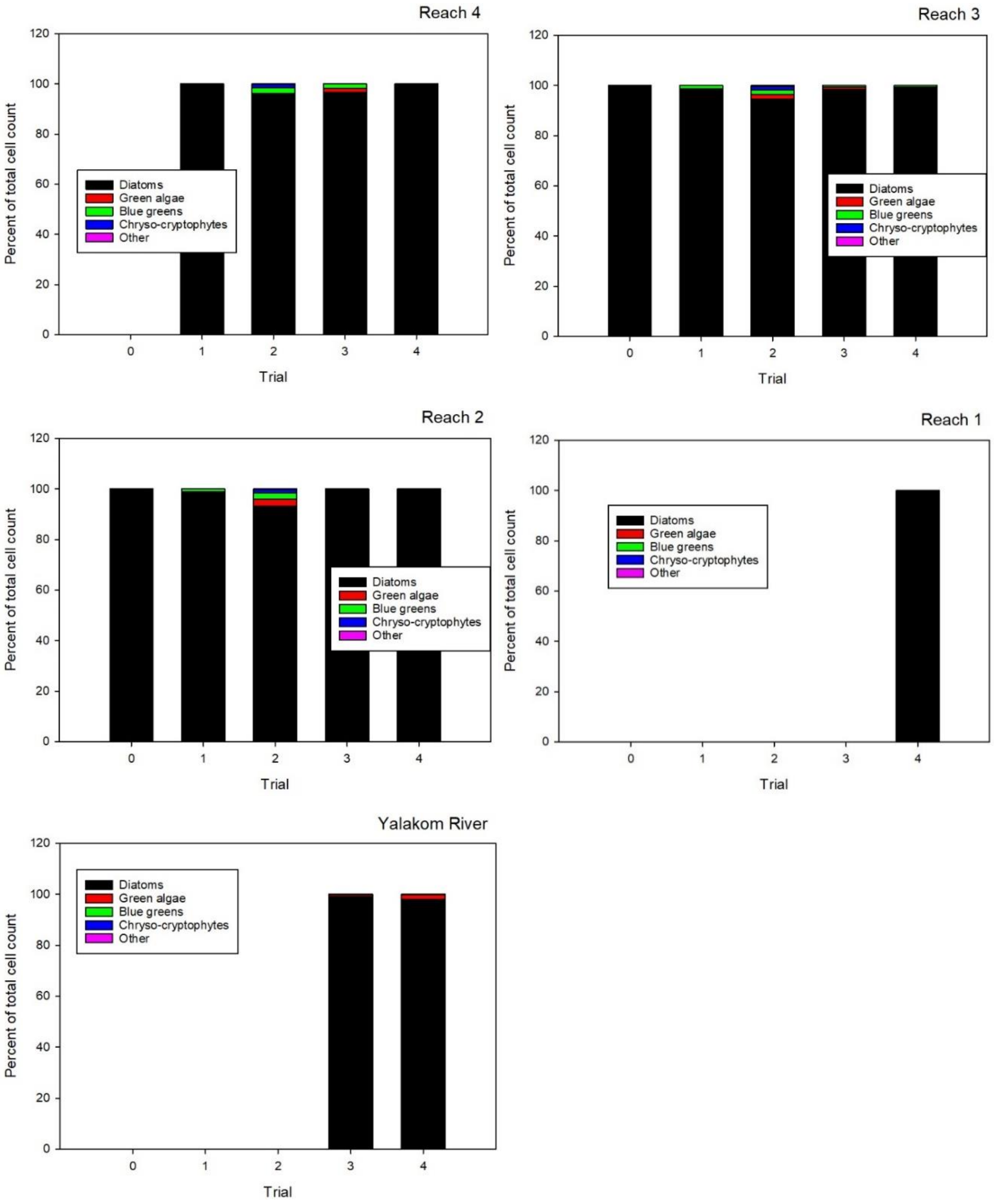


Figure 3.14 Algal divisions expressed as percent of total cell densities by Trial and Reach in the Lower Bridge River and at sites in the Yalakom River.

PB values were normally distributed (Shapiro-Wilks test) so no transformations were applied prior to running the ANOVA to test for Trial, Reach, and Pink effects on PB. Output is shown in Table 3.8. A significant interaction of Pink and Reach effects on PB (p=0.04) showed that presence of spawning Pink salmon produced greater PB than did absence of Pinks in Reach 2, but not in Reach 3 and Reach 4 (p=0.04, Figure 3.15). Furthermore, there was no relationship between PB and either SRP concentration, DIN concentration, or molar N:P among Reaches 2 through 4 in all years since study inception (Figure 3.17). This outcome showed that the range of nutrient concentrations was not enough to exceed the effect of other factors driving PB and produce an effect on PB. Those other factors did not include temperature because it also had no relationship with PB (Figure 3.17). Only the nutrient load from Pink salmon in Reach 2 was enough to produce a PB response over background. No significant interaction between Trial and the other factors meant the effect of Trial on PB could be examined independently (Figure 3.16). PB was greater during Trial 2 than during Trial 1 (Tukey test p=0.049) and PB during Post-high flow years was greater than during Trial 1 (Tukey test p=0.001) among all reaches regardless of the presence or absence of spawning Pink salmon. There was no significant difference in PB between the high flow years and the other Trial years (Tukey test p≥0.2). The overall Trial effect on PB showed a significant time course increase in PB from Trial 1 through the Post-high flow years.

Table 3.8 PB ANOVA table showing partitioning of variance between main effects (Trial, Reach, Pink) and residual variance (error). P values less than 0.05 showed a significant effect.

Source	Sums of squares	Degrees of freedom	Mean Squares	F-Value	p-Value
Trial	232	3	77.3	7.1	0.001
Reach	165	2	82.4	7.5	0.002
Pink	48.9	1	48.9	4.5	0.04
Trial*Reach	90.1	6	15.0	1.4	0.25
Trial*Pink	70.7	3	23.6	2.2	0.11
Reach*Pink	78.9	2	39.5	3.6	0.04
Trial*Reach*Pink	72.9	6	12.1	1.1	0.37
Error	458	42	10.9		

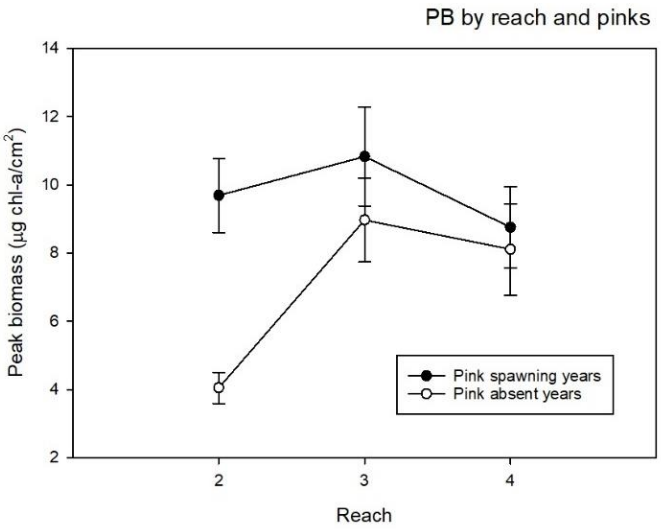


Figure 3.15 Mean periphyton peak biomass ± standard error of arithmetic mean by presence/absence of Pink spawning and Reach among all Trials. Years were replicates for calculation of the statistics.

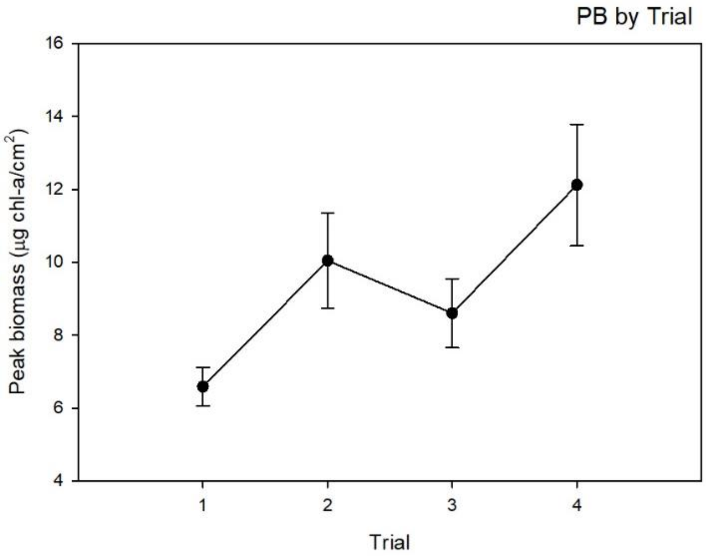


Figure 3.16 Mean periphyton peak biomass ± standard error of arithmetic mean by Trial among all Reaches.

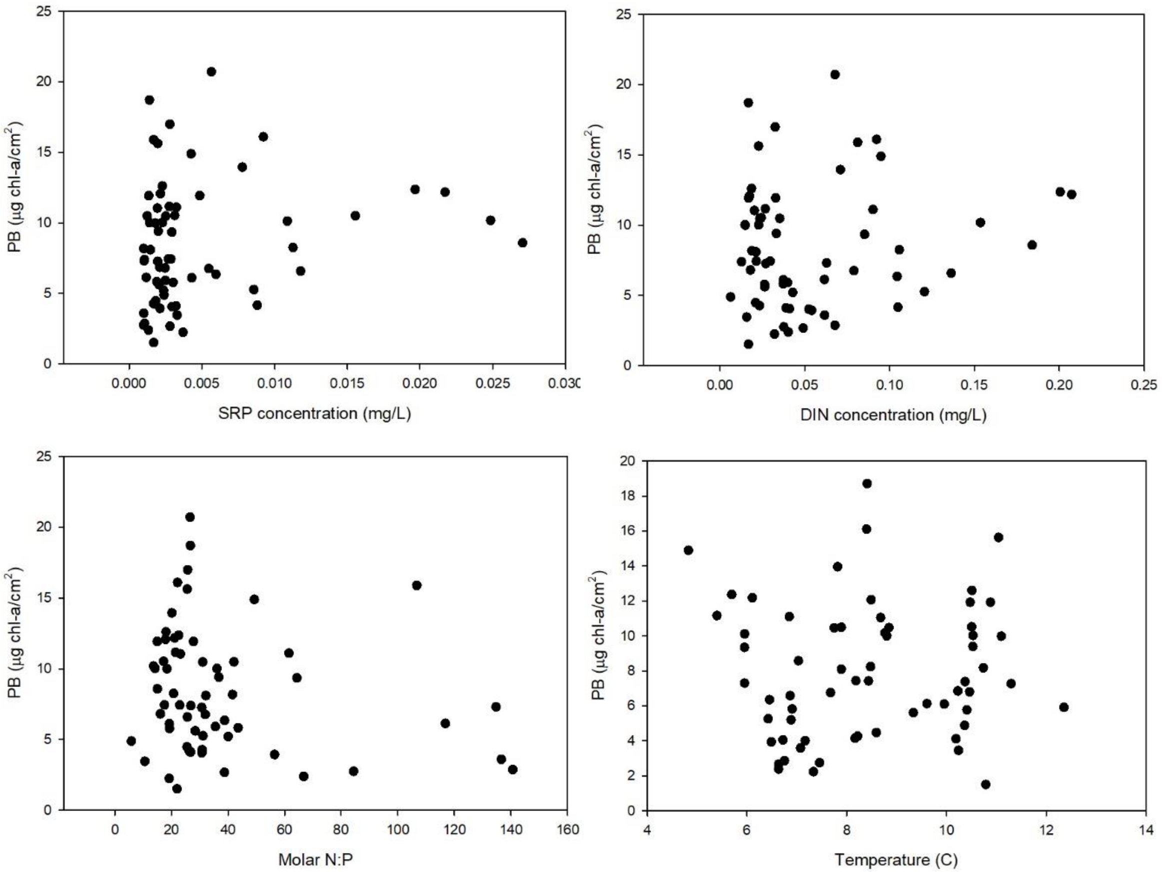


Figure 3.17 **Scatterplots of PB versus mean SRP concentration (top left), mean DIN concentration (top right), average molar N:P (bottom left), and mean temperature (bottom right) during the fall sampling periods among all years and reaches.**

Periphyton biomass was measured in Reach 1 only during the Post-High Flow years. Table 3.9 shows mean PB with standard errors for Reach 1 with a comparison to the Yalakom River and Reach 2. If we double the standard errors to arrive at approximately the 95% confidence interval, mean PB in Reach 1 was approximately the same as in Reach 2. This biomass was approximately 10 times greater than upstream in the Yalakom River at the same time.

Table 3.9 **Mean PB during Post-High Flow Years ± standard error in the Yalakom River and Reaches 2 and 1 of the Lower Bridge River.**

Reach	Mean PB during Post-High Flow Years ± standard error
Yalakom River	1.2 ± 0.29
Lower Bridge River Reach 2	9.0 ± 3.5
Lower Bridge River Reach 1	14.3 ± 4.7

3.3. Benthic Invertebrates

3.3.1. Trial, Reach, and Pink effects on benthos

Benthic invertebrates in the Lower Bridge River included Ephemeroptera (mayflies), Plecoptera (stoneflies), Tricoptera (caddisflies), Diptera (true flies, including chironomids), and “Other” taxa including Hemiptera, Podocopida, Coleoptera, Gordea, Haplotaxida, Tricladida, Basommatophora, Taeniopterygidae, terrestrial taxa from the riparian zone, and zooplankton from Carpenter Reservoir (Figure 3.18). Mean density of all taxa in the basket samplers was 71,000 animals·m^{-2} in Trial 0, 95,000 animals·m^{-2} in Trial 1, 94,000 animals·m^{-2} in Trial 2, a decline of 69% to 29,000 animals·m^{-2} in High Flow Years, and a subsequent increase to 85,000 animals·m^{-2} in the Post-High Flow years (Figure 3.19). Yalakom River samples included the same taxa found in the Lower Bridge River (Figure 3.18) with a mean invertebrate density of 47,625 animals·m^{-2} in 2018 (only one year of data available during High Flow Years) and 47,000 animals·m^{-2} in Post-High Flow Years. Mean invertebrate density in Reach 1 during the Post-High Flow years was 47,000 animals·m^{-2}. Taxa in Reach 1 were similar to those in the other Bridge River reaches.

There were no significant interactions among all combinations of Reach, Trial, and Pinks on total benthos (Table 3.10), EPT (Table 3.11), or chironomids (Table 3.12), which meant the main factors could be examined independently. Total benthos, EPT, and chironomid density were significantly lower during the High Flow Years compared to during the other Trials ($p<0.001$ to 0.007). There was no Reach effect on total benthos density among all Trials, regardless of presence or absence of Pinks ($p=0.528$), largely due to the same outcome on EPT ($p=0.175$). In contrast, chironomid density was lower in Reach 3 than in the other reaches among all Trials, again regardless of presence or absence of Pinks ($p=0.01$). Chironomid density was greater when Pink salmon spawners were present compared to when they were absent ($p=0.01$) among all Trials and Reaches but the EPT were not affected by Pinks ($p=0.64$).

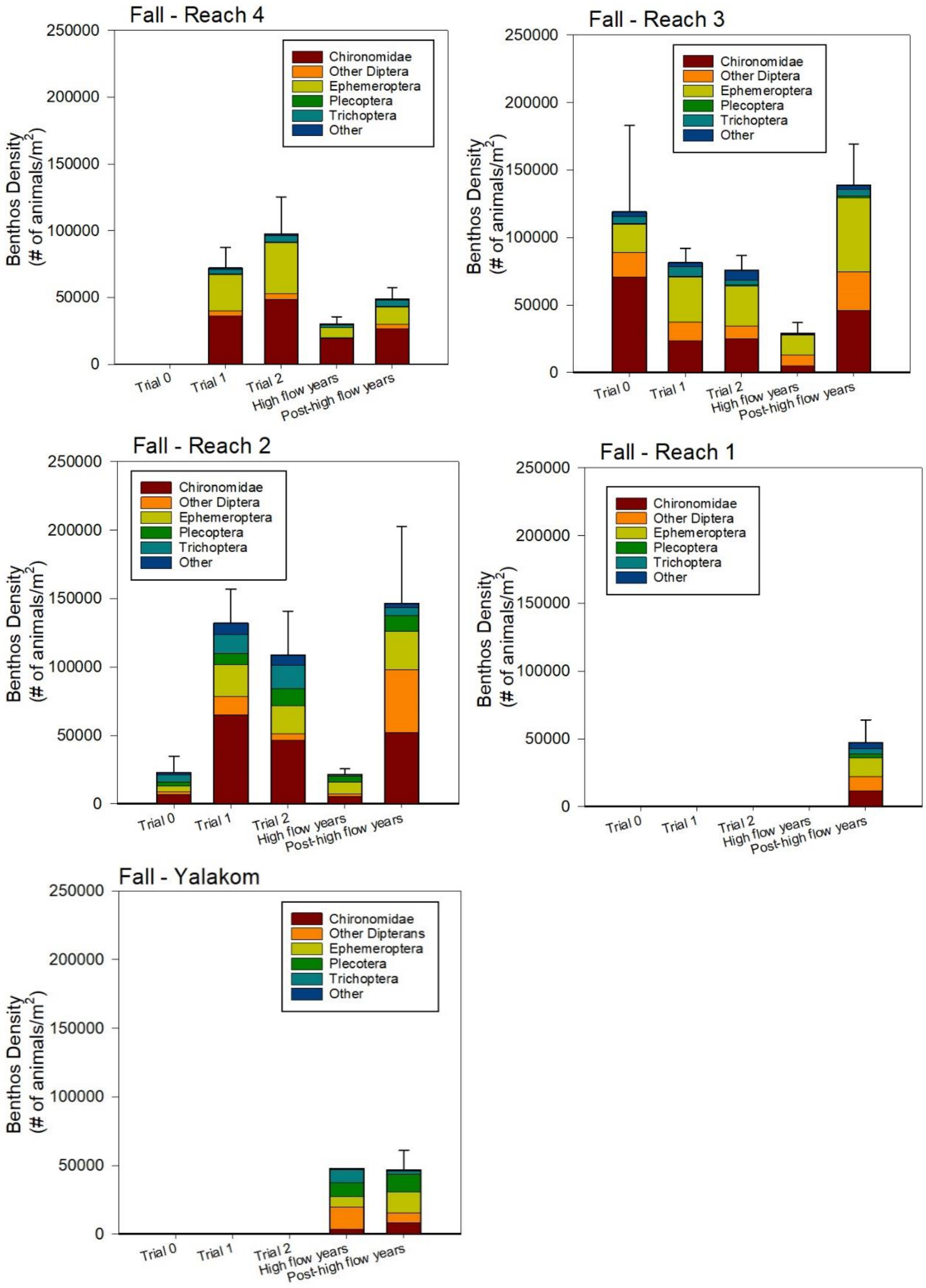


Figure 3.18 **Mean density (years are replicates) of invertebrate orders or other groups in the fall among Trials in Reach 4 (top left), Reach 3 (top right), Reach 2 (middle left), Reach 1 (middle right), and the Yalakom River (bottom left). Error bars are standard error of the arithmetic mean of total density.**

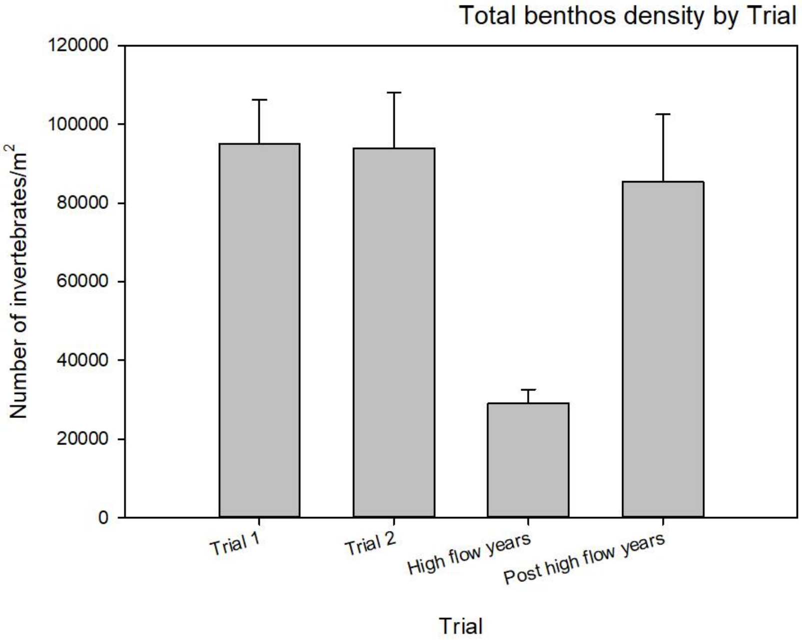


Figure 3.19 **Mean total benthos density (±standard error of arithmetic mean) by Trial. Invertebrate density during the high flow years was significantly less than during the other Trials when densities were similar ($p<0.001$).**

Table 3.10 **ANOVA table for test of Trial, Reach, and Pink effects on total benthos. Where a factor showed a significant effect, results of the Tukey test of pairwise contrasts are shown to indicate what factor levels were significantly different from the others ($p<0.05$).**

Analysis of variance statistics (Log_{10} total benthos+1)						Tukey test
Source	SS	df	Mean Squares	F-Ratio	p-Value	
Trial	1.801	3	0.600	7.815	<0.001	High Flow Years<Trial 1=Trial 2= Post-High Flow Years
Reach	0.100	2	0.050	0.651	0.528	Not applicable
Pink	0.072	1	0.072	0.944	0.338	Not applicable
Trial*Reach	0.341	6	0.057	0.740	0.621	Not applicable
Trial*Pink	0.380	3	0.127	1.649	0.195	Not applicable
Reach*Pink	0.077	2	0.038	0.501	0.610	Not applicable
Trial*Reach*Pink	0.120	6	0.020	0.260	0.952	Not applicable
Error	2.765	36	0.077			

Table 3.11 ANOVA table for test of Trial, Reach, and Pink effects on Ephemeroptera + Plecoptera + Tricoptera (EPT). Where a factor showed a significant effect, results of the Tukey test of pairwise contrasts are shown to indicate what factor levels were significantly different from the others (p<0.05).

Analysis of variance statistics (Log_{10} EPT+1)						Tukey test
Source	SS	df	Mean Squares	F-Ratio	p-Value	
Trial	1.496	3	0.499	4.762	0.007	High Flow Years<Trial 1=Trial 2= Post-High Flow Years
Reach	0.383	2	0.191	1.828	0.175	Not applicable
Pink	0.023	1	0.023	0.218	0.643	Not applicable
Trial*Reach	0.111	6	0.019	0.177	0.981	Not applicable
Trial*Pink	0.311	3	0.104	0.990	0.408	Not applicable
Reach*Pink	0.232	2	0.116	1.106	0.342	Not applicable
Trial*Reach*Pink	0.354	6	0.059	0.563	0.756	Not applicable
Error	3.770	36	0.105			

Table 3.12 ANOVA table for test of Trial, Reach, and Pink effects on Chironomidae. Where a factor showed a significant effect, results of the Tukey test of pairwise contrasts are shown to indicate what factor levels were significantly different from the others (p<0.05).

Analysis of variance statistics (Log_{10} Chironomidae+1)						Tukey test
Source	SS	df	Mean Squares	F-Ratio	p-Value	
Trial	2.526	3	0.842	6.850	0.001	High Flow Years<Trial 1=Trial 2= Post-High Flow Years
Reach	1.263	2	0.631	5.135	0.011	Reach 3<Reach 4= Reach 2
Pink	0.899	1	0.899	7.314	0.010	Pink on years > Pink off years
Trial*Reach	1.355	6	0.226	1.837	0.119	Not applicable
Trial*Pink	0.887	3	0.296	2.405	0.083	Not applicable
Reach*Pink	0.165	2	0.083	0.671	0.517	Not applicable
Trial*Reach*Pink	0.330	6	0.055	0.447	0.842	Not applicable
Error	4.426	36	0.123			

Family richness responded differently (Table 3.13). Richness during Pink-on years was greater than during Pink-off years in Trial 1 but not in the later Trials among all Reaches (Figure 3.20). Richness also increased significantly with distance from the dam among all Trials regardless of presence or absence of Pinks (p<0.001, Figure 3.21).

Table 3.13 ANOVA table for test of Trial, Reach, and Pink effects on benthic invertebrate Family Richness. Where a factor showed a significant effect, results of the Tukey test of pairwise contrasts are shown to indicate what factor levels were significantly different from the others (p<0.05).

Analysis of variance statistics (Family richness)						Tukey test
Source	**SS**	**df**	**Mean Squares**	**F-Ratio**	**p-Value**	
Trial	116.179	3	38.726	12.643	<0.001	Not applicable due to significant interaction with Pink effect (see Figure 3.20)
Reach	173.335	2	86.668	30.532	<0.001	Reach 2 >Reach3>Reach 4
Pink	0.874	1	0.874	0.308	0.582	Not applicable
Trial*Reach	14.084	6	2.347	0.827	0.557	Not applicable
Trial*Pink	28.531	3	9.510	3.350	0.030	Not applicable
Reach*Pink	0.040	2	0.020	0.007	0.993	Not applicable
Trial*Reach*Pink	4.163	6	0.694	0.244	0.958	Not applicable
Error	102.189	36	2.839			

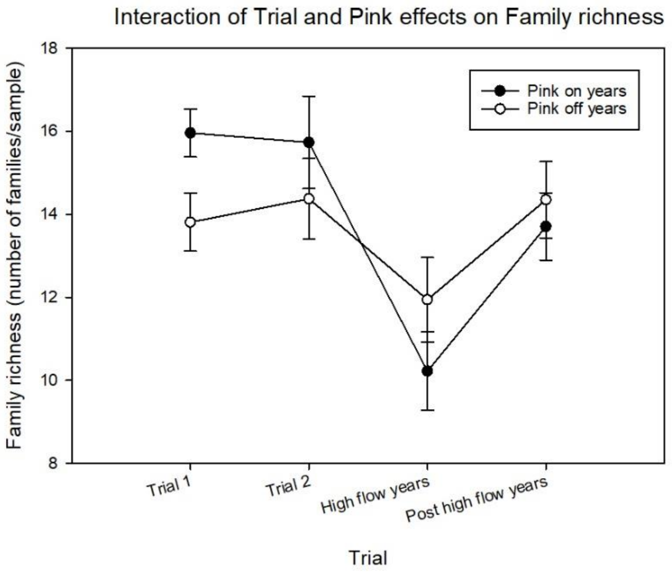


Figure 3.20 Interaction of Trial and Pink effects on benthos family richness among all reaches.

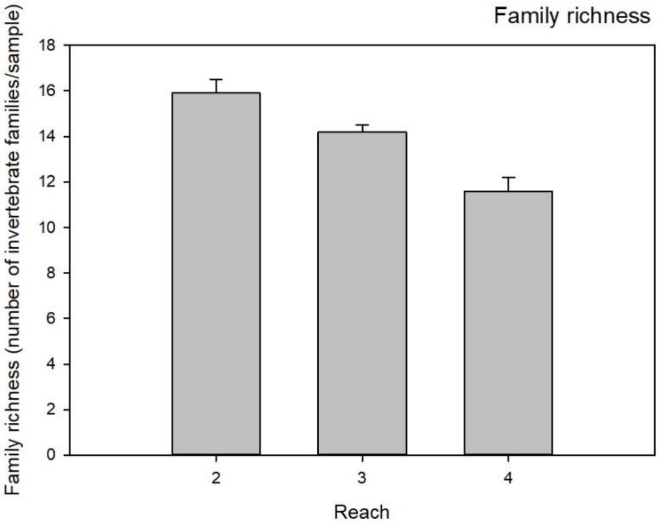


Figure 3.21 Change in invertebrate family richness by Reach among all trials regardless of presence or absence of Pink salmon spawning. Increasing richness from Reach 4 (closest to the dam) to Reach 2 (furthest from the dam and affected by inflow from the Yalakom River) was significant ($p<0.001$).

Benthic invertebrates were sampled in Reach 1 only during the Post-High Flow years. Table 3.14 shows mean densities with standard errors for Reach in those years with a comparison to invertebrate metrics in the Yalakom River and Reach 2. If we double the standard errors to arrive at approximately the 95% confidence interval, mean total and EPT density in Reach 1 was approximately the same as in Reach 2 and the Yalakom River. Chironomid density was greater in Reach 2 than in the Yalakom River and Reach 1. Using the same approach for family richness, Reach 1 and 2 richness was about the same as in the Yalakom River.

Table 3.14 Mean benthic invertebrate abundance during Post-High Flow Years ± standard error in the Yalakom River and Reaches 2 and 1 of the Lower Bridge River.

Reach	Mean invertebrate abundance in Post-high flow years ± standard error (number/m^2)			Family richness (number of families per sample) ± standard error
	Total abundance	EPT	Chironomids	
Yalakom River	46,680 ± 14,576	30,519 ± 7,509	8,147 ± 2,433	16 ± 1.3
Lower Bridge River Reach 2	146,390 ± 56,398	45,488 ± 17,309	52,283 ± 20529	16 ± 0.3
Lower Bridge River Reach 1	46,955 ± 17,081	20,323 ± 8,683	11,759 ± 3,036	14 ± 0.7

3.3.2. Testing a benthos response to nutrient addition

Presence of a Pink spawner effect on several of the benthos metrics triggered the BACI analysis to examine response by benthos to nutrient enrichment. The nutrients were NH_4-N and SRP concentrations coming from Pink salmon spawning and they were indicators of other nutrients that would have been released during decomposition of Pink carcasses (Figure 3.12). Reach 2 was the treatment reach where highest nutrient concentrations resulting from spawners among Pink-on years were found. Reach 4 was the control reach having small differences in nutrient concentrations between Pink-on and Pink-off years, thus showing few or no spawners using that reach. There were six “After” years having NH_4-N and SRP concentrations that were an order of magnitude greater than in Pink-off years and indicated presence of Pink salmon spawners in Reach 2 (2003, 2005, 2009, 2011, 2013, 2021). The high nutrient concentrations did not come from tributaries and were attributed to processes within the main river channel (Section 3.1.7). The only source was spawning salmon. There were six corresponding “Before” and even years when there were no Pink salmon runs (2002, 2004, 2008, 2010, 2012, 2018). The six years were replicates for the ANOVA part of the BACI analysis.

The ANOVA was significant (p=0.007), which showed a larger difference between Reach 2 and Reach 4 total benthos densities in the “After” years (presence of Pink salmon in Reach 2) than in the “Before” years (absence of Pink salmon in Reach 2). The ANOVA was run on untransformed data because the Shapiro-Wilk test for normality was not significant (p=0.75).

The response is shown in Figure 3.22. It shows no difference in benthos density between Reach 2 and Reach 4 in the “Before” years when Pink salmon were absent and N and P concentrations were low in both reaches (Table 3.15). Density doubled in Reach 2 with added NH_4-N, NO_3-N, and SRP concentrations from Pink spawning relative to change in the control, Reach 4. This doubling of invertebrate density was produced from a 24-times increase in NH_4-N concentration, a doubling of NO_3-N concentration, and a 10-times increase in SRP concentration (Table 3.15). NO_3-N is included in these comparisons because some of the NH_4-N that is released from Pink spawning and decomposition of carcasses will be oxidized via nitrification to NO_3-N. Molar N:P ratios show that growth of the algal community before nutrient enrichment was strongly N limited in the control reach (Reach 4) and co-limited by N and P in the treatment reach (Reach 2). With nutrient enrichment, growth was more N-limited than P-limited in both of the control and treatment reaches. The load of P from Pink spawning produced a large demand for N by biological production.

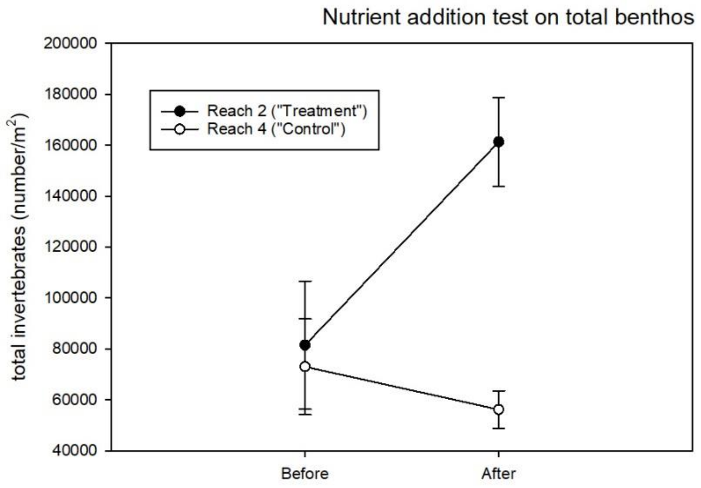


Figure 3.22 Total benthos density response to nutrient addition from Pink salmon using BACI layout.

Table 3.15 Mean concentration of NH_4-N, NO_3-N, and SRP ± standard deviation with molar N:P ratios without nutrient addition from Pink salmon spawning (Before) and with nutrient addition from Pink salmon spawning (After) in Reach 4 (Control) and Reach 2 (Treatment).

Reach	Molecule or compound or molar ratio	Concentration by time block (µg·L^{-1})	
		Before (n=6)	After (n=6)
Control	NH_4-N	3.4 ± 2.8	11.7 ± 10.9
Treatment	NH_4-N	4.8 ± 2.7	120 ± 128
Control	NO_3-N	14.3 ± 7.2	15.7 ± 6.2
Treatment	NO_3-N	36.7 ± 13.8	72.9 ± 35
Control	SRP	2.4 ± 0.8	3.1 ± 1.5
Treatment	SRP	2.0 ± 0.9	19.6 ± 24.3
Control	N:P	16	20
Treatment	N:P	46	22

3.4. Juvenile Fish Production

3.4.1. Size and Condition

Mean weight of mykiss fry (Age-0+) in all reaches was highest during the High Flow period (2016-2018) compared to the other treatment periods (Figure 3.23). This likely occurred for a few possible reasons: 1) reduced density (see Figure 3.27 in Section 3.4.2) which reduced competition for available food; 2) warmer temperatures during the summer rearing period (see Figure 3.7) which may have facilitated growth; or 3) the high flows selected for larger fish since they are more mobile and capable of competing for habitat space, while smaller fish may be more readily displaced downstream. Mean weight during the Post-high flow years (2019-2021) was also high, most likely due to reasons 1) and 2) above, which have continued to some extent, but also due to recovered benthic invertebrate abundance (see Figure 3.19 in Section 3.3) after three years of a return to lower (i.e., at or near Trial 2) peak flows. Growth in reach 3 during the Trial 0 pre-flow period (0 $m^3 \cdot s^{-1}$) was also high likely due to ample benthic invertebrate abundance, combined with the quality rearing conditions in this reach prior to the flow release. Size of mykiss fry among the reaches was fairly equivalent within each flow treatment period.

Mean size for mykiss parr (Age-1+) was also greatest during the High flow and Post-high flow periods. Mykiss parr tended to be largest in reaches 3 and 4 during the High flow and Post-high flow years, whereas they had been more equivalent among the reaches during the previous treatment periods. However, these results for mykiss parr were based on only two years available for both the High flow and Post-High flow treatment periods (see Table 2.3 and rationale provided in Section 2.8.4).

Patterns in mean weight for coho fry across flow treatments in reaches 2, 3 and 4 closely matched the patterns seen for mykiss fry which reflected increased size during the High flow and Post-high flow periods, and higher growth in Reach 3 during Trial 0, likely for the same set of reasons provided for mykiss fry, above. On average, the coho fry were consistently larger than the mykiss fry at the time of sampling due to their earlier emergence timing in the year (i.e., longer period of growth). Coho fry have typically been largest in Reach 4 among each of the flow treatments, except for the Pre-flow period when there was no flow in this reach. However, as with mykiss, there was also considerable overlap in the standard error bars among reaches and flow treatment periods in many cases.

In Reach 2, mean weight of Chinook fry was higher under the Trial 2 (6 $m^3 \cdot s^{-1}$), High flow and Post-high flow years relative to the Trial 0 (0 $m^3 \cdot s^{-1}$) and Trial 1 (3 $m^3 \cdot s^{-1}$) treatments, probably due to reduced densities. In reaches 3 and 4, mean weights were higher than in Reach 2 (for all flow treatments) and were highest under the High flow and Post-high flow treatments (and equivalent among them). Among the fry, Chinook were the largest of the target species and the size gradient among the reaches corresponded with the patterns of abundance and predicted emergence timing for Chinook which likely resulted in reduced competition and a longer growing period in reaches 3 and 4 than in Reach 2.

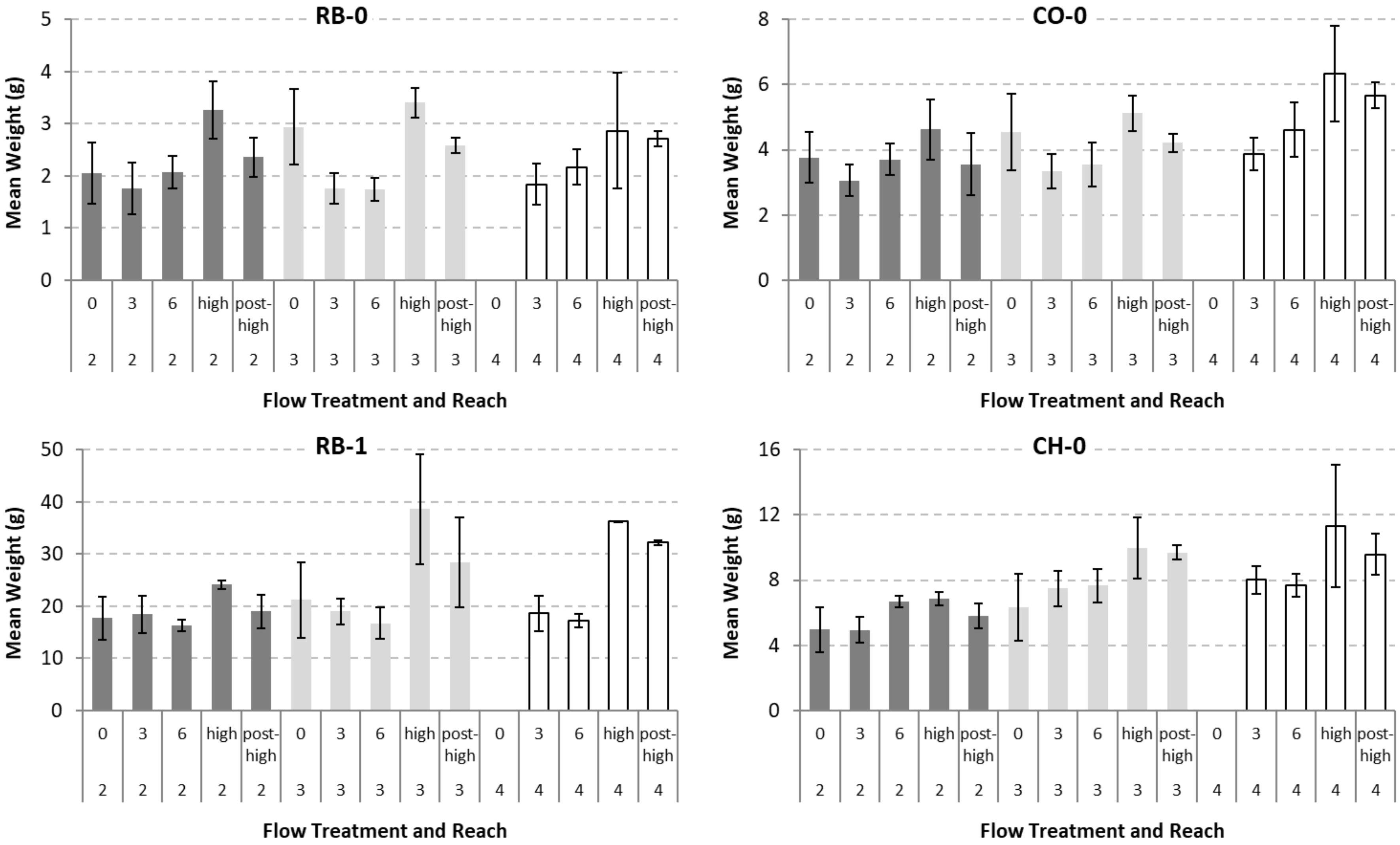


Figure 3.23 Mean juvenile salmonid weight during fall standing stock assessments across flow treatments (0, 3, and 6 $m^3 \cdot s^{-1}$ trials and the high and post-high flow periods) and reaches (2, 3, and 4). RB-0, RB-1, CO-0, and CH-0 denote mykiss fry, mykiss parr, coho fry, and Chinook fry, respectively. Height of bars represents the means of annual values for each reach-flow treatment combination and error bars denote 1 standard deviation (variation in annual values within treatments).

Given uncertainty in the factors driving differences in mean weight among the flow trials (particularly for the High flow years when food availability was dramatically reduced), we calculated mean condition factor values since this better reflects actual body condition of the fish sampled (i.e., expressed as Fulton's Condition Factor, K), rather than just size (Figure 3.24). Interestingly, the condition factor values showed a different pattern among flow trials than the mean weight data: For each target species and age class there has been a general trend of declining condition factor values across the study period (although condition factor still remains fairly good; K value >1). This was true for mykiss fry, coho fry and Chinook fry, particularly in reaches 2 and 3. Exceptions for the fry stage of each target species were in Reach 4 where condition factor has either not changed (i.e., for mykiss and coho), or highest K values were during the High flow years (i.e., for Chinook; however, this K value was also based on the smallest sample size in the analysis (*n*= 13 fish)). For mykiss parr, lowest K values were during the Post-high flow years (with only 2 year-replicates to-date), but highest K values for this age class were in flow trials 0 and 1 (with substantial overlap in standard error bars among flow trials 0 to 2). Condition factors in Trial 1 vs. Trial 2 were generally very similar for all species and age class combinations.

We also plotted condition factor values from 2019 to 2021 with the Trial 2 mean (± SD; based on 2011 to 2015 values) to compare relative values before and after the high flow years (Figure 3.25). These comparisons showed that, despite similar flow release volumes throughout the year (except moderately higher peak flows in 2021), K values in the Post-high flow years were generally within or below the Trial 2 lower standard deviation line for each species and age class in each reach. Exceptions were in Reach 3 for mykiss fry in 2020 and 2021, mykiss parr in 2019, and Chinook fry in 2019 and 2020, which were equivalent to the respective Trial 2 means, and in Reach 4 for mykiss fry in 2021, Chinook fry in 2019 and 2021, and coho fry in 2020 which had higher K values than the Trial 2 mean. Overall, for all reaches combined, K values were notably lower than Trial 2 values for mykiss fry (Age-0+), coho fry and Chinook fry in 2019, for mykiss parr (Age-1) and Chinook fry in 2020, and for mykiss parr, coho fry and Chinook fry in 2021. Condition factors in 2021 were equivalent to 2020 values for mykiss fry and parr, but was lower than the Trial 2 and other Post-high flow years for coho and Chinook fry.

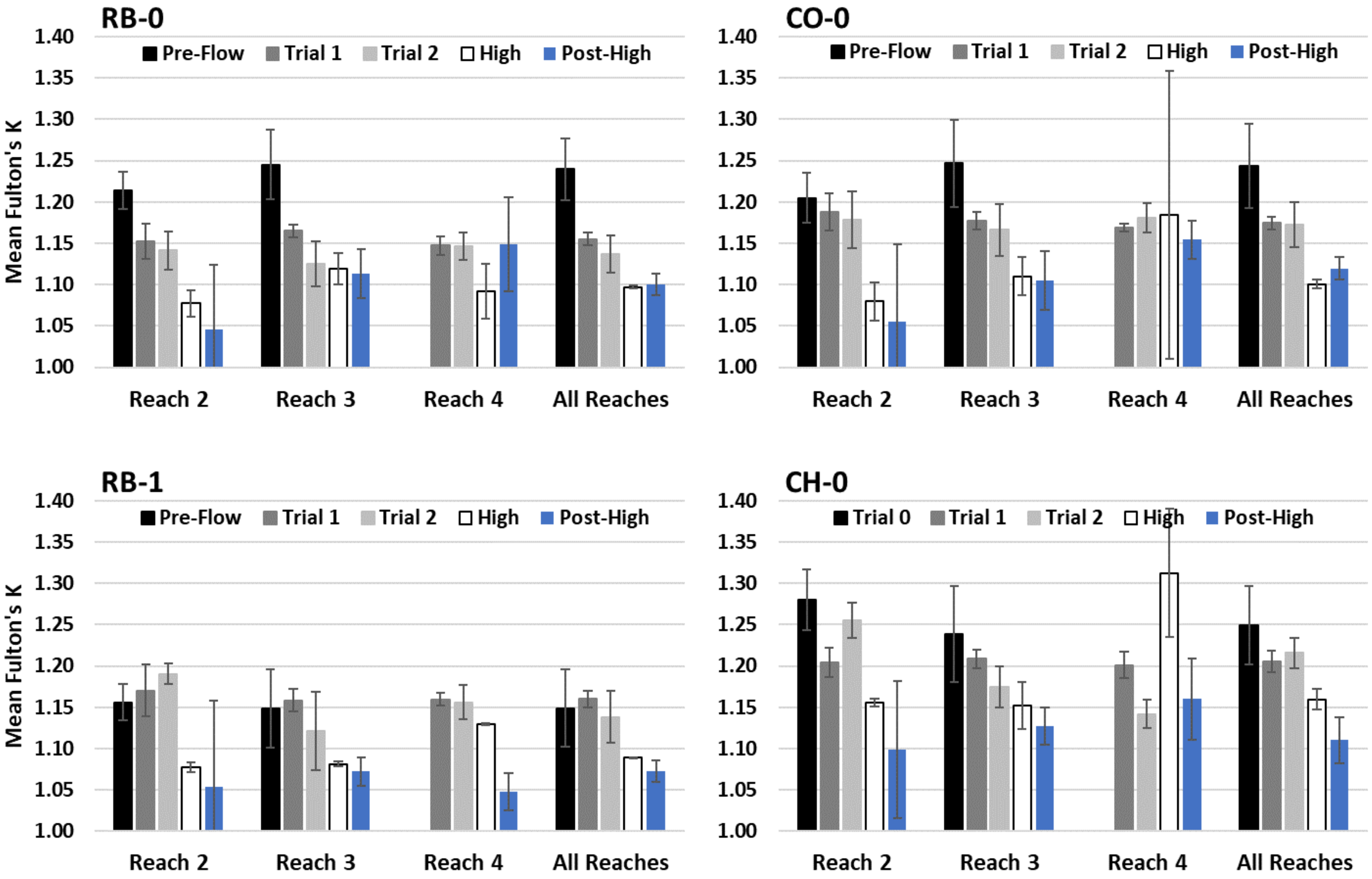


Figure 3.24 Mean condition factor for juvenile salmonids during fall standing stock assessments across flow treatments (0, 3, and 6 $m^3 \cdot s^{-1}$ trials and the high and post-high flow periods) and reaches (2, 3 and 4). Species and age designations are the same as described for Figure 3.23. Height of bars represents the means of annual values for each reach-flow treatment combination and error bars denote the standard error of the mean.

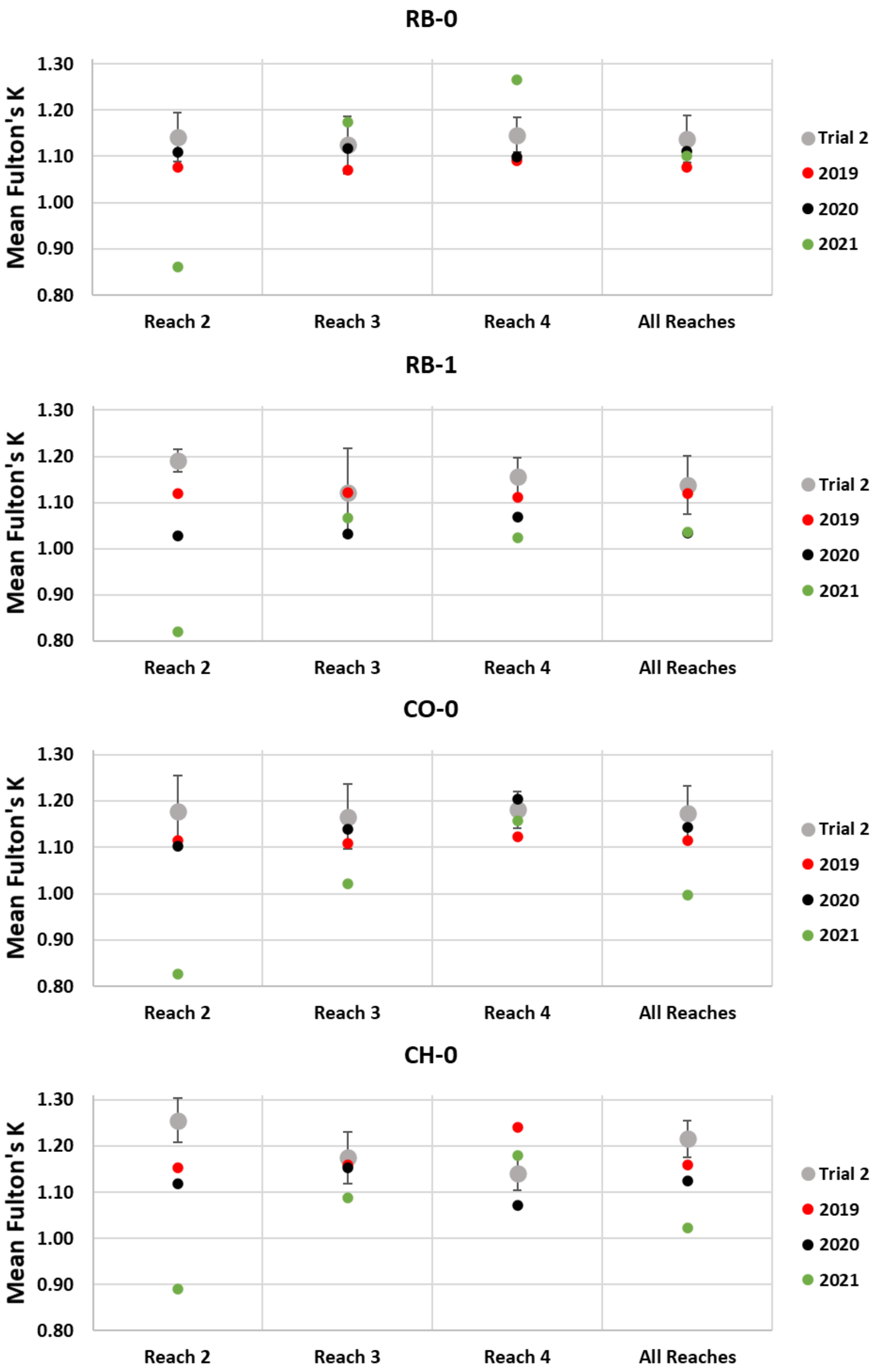


Figure 3.25 Mean condition factor (Fulton's K value; ±SE) between 2019, 2020, 2021 and all Trial 2 years for mykiss fry, mykiss parr, coho fry and Chinook fry in reaches 2, 3 and 4.

3.4.2. Abundance and Biomass

For all juvenile salmonid species combined, abundance and biomass increased from Trial 0 (the Pre-flow period) to Trial 1, decreased slightly in Trial 2 and then decreased substantially under high flows (Figure 3.26). Relative to the values from the earlier trials, abundance and biomass estimates in 2019-2021 were low (abundance and biomass were substantially lower than the Trial 2 average under similar flow release discharges). The abundance and biomass values in Figure 3.26 are shown relative to the average flow release discharge in spring-summer (i.e., May to July) since this was the period of the year when discharges varied the most among the trials (see Figure 1.3 in Section 1.3).

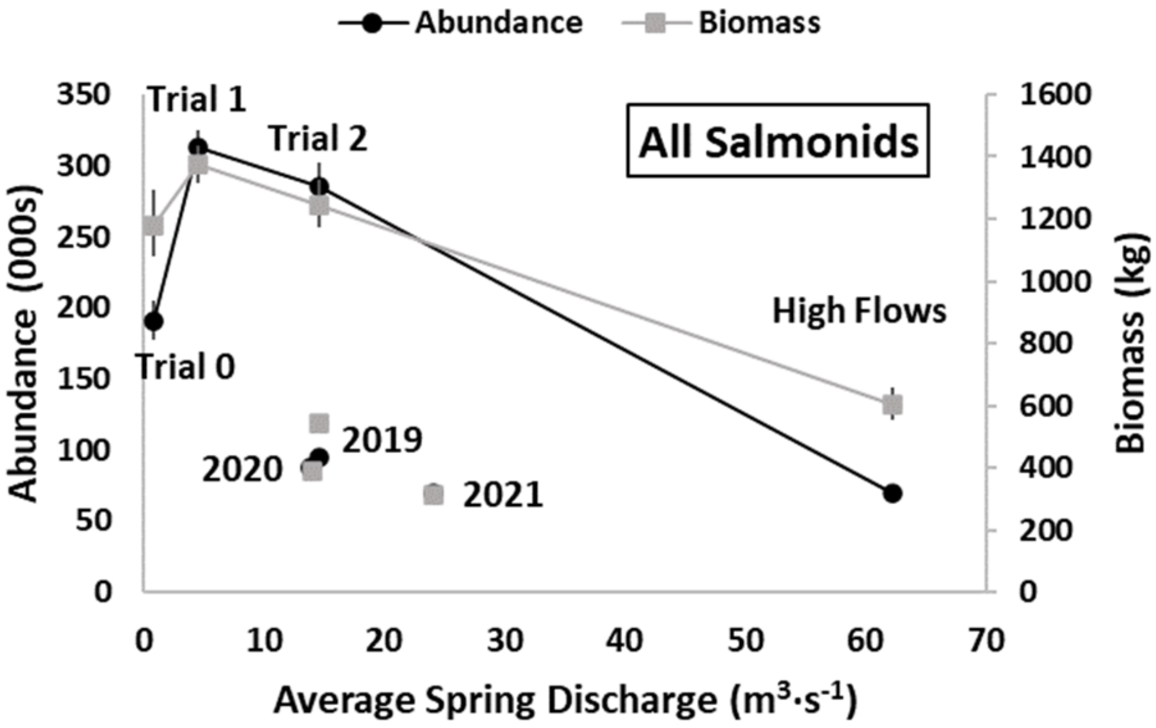


Figure 3.26 Juvenile salmonid abundance (1° y-axis) and biomass (2° y-axis) in the Lower Bridge River as a function of the average May-July flow releases from Terzaghi Dam for each flow treatment (with 95% credible intervals). The individual data points for 2019, 2020 and 2021 are also shown.

Among the flow treatments, fish production response to flow (abundance and biomass) varied by species and age class (Table 3.16, In TERMS of recovery post-high flows (i.e., in 2019 to 2021), all species and age classes have fared relatively poorly to-date. Since the return to lower peak flows (i.e., at or near the Trial 2 levels) following the High flow years, mykiss fry and coho fry abundance increased very modestly (+9K mykiss fry and +11K coho fry); however, the estimates for mykiss parr and Chinook fry actually decreased relative to the High flow period (-2K and -3K on average, respectively) (Table 3.16a and Table 3.17). For Biomass during the Post-high flow treatment, only coho fry increased, the other species and age classes all decreased relative to the High Flow years (Table 3.16b). The abundance and biomass estimates were fairly equivalent between 2019 and 2020 for all species/age classes and substantially lower than the Trial 2 estimates, particularly for mykiss (fry and parr) and coho fry (Figure 3.27). For Chinook, the 2019 and 2020 estimates were lower, but the Trial 2 estimates were already low (compared to the Pre-flow estimates) for this species. In 2021, with moderately higher peak flows than Trial 2, the abundance and biomass for mykiss fry and coho fry were lower than in 2019 and 2020, whereas mykiss parr and Chinook fry were in a similar range as those previous two Post-high flow years.

Table 3.17 and Figure 3.27). Mykiss fry and coho fry responded the most favourably to the initial flow release trials (Trial 1 and Trial 2) and, proportionately, decreased the most under the High flows. Mykiss parr were highest (and relatively equivalent) among the Pre-flow, Trial 1 and Trial 2 periods, and then decreased substantially under High flows. Chinook fry decreased during Trial 1 and remained low during Trial 2 and High flows.

Table 3.16 Mean total abundance (a, '000s) and biomass (b, kg) of juvenile salmonids in the Lower Bridge River across all reaches by flow treatment (year-specific results for 2019 to 2021 included (grey columns)). RB-0, RB-1, CO-0, and CH-0 denote Age-0+ mykiss, Age-1 mykiss, Age-0+ coho, and Age-0+ Chinook, respectively.

a) Abundance

Species-Age	Trial 0	Trial 1	Trial 2	High Flow	Post-High Flow			
					2019	2020	2021	Mean
RB-0	90	174	162	38	51	53	37	47
RB-1	36	35	33	10	8	6	9	8
CO-0	25	81	76	8	23	21	13	19
CH-0	38	22	13	13	12	7	11	10

b) Biomass

Species-Age	Trial 0	Trial 1	Trial 2	High Flow	Post-High Flow			
					2019	2020	2021	Mean
RB-0	249	305	282	124	141	125	73	113
RB-1	690	653	554	326	243	127	150	174
CO-0	108	281	255	39	89	94	32	72
CH-0	228	134	91	114	72	43	54	56

In terms of recovery post-high flows (i.e., in 2019 to 2021), all species and age classes have fared relatively poorly to-date. Since the return to lower peak flows (i.e., at or near the Trial 2 levels) following the High flow years, mykiss fry and coho fry abundance increased very modestly (+9K mykiss fry and +11K coho fry); however, the estimates for mykiss parr and Chinook fry actually decreased relative to the High flow period (-2K and -3K on average, respectively) (Table 3.16a and Table 3.17). For Biomass during the Post-high flow treatment, only coho fry increased, the other species and age classes all decreased relative to the High Flow years (Table 3.16b). The abundance and biomass estimates were fairly equivalent between 2019 and 2020 for all species/age classes and substantially lower than the Trial 2 estimates, particularly for mykiss (fry and parr) and coho fry (Figure 3.27). For Chinook, the 2019 and 2020 estimates were lower, but the Trial 2 estimates were already low (compared to the Pre-flow estimates) for this species. In 2021, with moderately higher peak flows than Trial 2, the abundance and biomass for mykiss fry and coho fry were lower than in 2019 and 2020, whereas mykiss parr and Chinook fry were in a similar range as those previous two Post-high flow years.

Table 3.17 Relative number of fish produced (by species and age class) under each flow treatment. Each value reflects production by the flow treatment in the column label relative to the flow treatment in the row label (1.0 = equivalent production). Matrix cells comparing Post-high flows vs. Trial 2 averages are highlighted yellow.

Species-Age Class		Flow Treatment (Mean Annual Release)			
		Trial 1 ($3\ m^3 \cdot s^{-1}$)	Trial 2 ($6\ m^3 \cdot s^{-1}$)	High Flows ($>18\ m^3 \cdot s^{-1}$)	Post-High Flows ($6\text{-}9\ m^3 \cdot s^{-1}$)
RB Age-0+	Pre-Flow	1.9	1.8	0.4	0.5
	Trial 1		0.9	0.2	0.3
	Trial 2			0.2	0.3
	High Flows				1.2
RB Age-1	Pre-Flow	1.0	0.9	0.3	0.2
	Trial 1		0.9	0.3	0.2
	Trial 2			0.3	0.2
	High Flows				0.8
CO Age-0+	Pre-Flow	3.2	3.0	0.3	0.8
	Trial 1		0.9	0.1	0.2
	Trial 2			0.1	0.2
	High Flows				2.4
CH Age-0+	Pre-Flow	0.6	0.3	0.3	0.3
	Trial 1		0.6	0.6	0.4
	Trial 2			1.0	0.8
	High Flows				0.8
All Salmonids	Pre-Flow	1.7	1.5	0.4	0.4
	Trial 1		0.9	0.2	0.3
	Trial 2			0.2	0.3
	High Flows				1.2

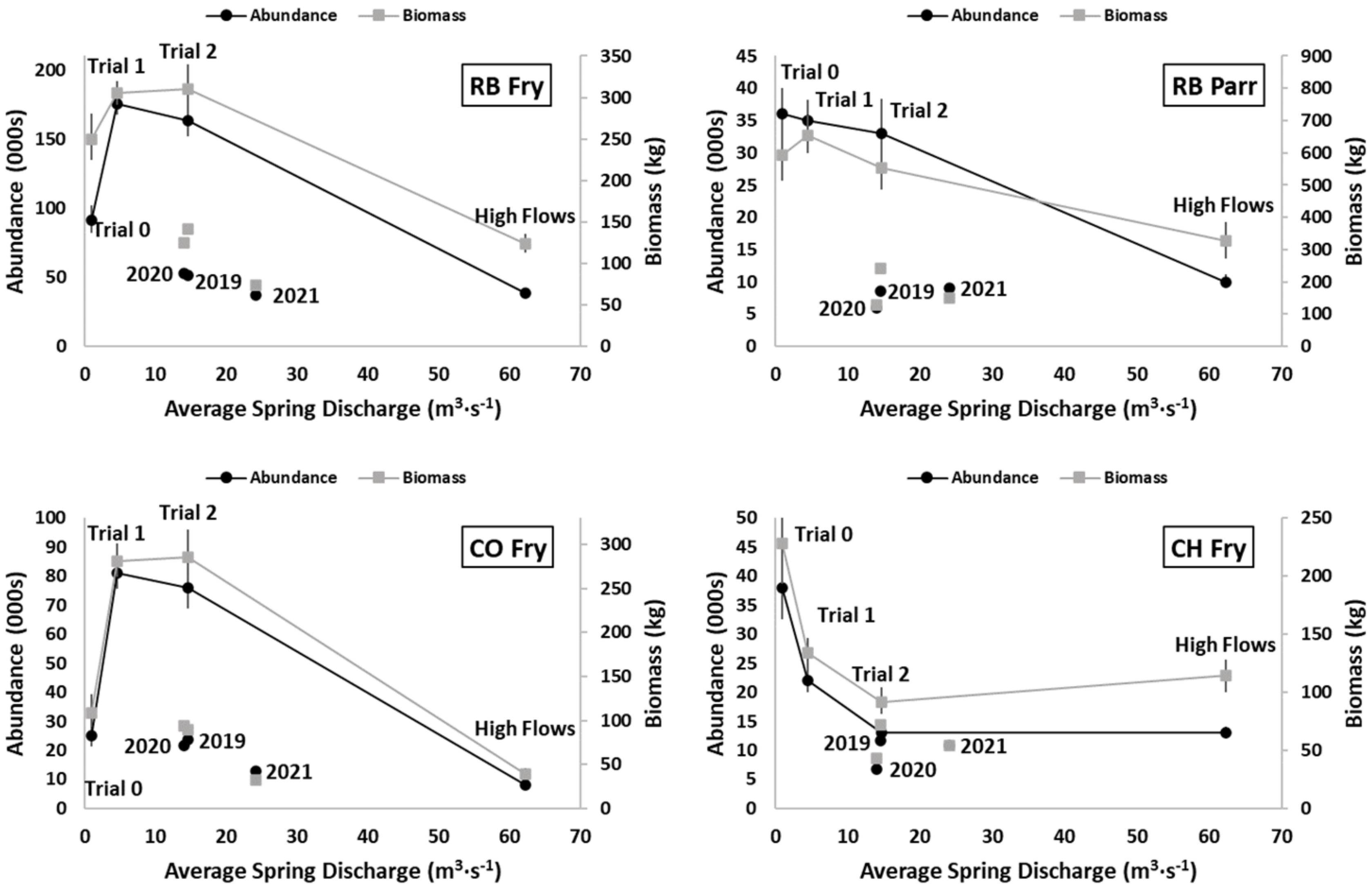


Figure 3.27 **Abundance (1° y-axis) and biomass (2° y-axis) for mykiss fry (top left), mykiss parr (top right), coho fry (bottom left) and Chinook fry (bottom right) as a function of the average May-July flow releases from Terzaghi Dam for each flow treatment (with 95% credible intervals). The individual data points for 2019 – 2021 are also shown on each plot.**

On a reach basis, increasing flow from Trial 0 (0 $m^3 \cdot s^{-1}$ release) to the Trial 1 (3 $m^3 \cdot s^{-1}$) treatment led to increases in abundance of mykiss fry in reaches 2 (+13K fish) and 3 (+24K fish) and there was substantial new production in Reach 4 (+47K fish)(Figure 3.28). Production under Trial 2 (6 $m^3 \cdot s^{-1}$) was nearly equivalent, with Reach 4 contributing a little less (-11K), Reach 3 contributing about the same (-4K), and Reach 2 contributing slightly more (+3K) than during Trial 1. Under high flows, when abundance was substantially reduced, all of the study reaches were affected. Mykiss fry production decreased by 33K, 72K and 21K relative to the Trial 2 abundances in reaches 4, 3 and 2, respectively. During the Post-high flow years (2019-2021), the recovery was strongest in Reach 3 (+7K on average relative to the High flow abundance), and more modest in reaches 4 (+1K) and 2 (no change). However, relative to the Trial 2 averages, the abundances for mykiss fry during the Post-high flow years (when flow releases were similar to Trial 2) were -31K, -64K, and -21K for reaches 4, 3 and 2, respectively.

Mykiss parr abundance increased by 1K in Reach 2 from Trial 0 to Trial 1 while there was a large decrease by 13K in Reach 3. However, Trial 1 also produced about 12K additional parr in Reach 4, which compensated for the losses in Reach 3. Under Trial 2, mykiss parr abundance increased in Reach 3 (by 4K relative to the Trial 1 abundance), decreased in Reach 4 (by 5K), and slightly decreased in Reach 2 (by 1K). For the study area as whole (i.e., reaches 2, 3 and 4 combined) the production of mykiss parr was equivalent for trials 0, 1 and 2. Under the high flows, the response of mykiss parr followed a similar pattern to the fry: Reaches 3 and 4 were hit the hardest (-20K and -5K relative to the Trial 2 abundance), whereas Reach 2 abundance increased slightly (by ~1K). During the Post-high flow period, mykiss parr abundance has been very similar to the high flow values in reaches 3 and 4, with a small drop (by 2K) in Reach 2. Relative to the Trial 2 years, there are ~21K fewer mykiss parr in Reach 3, ~6K fewer in Reach 4, and about the same production in Reach 2. Although it must be noted for these comparisons that the High flow and Post-high flow production is currently based on two years of data each (2017-2018, and 2020-2021, respectively) since 2016 and 2019 were transition years between flow treatments for this age class.

Coho fry abundance trends followed those for mykiss fry with increases in reaches 2 (+3K fish) and 3 (+32K) between Trial 0 and Trial 1 and substantial gains in Reach 4 (+21K), and little change in abundance under Trial 2 (i.e., -6K, +1K and +1K for reaches 4, 3 and 2, respectively) (Figure 3.28). Like the mykiss fry, coho fry abundance was substantially reduced by the high flows: -15K in Reach 4, -51K in Reach 3, and -3K in Reach 2. During the Post-high flow period to-date, coho fry production has increased modestly (i.e., +3K in Reach 4, +8K in Reach 3, and no change in Reach 2) relative to the abundances during the High flow years. However, these abundances are still substantively less than the Trial 2 production, particularly in reaches 3 and 4 which were -43K and -12K relative to the Trial 2 averages for these reaches. Post-high flow abundance in Reach 2 was 3K less than the Trial 2 average for this reach which has not been a big contributor to coho production during any of the flow treatments.

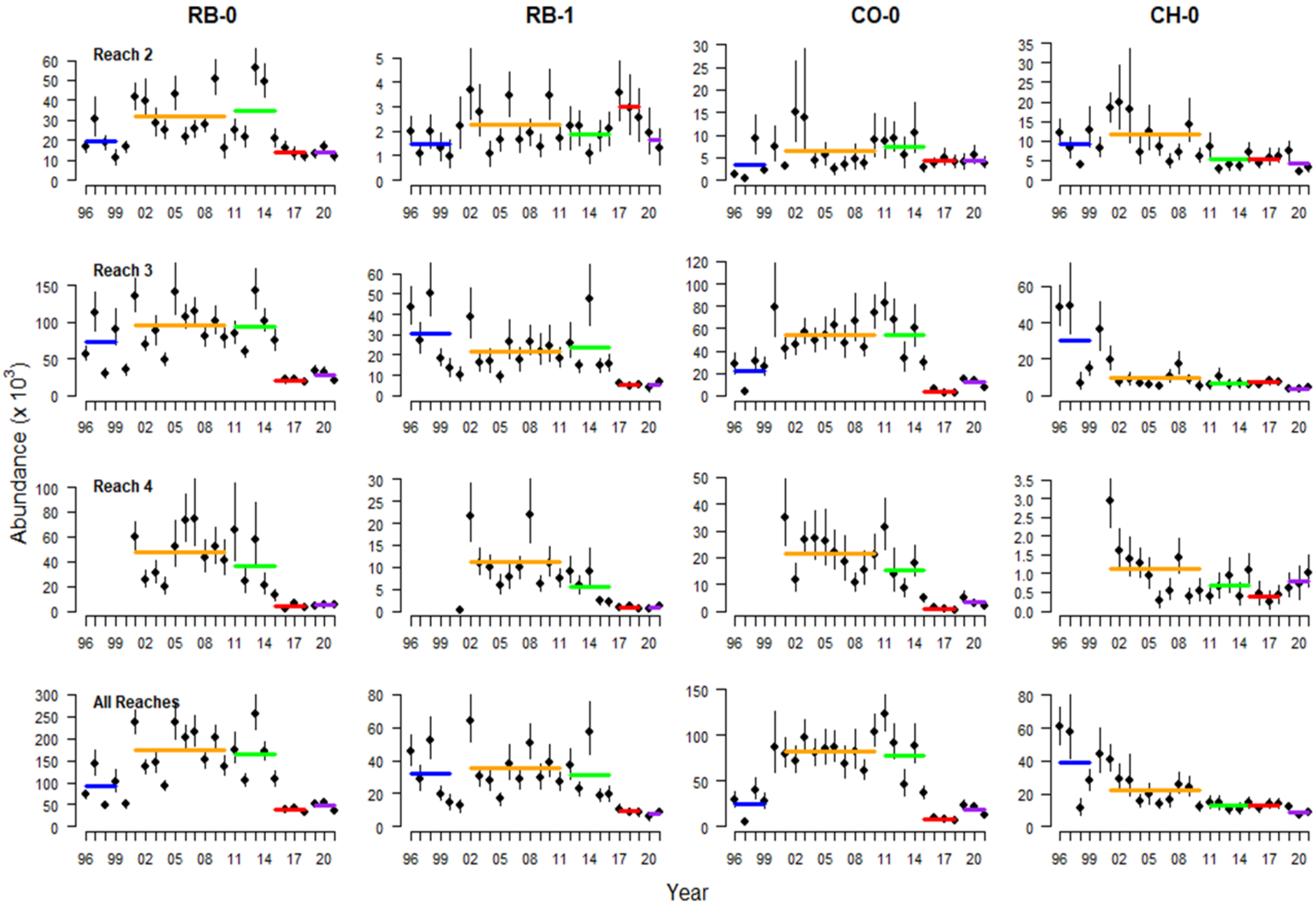


Figure 3.28 **Abundance (in thousands) of juvenile salmonids in the lower Bridge River by reach (row) and species-age class (column). Points and vertical lines show mean values and 90% credible intervals from posterior distributions of abundance for each year from the hierarchical Bayesian model, respectively. Blue, orange, green, red and purple lines show the mean values for trials 0, 1, 2, high flow and post-high flow treatments, respectively. RB-0, RB-1, CO-0, and CH-0 denote age-0 mykiss, age-1+ mykiss, age-0 coho, and age-0 Chinook, respectively.**

Chinook fry abundance increased slightly in Reach 2 (+2K) under the Trial 1 treatment relative to Trial 0, but declined in Reach 3 (-20K) likely owing to higher incubation temperatures which result in premature emergence in that reach (Figure 3.28 and Figure 3.8 in Section 3.1.5). Chinook recruitment in Reach 4 has been low (~0.3 to 3K) across all flow treatments. Due to a continuation of these temperature conditions from the release (and similar release discharges) during the incubation period for Chinook, abundances during stock assessment sampling have remained low within reaches and among flow treatments since Trial 1 for this species. Unlike the case for mykiss and coho fry, the high flows from 2016–2018 did not result in a further decline in Chinook fry abundance (relative to Trial 2) within the study area, perhaps because their abundance was already reduced. Post-high flow abundances were ~3K lower in Reach 3 and ~1K lower in Reach 2 relative to the Trial 2 and High Flow production in these reaches. The continued reduced abundance of Chinook fry across multiple flow treatments suggests that, under current constraints, flow volumes may be less a driving factor for recruitment of this species within the study area than the incubation issue.

So, for the species and age classes that increased since the return to lower peak flows, the change was largely attributable to Reach 3 (Figure 3.28). For mykiss fry, the ~7K additional fish in Reach 3 accounted for ~83% of the Post-high flow increase for that species-age class; and for coho fry, the additional ~8K fish in Reach 3 accounted for ~75% of the Post-high flow increase for that species-age class. The ~3K increase in coho fry in Reach 4 accounted for ~23% of the Post-high flow increase. As noted above, the contributions by reach for mykiss parr and Chinook fry were generally equivalent to, or lower than, the distribution during the high flow years.

Despite some differences in mean weights among species and age classes under the different flow treatments described in Section 3.4.1 (see Figure 3.23), the trends in biomass among flow treatments generally followed those based on abundance (Table 3.16b, Figure 3.27 and Figure 3.29). This was because the changes in abundance were more substantial than the relative changes in mean weights among treatments. However, the higher mean weights during the High flow and Post-high flow years had a slight moderating effect on the change in biomass (relative to Trial 2) than the change in abundance. This moderating effect was evident for each species and age class during the High flow and Post-high flow treatments and was most notable for mykiss (fry and parr) since the increases in mean weights were more substantial for this species. As such, the biomass estimates during the Post-high flow years were 40%, 31%, 28% and 62% of the Trial 2 estimates for mykiss fry, mykiss parr, coho fry and Chinook fry, respectively (compared to 29%, 24%, 25% and 77%, respectively, for abundance).

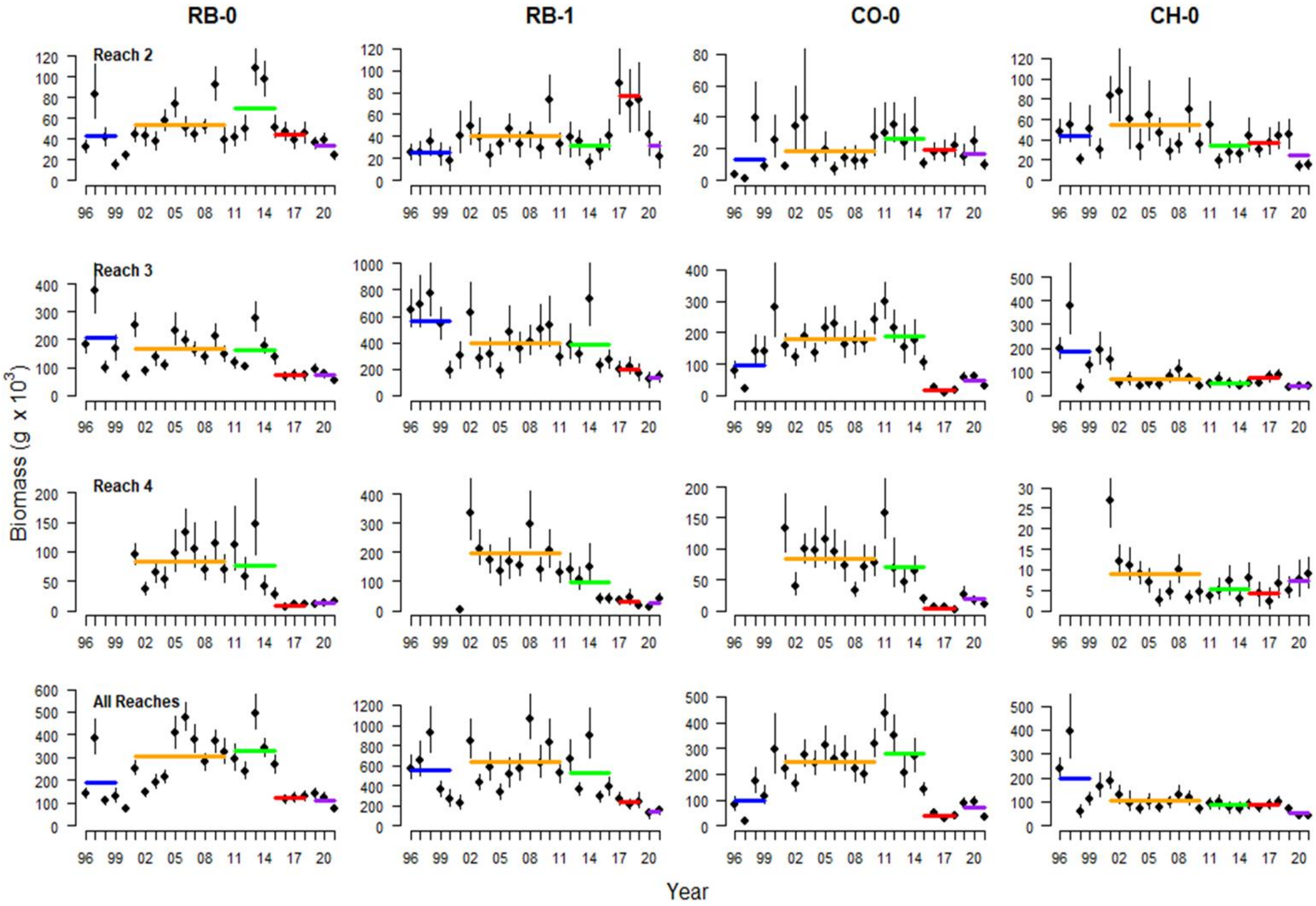


Figure 3.29 Biomass (in thousands of grams or kilograms) of juvenile salmonids in the Lower Bridge River by reach (row) and species-age class (column). See caption for Figure 3.28 for details.

Mixed Effects Model Results

We used a mixed effects model to formally test for flow effects (according to the various flow treatments from Terzaghi Dam) on the density and biomass of juvenile salmonids in the study area across the years of monitoring to-date. In the Year 9 report, we tested two approaches: one approach assessed flow effects that *were not* stratified by reach, and another where they *were* stratified by reach (Sneep et al. 2021). The intent of the two approaches was to determine if the effects of the various flow releases on each target species and age class could be adequately characterized for the study area as a whole or if there were reach-specific effects (including the re-wetting of Reach 4). As described in the Year 9 report, the reach-stratified model was more informative for interpreting the BRGMON-1 results based on evaluation of the Deviance Information Criteria (DIC) scores (see the Year 9 report for a more detailed explanation; Sneep et al. 2021). As such, we have not repeated the comparison of stratified and unstratified models here but instead carried forward with the stratified model and added covariates (as described in Section 2.8.5) to test for effects or interactions of other potentially informative variables in addition to flow.

The mixed effects model fit the data well, explaining between 0.86 and 0.96 of the variation in log density or biomass across reaches and years (Data r^2 in Table 3.18). Fixed effects of reach and flow treatment explained between 0.63 and 0.87 of the total prediction variation in log density or biomass across all reaches and years (Fixed effects r^2). Reach and flow treatment effects explained more variation in juvenile densities and biomass for mykiss parr (RB-1; 0.87-0.90) and coho fry (CO-0; 0.82-0.83) than mykiss fry (RB-0; 0.72-0.80) and Chinook fry (CH-0; 0.63-0.71).

Table 3.18 Data r^2 (square of the Pearson correlation coefficient) and fixed effects r^2 (see description in Section 2.9.4) values for describing fit and quantifying the explanatory ability of reach and flow treatment fixed effects. Results are based on a model that includes reach, year, and flow treatment effects (by reach) but without covariate effects (eqn. 1).

Species-Age	Data r^2		Fixed Effects r^2	
	Density	Biomass	Density	Biomass
RB-0	0.96	0.94	0.80	0.72
RB-1	0.95	0.95	0.87	0.90
CO-0	0.92	0.92	0.83	0.82
CH-0	0.91	0.86	0.71	0.63

The model predicted that mykiss fry and parr and coho fry densities (Figure 3.30) and biomass (Figure 3.31) in Reach 4 peaked under Trial 1 (3 $m^3 \cdot s^{-1}$ mean annual flow), declined under Trial 2 (6 $m^3 \cdot s^{-1}$ mean annual flow), and were low under both High and Post-high flow treatments. In

Reach 3, mykiss and coho density and biomass were similar between Trial 1 and Trial 2 but were also low under the High flow and Post-high flow treatments. In contrast, Chinook fry density and biomass were highest during the Pre-flow period (Trial 0) and Trial 1 in Reach 2, and highest during the Pre-flow period in Reach 3. Density and biomass of Chinook fry was always low in Reach 4 and was not appreciably affected by flow treatment, other than the small benefit due to rewetting this reach starting in Trial 1.

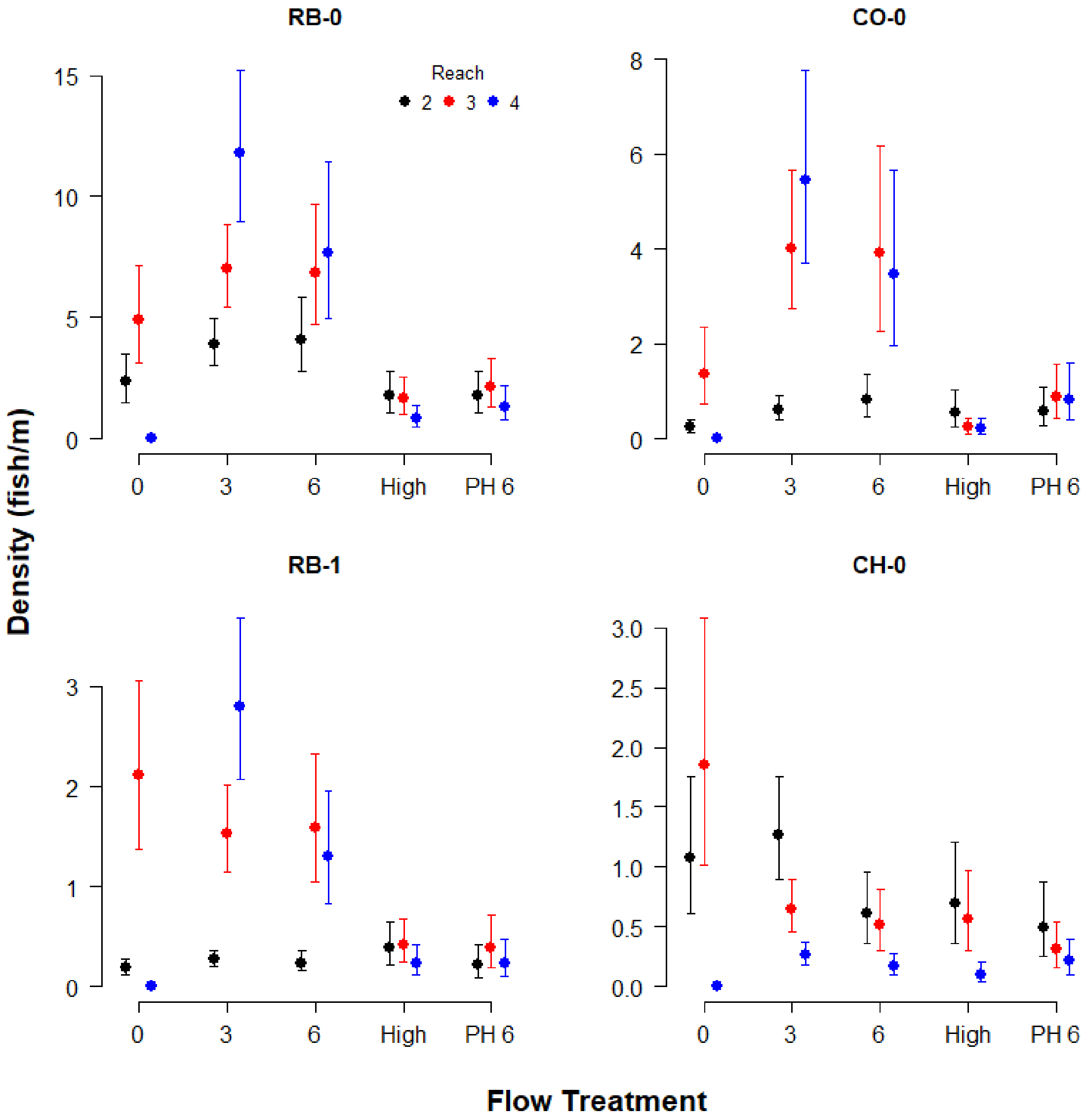


Figure 3.30 Results of the mixed effect model showing predicted density by flow treatment and reach for mykiss fry (RB-0; top left), mykiss parr (RB-1; bottom left), coho fry (CO-0; top right), and Chinook fry (CH-0; bottom right). Flow treatments are:

0 m^3·s^{-1} = Trial 0, 3 m^3·s^{-1} = Trial 1, 6 m^3·s^{-1} = Trial 2, High = High flows, and PH 6 = Post-high (6-9 m^3·s^{-1}) flows.

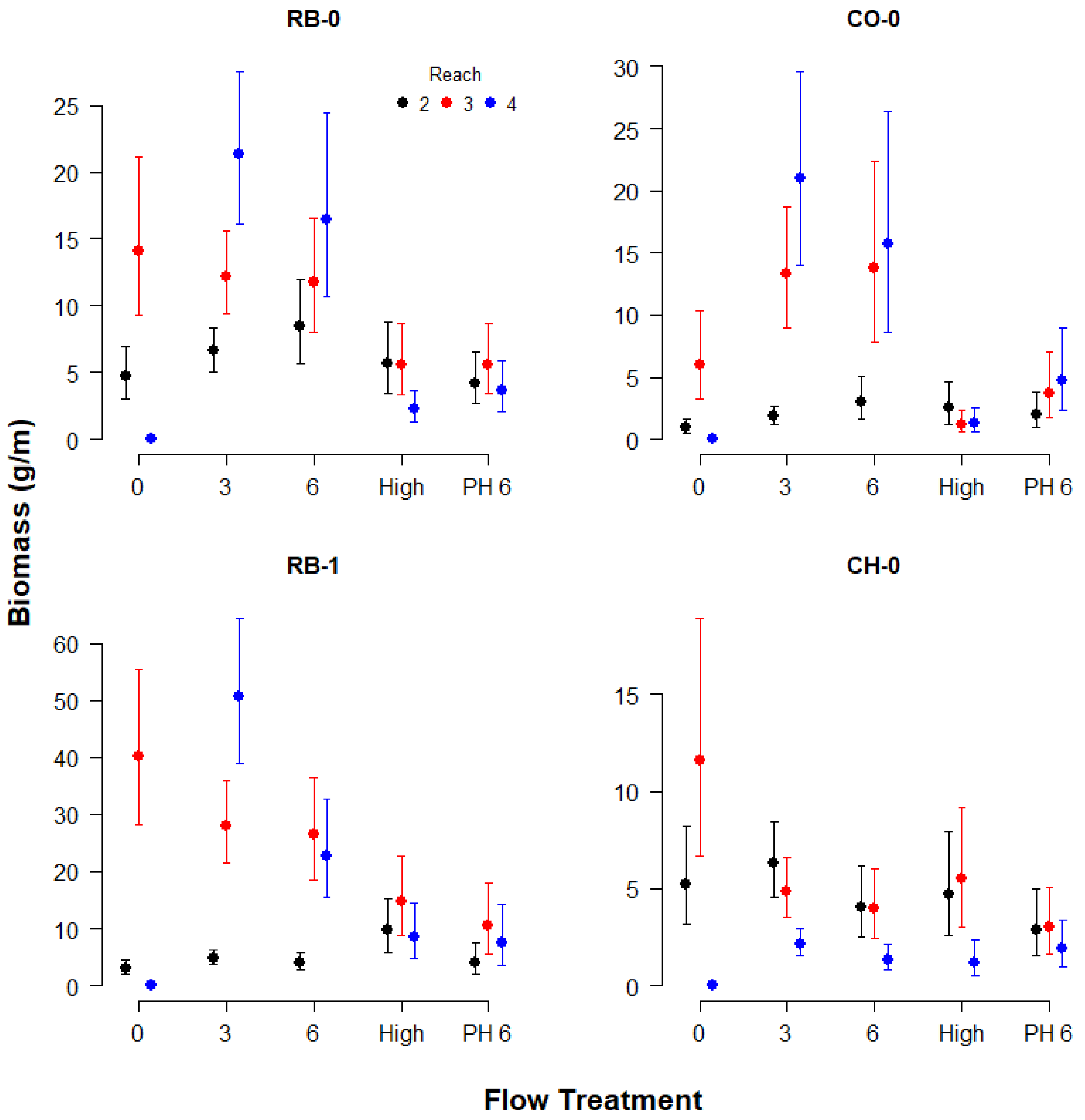


Figure 3.31 Results of the mixed effect model showing predicted biomass by flow treatment and reach for mykiss fry (RB-0; top left), mykiss parr (RB-1; bottom left), coho fry (CO-0; top right), and Chinook fry (CH-0; bottom right). See Figure 3.30 caption for details.

A major advantage of the mixed effects model is that it provides credible intervals that accurately reflect the extent of variation in density or biomass across years within flow treatments. This variation can be extensive resulting in substantive overlap in average density and biomass predictions between flow treatments. For example, mykiss and coho fry densities in Reach 4 were higher under Trial 1 compared to Trial 2, but there was ~ 50% overlap in credible intervals (Figure 3.30 and Figure 3.31). This indicates there was a high probability that the Reach 4 densities were not appreciably different between these two trials once the variation among years within trials is considered. There was similar or higher overlap between High flow and Post-high flow treatments for all species-age classes.

Estimated covariate effect sizes were generally close to zero and all estimates were highly uncertain (average coefficient of variation was 450%) (Appendix C, Table C1). Deviance information criteria scores indicated that covariates did not increase the predictive capability of the model relative to the version that only included reach and reach-specific flow treatment effects (covariate = Null). The lack of a spawner covariate effect is consistent with the stock-recruit analysis, which generally does not show increasing juvenile abundance with increasing spawner abundance beyond relatively low thresholds (see Section 3.4.3).

Reach 1 Results

Three years of stock assessment sampling results were available for Reach 1 (2019-2021) which enabled comparison of abundance results among all four reaches of the Lower Bridge River for the Post-high flow years (Figure 3.32). Analysis of the Post-high flow sampling data from Reach 1 yielded mean abundance results by species/age class of approx.: 22,000, 4,000, 11,000, and 8,000 for mykiss fry, mykiss parr, coho fry and Chinook fry, respectively (Figure 3.32 Top). This abundance pattern among species-age classes in Reach 1 was generally the same for each year of results available. However, the numbers of mykiss fry, mykiss parr and Chinook fry increased by approx. 8K, 7K, and 8K fish, respectively, from 2019 to 2021 suggesting a modest amount of recovery for these species-age classes in this reach following the years of high flows from 2016-2018 (although abundances in Reach 1 before the Post-high period are unknown). The abundance of coho fry remained approx. the same in Reach 1 across the Post-high flow years.

The totals for all species in Reach 1 were ~32,000 fish in 2019, ~49,000 fish in 2020, and ~56,000 fish in 2021, for a mean total of ~46,000 for the Post-high flow treatment. This represented 36% (or approx. 1/3rd) of the total juvenile salmonid production (i.e., ~128,000 fish) in the study area as a whole for this period. However, Reach 1 constitutes nearly half (i.e., 20 km) of the total length of the study area (i.e., 40.9 km). Therefore, lineal densities (# of fish per km) were generally among the lowest in Reach 1 compared to the other study reaches for most species and age classes (Figure 3.32 Bottom). However, at this stage of post-high flow recovery, lineal densities in Reach 1 were: on par with Reach 4 for mykiss fry; on par with Reach 2 and Reach 4 for mykiss parr; on parr with Reach 2 for coho fry, and slightly higher than reaches 3 and 4 for Chinook fry (Figure 3.32 Bottom).

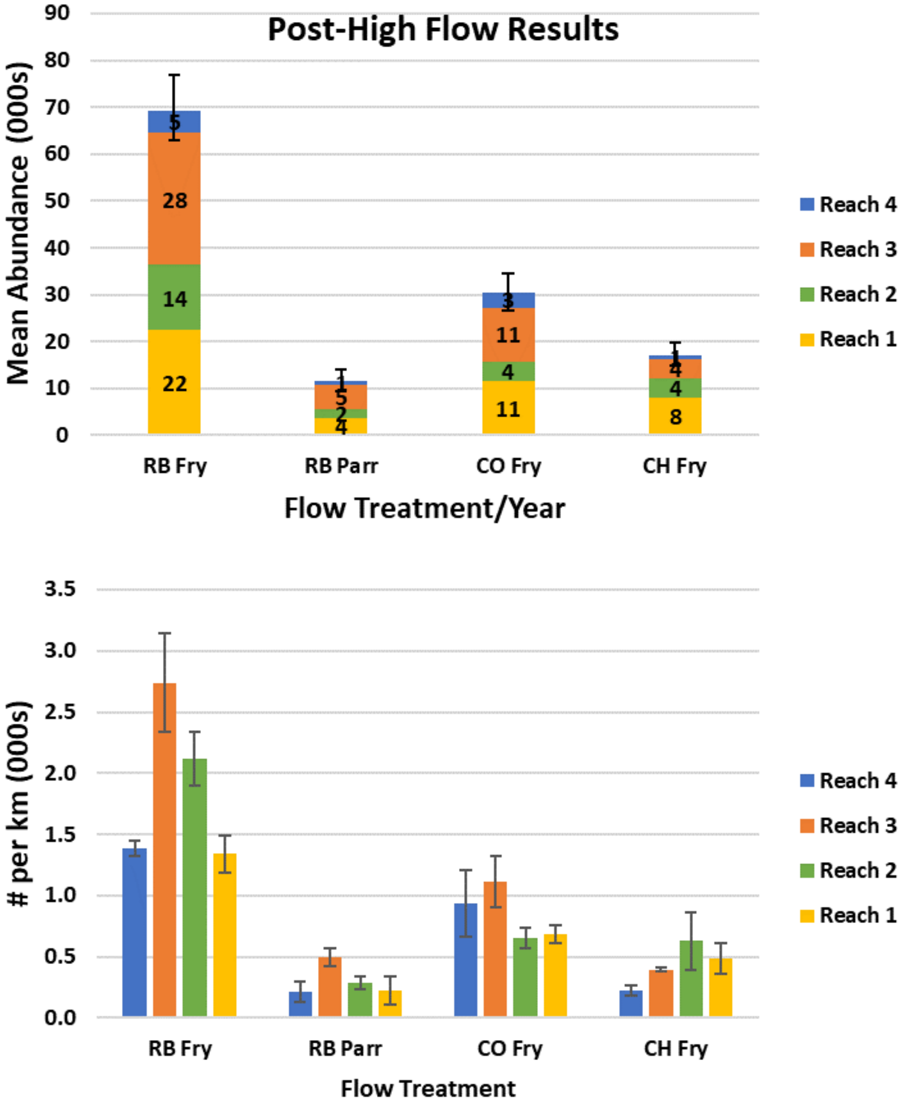


Figure 3.32 **Abundance (in thousands) of mykiss fry (RB Fry), mykiss parr (RB Parr), coho fry (CO Fry) and Chinook fry (CH Fry) among reaches of the Lower Bridge River (Top), and lineal densities (# of fish per km) by species and reach (Bottom) for the Post-high flow years (2019-2021). Error bars in the top plot are 90% credible intervals from posterior distributions of abundance for each year from the hierarchical Bayesian model and error bars in the lower plot are ± standard error.**

3.4.3. Stock-Recruitment

In general, the shift in escapement-fry stock-recruitment curves for coho and Chinook across different flow treatments reflected the changes in fry abundance seen in the juvenile abundance analysis (Section 3.4.2). For this Year 10 report, a new curve was generated for the Post-high flow treatment (2019-2021) for coho and Chinook; however, with only 3 datapoints each for the High Flow and Post-high Flow years, the curves generated for both of those treatments must be considered rather tenuous.

The maximum likelihood estimate for coho spawner escapement in 2020 was 537 (95% confidence intervals: 345–835) for reaches 3 and 4 (White et al. 2021). This AUC-based estimate for coho was in the range of several other escapement estimates during previous flow treatments (e.g., 1997 *n*= 600; 2006 *n*= 653; 2010 *n*= 449; 2014 *n*= 410; 2016 *n*= 473; 2017 *n*= 451). It was paired with the 2021 coho fry abundance estimate of 12,770 (90% credible intervals: 10,190–15,630). The estimate for Chinook spawners in 2020 was 96 for reaches 3 and 4, which included the visual-based AUC estimate of 16 (95% confidence intervals: 6–41) coupled with the 80 spawners enumerated at the fish fence (White et al. 2021). This estimate was similar to, or higher than, estimates from several previous years (e.g., 2009 (*n*= 21); 2011 (*n*= 82); 2017 (*n*= 120); 2018 (*n*= 25); and 2019 (*n*= 161). However, as White et al. 2021 report, there were insufficient radio tag and visual tag data to estimate observer efficiency and survey life for 2020, so average values were used. The 2020 escapement estimate was paired with the 2021 Chinook fry estimate of 8,500 (90% credible intervals: 6,130–11,030), which was the second lowest recruitment value for this species (after 2020; *n*= 6,820) since monitoring began in 1996.

Coho fry abundance increased under the Trial 1 and Trial 2 treatments relative to the Trial 0 pre-flow period (Figure 3.33). The magnitude in the shift of the stock-recruitment curve for coho fry (e^{λ}) was 2.9, 2.3, 0.35, and 0.70 for Trial 1, Trial 2, High Flow, and Post-high flow years, respectively. That is, for a given level of escapement, the stock-recruitment model indicates an approximate 2- to 3-fold increase under trials 1 and 2 relative to pre-flow conditions, respectively, and a reduction by 65% under high flows and 30% during the Post-high flow years. As noted above, the 2021 datapoint for coho was in the same range of escapement estimates from several other years, including during Trial 2 (which had a similar hydrograph). However, those escapements in the earlier flow trials produced between 20,000 and 115,000 more coho fry than the High Flow and Post-high Flow years.

The points in Figure 3.33 (bottom plot) result in a steep initial slope which is not uncommon for coho populations where escapement and smolt production has been monitored (Korman and Tompkins 2014). However, it is important to note that the estimated initial slope for this set of curves hit the boundary of our maximum assumed value (1500 egg/female x 0.5 females/total spawners x 50% egg-fry survival rate = 375 fry/spawner) and would be unrealistically steep if we had not constrained this parameter.

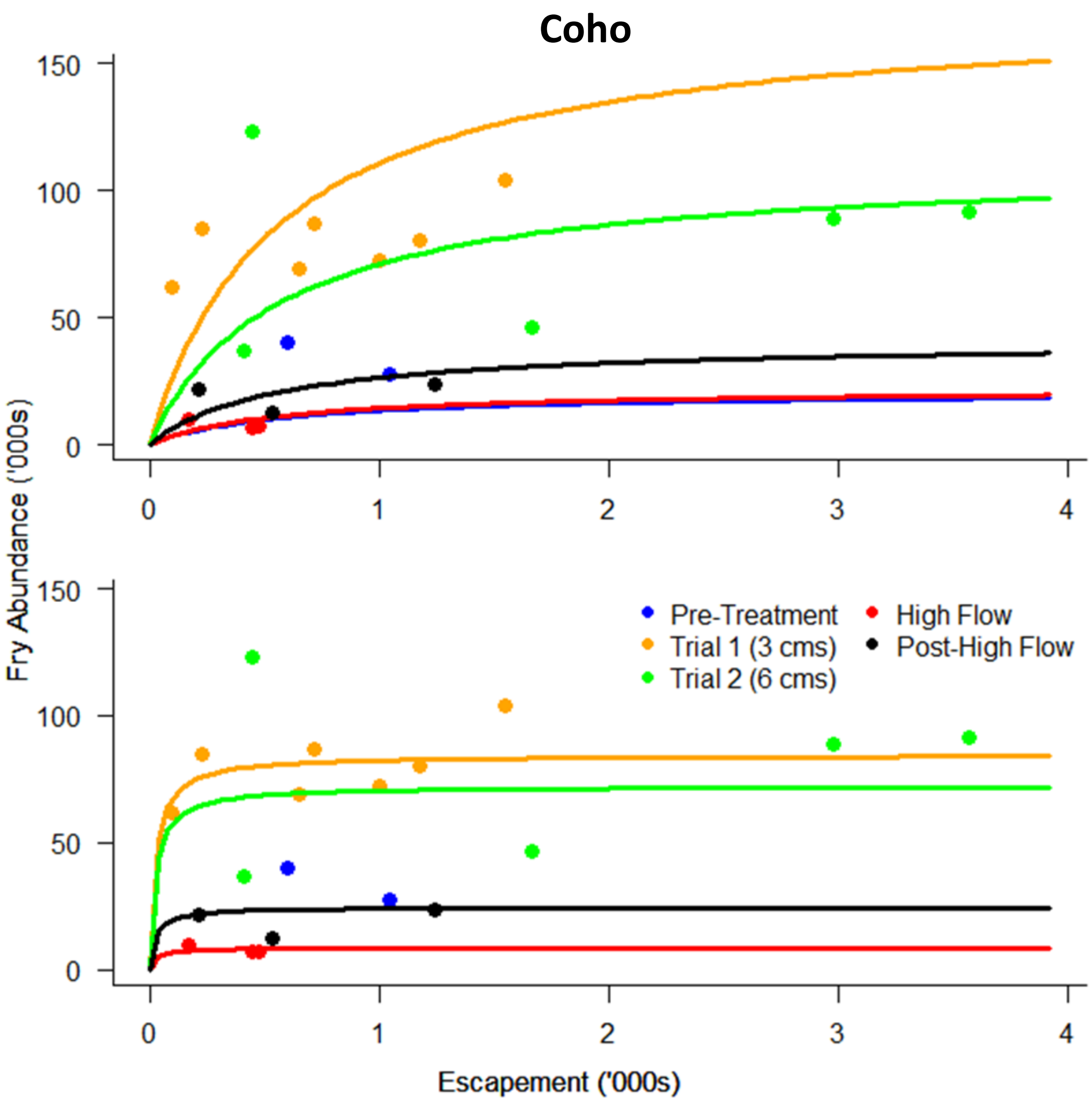


Figure 3.33 **Coho spawner-fry Beverton-Holt stock-recruitment curves fit assuming a maximum initial slope of 37.5 fry/spawner (5% egg-to-fry survival; Top Plot) and 375 fry/spawner (50% egg-fry survival; Bottom Plot). Points show annual estimates of escapement and fry abundance. The colour of each point and curve corresponds to flow treatment (see plot legend).**

Assuming a lower maximum initial slope (e.g., 37.5 fry/spawner based on a 5% egg-fry survival rate) constrains the curves to a much greater extent (Figure 3.33 top plot). In this case, there are many data points that have escapements that are less than required to maximize fry production. These more constrained curves are rather different than the set described above yet provide a near equivalent fit to the data (due to the wide scatter of points and limited *n* size for defining each curve). The difference in log-likelihood measuring the fit of the curves in the top plot vs. the bottom plot of Figure 3.33 is less than 2 units and therefore one set of curves cannot be considered more reliable than the other based on fit alone.

The escapement-fry stock-recruitment curve for Chinook also had a very steep initial slope that was constrained by our assumption that it could not exceed 1250 fry/spawner (5000 eggs/female x 0.5 females/spawner x 50% egg-fry survival rate, Figure 3.34 bottom plot). The stock-recruitment λ values indicate that recruitment under Trial 1, Trial 2, the High flow and the Post-high flow treatments were 74%, 46%, 44% and 80% of the recruitment under the Pre-flow conditions (Figure 3.34 bottom plot). Owing to the steep initial slope on this set of curves (based on 50% egg-to-fry survival) there was no clear indication that Chinook escapements have been limiting fry abundance among the flow treatments, and the Post-high flow datapoints were within a large cluster of datapoints near the origin on the x-axis. However, like the case for coho, the initial slope of the stock-recruitment curve for Chinook depends on the maximum initial slope constraint. When we lower egg-fry survival to 5% (initial slope constraint = 5000 x 0.5 x 0.05 = 125 fry/spawner) the model makes the unlikely prediction of a positive effect of the flow treatments on Chinook production capacity relative to the pre-flow conditions (Figure 3.34 top plot). Yet, again, this more constrained curve provides a near equivalent fit to the data (the likelihood difference between fits is less than 2 units), which highlights the limitations on drawing firmer conclusions based on the available data. Note that on both plots in Figure 3.34 the curve for the Post-high flow treatment is the highest despite relatively low escapement and low recruitment in each of those years. This highlights the limitations of drawing stock-recruitment curves based on only three datapoints that are near the origin of both the x- and y-axes.

Given the uncertainty regarding which set of curves (i.e., based on 50% vs 5% egg-fry survival) are more reliable for both of these species, the stock-recruit analysis is not ideal for allowing us to confidently differentiate flow effects from potential stock size effects (escapement). Though there is a fairly large sample size overall (*n*= 21 data points for coho and *n*= 23 datapoints for Chinook), the data must be parsed according to flow treatment because we have observed different levels of production due to variable incubation and rearing conditions under the different flow trials (i.e., a flow treatment effect; see Section 3.4.2). As a result, there is a much smaller *n* size for defining the initial slope of each individual curve. Characterization of fecundity, male:female spawner proportion, and egg-to-fry survival for the LBR populations would provide in-situ data to address the set of assumptions currently applied to the stock-recruitment model; however, we would not have these data for past years and flow treatments to understand how these parameters may also have changed over time.

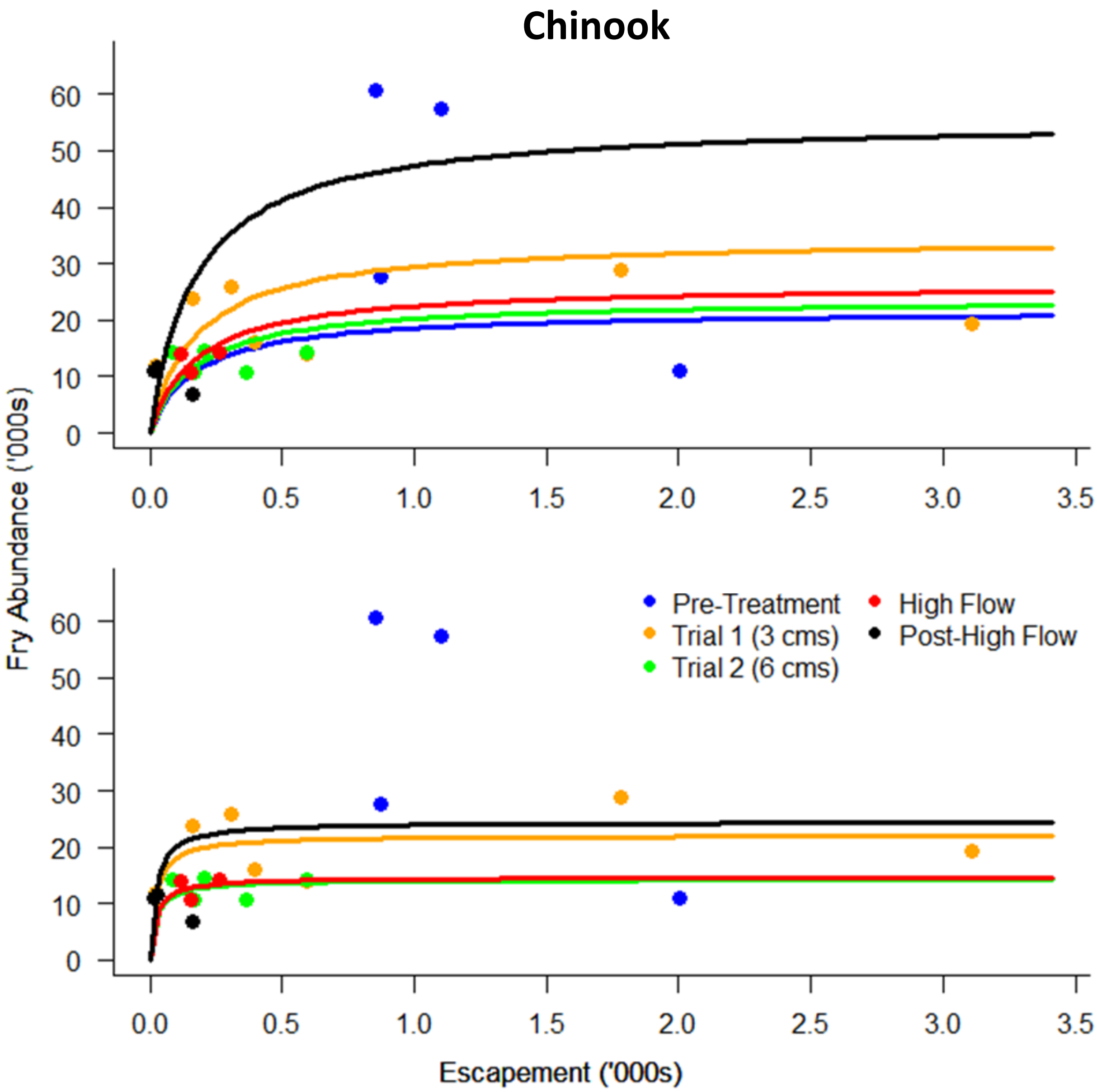


Figure 3.34 Chinook spawner-fry Beverton-Holt stock-recruitment curves fit assuming a maximum initial slope of 125 fry/spawner (5% egg-to-fry survival; Top Plot) and 1250 fry/spawner (50% egg-fry survival; Bottom Plot). Points show annual estimates of escapement and fry abundance. The colour of each point and curve corresponds to flow treatment (see plot legend).

As such, though the stock-recruitment analysis is a bit of a crude tool for the purposes of this program, it does suggest that changes in escapements among years may have been a less significant driver than flow treatment effects on juvenile production for these species to-date, but that escapements at the low end of the range during the Modified Operations years (High flows and Post-high flows) could be contributing to slower recovery from the effects of the high flows. To supplement the stock-recruit analysis, we also applied annual spawner escapement estimates as a covariate in the mixed effects model for each species (see description of this covariate in Section 2.8.5). Like the other covariates tested, the effect sizes for the escapement metrics were generally close to zero and highly uncertain (Appendix C, Table C1). The model results indicated that the escapement information did not increase the predictive capability of the model relative to the version that only included reach and reach-specific flow treatment effects. This was likely because juvenile abundance does not tend to increase with increasing spawner abundance above relatively low escapement thresholds, as supported by the stock-recruit analysis.

3.5. Modified Operations Monitoring

3.5.1. High Flow Monitoring

The modified operations (i.e., > Trial 2 peak flow) period in 2021, along with a description of the discharge and continuous water temperature measurements at a range of stations downstream of the dam are provided in Section 3.1 and Appendix D. Carpenter Reservoir was drawn down to 620.9 m in 2021, which was approx. 4 m higher than the minimum elevations in the other Post-high flow years (i.e., 2019 min.= 616.2 m and 2020 min.= 617.6 m). The reservoir was filling across the high flow period in 2021 from 620.9 m to 641.9 m, a total change of 21.0 m. The fill rate was fairly consistent for most of the peak discharge period (average +0.33 m/day) and then diminished by the middle of July to an average of +0.07 m/day.

Water Quality

Air and water temperatures generally increased in the LBR across the monitoring dates; however, the spot water temperature measurements taken on the high flow survey dates were typically warmer than the data recorded by the loggers (by up to 6°C in some cases). Turbidity levels increased from the May to the June survey dates, then decreased during July, before increasing again at the end of August. This pattern of turbidity was generally reflected in the continuous turbidity data recorded by the loggers (see Section 3.1.6) but the loggers recorded a much broader range of turbidity levels and provided a more complete time series between March and November 2021. Total dissolved gas (TDG) levels were not available for the May survey, but generally remained stable across the rest of the modified operations surveys (i.e., between 100% and 106% saturation) among locations (Table 3.19).

The spot water temperatures at the mainstem LBR sites ranged between 9.7°C (at the Terzaghi Plunge Pool on 11 May) and 21.5°C (at the bottom of Reach 3 (Rkm 25.5) on 27 July) and there was a general increase in temperature from upstream to downstream sites on each date due to the influence of ambient temperatures across this seasonal period. The Yalakom River temperatures were consistently colder than the LBR temperatures on each survey date (as also shown by the plots of daily logger data in Appendix D). The upper range of the spot measurements were up to 8°C warmer than daily average temperatures for this period during trials 1 and 2; however, as noted above, there were discrepancies between the spot measurements and the hourly logger measurements recorded at or near the same locations, particularly on the dates when the highest water temperatures were recorded. The maximum hourly temperatures recorded by the loggers on the survey dates were 14.6°C (on Aug 30), 15.1°C (on Aug 30) and 15.6°C (on Jul 27) at Rkm 39.9 (Reach 4), Rkm 30.4 (middle of Reach 3) and Rkm 26.4 (bottom of Reach 3), respectively. These temperatures were warmer than the Trial 2 averages on these dates by 0.4-2.7°C (also see Figure 3.7 and description of temperature differences among trials in Section 3.1.5) and were within the optimal range for rearing by Chinook, coho and steelhead, as cited in the literature (McCullough et al. 2001; Oliver & Fidler 2001; Myrick & Cech 2000). These warmer spring-summer temperatures may have contributed to the larger mean size of the juveniles from these species by the time of the annual stock assessment sampling in September during the High flow and Post-high flow years relative to Trial 2.

Spot turbidity values in the LBR mainstem ranged from 10.0-20.8 NTU across locations and survey dates in 2021. Turbidities were consistently lower in the Yalakom River (range: 2.4-7.4 NTU) on the survey dates and generally decreased from upstream to downstream in the mainstem LBR, likely due to the attenuation of less turbid tributary inflows. The maximum turbidity values documented by the spot measurements in 2021 were lower than the maximum values during the high flow years (i.e., 47.9-57.9 NTU in 2017; and 35.9-38.0 NTU in 2018). However, as noted above, the results of the 2021 spot measurements covered a much narrower range than the continuous data provided by the turbidity loggers installed between March and November 2021 (refer to Section 3.1.6 for the turbidity logger results).

TDG (% saturation) levels were highest at the monitoring location closest to the dam, ranging between 105-106% on the June and July survey dates (i.e., the 25 $m^3 \cdot s^{-1}$ peak flow release period) and then decreased to 102% on 30 August (when the flow release was 3 $m^3 \cdot s^{-1}$ from the dam). TDG levels at all other monitoring locations and survey dates in 2021 were very similar (range: 100%-103%), indicating that the slightly higher readings closest to the dam diminish quickly with distance downstream. Spot TDG measurements were higher during the high flow releases (up to 102 $m^3 \cdot s^{-1}$) in 2018: Highest recorded values were between 110-112% at the Terzaghi plunge pool; 105-107% at Russel Springs (Rkm 30.4), and 106-109% at the U/S of Yalakom site (Rkm 25.5) that year. There may be a correlation between flow release discharge and TDG saturation level; however, the mechanism causing the higher TDG at high flows from the low-level outlet at Terzaghi Dam has not been determined. In 2021, the observed saturation levels were below the thresholds for triggering a mitigation response according to Table 1 of BC Hydro's Total Dissolved Gas Management Strategy (i.e., BC Hydro 2014). Despite this, a low-level assessment (described under "kokanee entrainment monitoring" below) was conducted on each survey date to look for fish exhibiting signs of ill-effects, or mortalities.

Table 3.19 Summary of water quality measurements taken at 3 monitoring locations in the Lower Bridge River and 1 location in the Yalakom River across the high flow release period in 2021.

Site Parameter	**2021 Survey Dates**							
	May	**June**		**July**			**Aug** [a]	
	11	**8**	**10**	**6**	**20**	**27**	**26**	**30**
Carp. Res. Elevation (m)	622	630	630	639	641	642	645	645
TRZ Release ($m^3 \cdot s^{-1}$)	24	26	25	25	25	25	3	3
TRZ Plunge Pool (Rkm 40.9)								
Water Temperature (°C)	9.7		12.8	13.1		13.7		14.3
Air Temperature (°C)			18.3	23.6		20.8		16.4
TDG (%)			106	106		105		102
Turbidity (NTU)	17.3		20.8	13.3		11.6		18.4
Conductivity (µS/cm)	141	99.1			78.1		68.2	
Alkalinity (mg/L)	55.7	37.2			36.0		25.6	
Russel Springs (Rkm 30.4)								
Water Temperature (°C)	10.6		12.7	16.7		17.2		14.5
Air Temperature (°C)			17.5	27.2		28.1		17.7
TDG (%)			101	101		103		100
Turbidity (NTU)	16.0		19.7	12.3		10.0		17.8
Conductivity (µS/cm)	142	102			80.8		87.2	
Alkalinity (mg/L)	55.9	39.7			38.2		34.4	
U/S Yalakom (Rkm 25.5)								
Water Temperature (°C)	11.3		13.3	18		21.5		16.2
Air Temperature (°C)			19.3	30.8		30.2		21.8
TDG (%)			102	100		100		100
Turbidity (NTU)	18.5		20.1	11.1		10.1		14.9
Conductivity (µS/cm)	143	102			83.7		96.7	
Alkalinity (mg/L)	56.6	40.3			30.9		38.5	
Yalakom River								
Water Temperature (°C)	7.4		7.9	14.9		14.6		12.8
Air Temperature (°C)			21.1	29.4		29.4		16
TDG (%)			101	101		101		100
Turbidity (NTU)	3.0		3.6	7.4		3.6		2.4
Conductivity (µS/cm)	223	174			178		205	
Alkalinity (mg/L)	99.8	77.7			82.1		100	

[a] The measurements in August were taken after flows had been ramped down to Trial 2 levels on 30 July 2021.

Conductivity and alkalinity were measured from water samples collected monthly at index site locations in the LBR and Yalakom River (dates of collection shown in Table 3.19). At peak flows in 2021, values were relatively consistent among the sites but varied according to monthly sample period. The conductivity range among LBR mainstem sites was 141 to 143 µS/cm on 11 May, 99.1

to 102 µS/cm on 8 June, 78.1 to 83.7 µS/cm on 20 July, and 68.2 to 96.7 µS/cm on 26 August. Alkalinity ranged from 55.7 to 55.6 mg/L CaCO3 on 11 May, 37.2 to 40.3 mg/L CaCO3 on 8 June, 30.9 to 36.0 mg/L CaCO3 on 20 July, and 25.6 to 38.5 mg/L CaCO3 on 26 August. The values for both parameters were higher in the Yalakom River, ranging between 174 and 223 µS/cm (conductivity) and 77.7 and 100 mg/L CaCO3 (alkalinity).

For reference against values for these parameters from past flow treatments, the mean conductivity and alkalinity by trial and season for the same locations in reaches 3 and 4 are provided in Table 3.20. The flow release from Carpenter Reservoir is characterized by lower conductivity and alkalinity than the groundwater and tributary inflows to the LBR channel. Therefore, following the onset of the continuous flow release, and particularly when the flow release magnitude has dominated flow volumes (i.e., the High Flow years and during the peak flow period in 2021), the values for these parameters have tended to decrease and become more consistent among stations, particularly above the Yalakom confluence. The conductivity and alkalinity values documented in 2021 were all within the range of values from previous flow treatments for these parameters.

Table 3.20 Summary of mean conductivity and alkalinity values for Site A in Reach 4 and sites D and E in Reach 3 by flow trial and season.

Flow Treatment	Season	Parameter					
		Conductivity (µS/cm)			Alkalinity, as $CaCO_3$ (mg/L)		
		Site A (Rkm 39.9)	Site D (Rkm 30.4)	Site E (Rkm 26.4)	Site A (Rkm 39.9)	Site D (Rkm 30.4)	Site E (Rkm 26.4)
0 (Pre-flow)	**Spring**	-	178	188	-	149	136
	Summer	-	187	183	-	156	141
	Fall	-	224	224	-	178	148
1 (3 $m^3 \cdot s^{-1}$)	**Spring**	107	122	122	84	98	98
	Summer	82	109	104	64	85	85
	Fall	72	100	103	57	78	81
2 (6 $m^3 \cdot s^{-1}$)	**Fall**	79	110	114	42	61	65
3 (High Flow)	**Fall**	-	-	-	28	43	49
4 (Post-High)	**Spring**	70	99	111	42	45	45
	Summer	109	112	113	29	35	35
	Fall	73	86	92	28	39	44

Kokanee Entrainment

Unlike during some of the previous high flow years (i.e., 2016 and 2017), entrained kokanee were not observed within the 1.5 km survey area below the dam in 2021. Surveys were conducted on four dates, including: 10 June, 6 July, 27 July, and 30 August. Flow releases from Terzaghi Dam were 25 $m^3 \cdot s^{-1}$ on the first 3 surveys and 3 $m^3 \cdot s^{-1}$ on the last survey. As noted in past high flow monitoring reports, effective enumeration of fish in the river from shore was hampered by the high flows and turbidity levels, which made for poor visibility conditions into the water.

Characterization of the specific reservoir and flow release conditions that may drive kokanee entrainment remain elusive. A total of 83 kokanee were observed across 13 survey dates in June and July 2016, when Carpenter Reservoir levels were filling from 632.3 to 638.0 m, and flow release magnitudes were between 35 and 96 $m^3 \cdot s^{-1}$ (McHugh et al. 2016). In 2017 and 2018, Carpenter Reservoir elevations and flow release magnitudes were fairly similar across the high flow period, but 48 entrained kokanee were observed in 2017 and none were observed in 2018 (O'Farrell and McHugh 2017). At this point it is difficult to say whether more years of monitoring data will help to sort out the potential interaction between Carpenter Reservoir elevations and Terzaghi Dam discharges on the incidence of kokanee entrainment observed in the LBR, but it may still be useful for establishing a relative index of entrainment in each year of modified operations, as well as documenting any evidence of TDG-related effects on fish.

Bank Erosion and Sediment Recruitment

Eighteen bank erosion sites were assessed during the peak flow period in 2021. Fifteen sites were on river right and three were on river left. All sites, except one, appeared to be comprised of naturally-derived substrate materials; the river left site at Fraser Lake (Rkm 33.3) was comprised of road-based materials, and was located just upstream of the rip rap placed at this site to address erosion issues in 2016. See the map produced by CNBS (formerly Coldstream Ecology Ltd.) in Appendix E for the erosion monitoring site locations.

Twelve of the eighteen sites showed signs of erosion and substrate recruitment to the river from the 2021 peak flows (Table 3.21). In most cases the erosion of material was caused by the interaction of the widened river channel (at peak flows) with the base of an active alluvial slide area adjacent to the river (Photo 3.1). Two erosion sites were considered medium-sized (<500 m^2), and both occurred in Reach 2: the "Horseshoe Bend (Top Corner)" site at Rkm 23.6 was ~200 m^2 and the "Below Horseshoe" site at Rkm 22.5 was ~150 m^2 (**Error! Reference source not found.**). Ten sites were classified as small (<50 m^2) covering a total of 113.5 m^2 altogether. The remaining six assessed sites (i.e., Plunge Pool, Fraser Lake, 242, 243, #244b and #249) did not show any signs of bank erosion under the 2021 peak flows.

Table 3.21 Summary of erosion and substrate recruitment sites observed during surveys across the peak flow period (Jun/Jul/Aug) in 2021.

Location Names	UTM Coordinates (Zone 10U)	River Bank (L or R)	Reach	Active Erosion (Y/N)	Approx. Length (m)	Approx. Width (m)	Approx. Area (m^2)	Substrate Composition (%)			
								Boulder	Cobble	Gravel	Fines
Plunge Pool	555098E 5626540N	R	4	N	20	0.5	10	5%	70%	-	20%
234	555498E 5626306N	R	4	Y	10	0.5	5	-	25%	25%	50%
236	555928E 5626290N	R	4	Y	10	0.5	5	100%	-	-	-
238	558173E 5627147N	R	3	Y	-	-	-	-	-	-	-
239	558144E 5627498N	R	3	Y	12	0.5	6	-	50%	-	50%
35	557796E 5627951N	R	3	Y	15	0.5	7.5	100%	-	-	-
Corner Site	557703E 5628160N	L	3	Y	20	0.5	10	75%	20%	5%	-
Top of new riprap at Fraser Lake	558133E 5629782N	L	3	N	10	0.5	5	100%	-	-	-
242	558145E 5629773N	R	3	N	20	0.5	10	10%	45%	45%	-
243	558136E 5630037N	R	3	N	20	0.5	10	-	50%	-	50%
244	558070E 5630204N	R	3	Y	40	0.5	20	-	50%	-	50%
244b	558028E 5630249N	R	3	N	20	0.5	10	5%	50%	45%	-
246	556495E 5631563N	R	3	Y	15	1	15	5%	75%	-	20%
247	556492E 5631464N	R	3	Y	15	1	15	5%	35%	60%	-
249	555750E 5632518N	L	3	N	20	0.5	10	-	50%	50%	-
250	555909E 5632828N	R	3	Y	20	1.5	30	-	-	100%	-
Horseshoe Bend (Top Corner)	559302E 5634649N	R	2	Y	100	2	200	-	25%	25%	50%
Below Horseshoe	559608E 5634038N	R	2	Y	150	1	150	10%	10%	-	80%

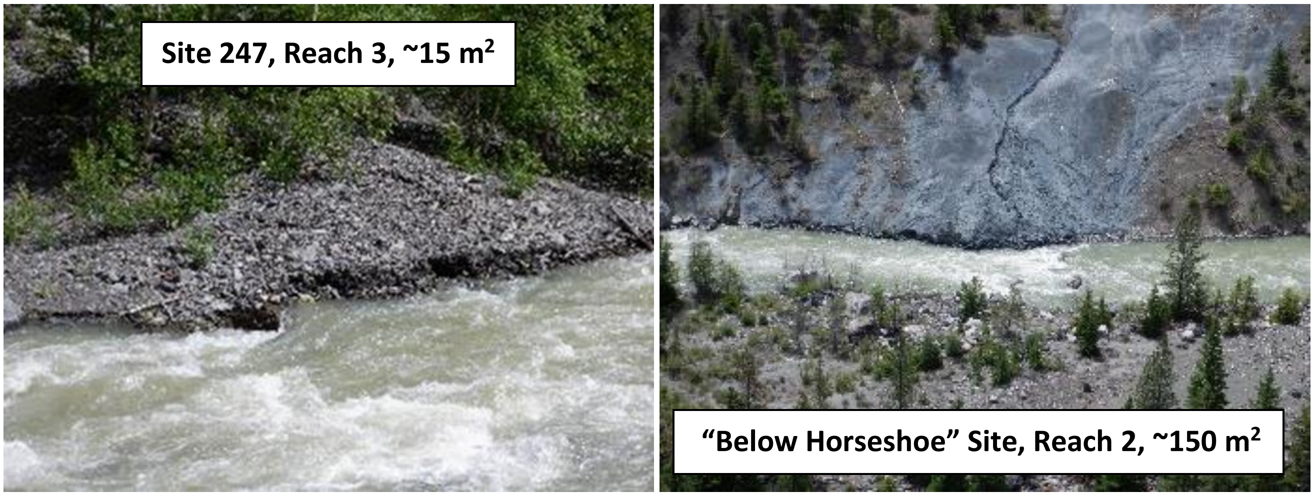


Photo 3.1 Examples of bank erosion at 2 sites in the Lower Bridge River at the interface between the widened river channel and the base of active alluvial slides at peak release flows (~25 $m^3 \cdot s^{-1}$) in 2021. Photos provided by Coldstream Nature-Based Solutions Inc.

3.5.2. Mitigation Effectiveness Monitoring

Juvenile Salmonid Seasonal Habitat Use in Enhanced Off-channel Habitats

A set of pre-selected sites were sampled for juvenile salmonids in two off-channel habitats (Bluenose in Reach 4 and Applesprings in Reach 1; see Appendix A for locations and maps of these sites) at the peak of the Post-high flow hydrograph in late spring (mid June) and early summer (mid July) 2020 and 2021. Sites were open and sampled in a single pass so there was no way of determining differences in capture probability among sites or sample sessions, but we divided the 1-pass catch by the distance sampled to generate relative lineal densities (# of fish per 100 m) for each species and age class by site to facilitate some comparisons (Figure 3.35). Results from 2020 were intended to serve as a "baseline" for lower flow releases (i.e., the Trial 2 peak – 15 $m^3 \cdot s^{-1}$) against which results collected during a high(er) flow release year (i.e., 2021 - 25 $m^3 \cdot s^{-1}$) could be compared for addressing Modified Operations Management Question #5 (see Section 1.4.2). A potential shift in habitat use (from mainstem to off-channel) would be determined by detecting an increase in the lineal density of fish in the off-channel habitats that coincides with the timing of the flow increase from Terzaghi Dam and is different than the patterns in lineal densities documented under baseline conditions.

For juvenile salmonids in the off-channel habitats, coho densities were the highest, particularly in the Bluenose Outflow and Applesprings sites in June (Figure 3.35, middle plots) during both years. Site-specific densities in June 2021 were: 63, 30, 50 and 61 fish/100 m in the upper, middle, lower, and outflow channels at Applesprings, respectively, and 0 and 179 fish/100 m in the pond and outflow channel at Bluenose, respectively. Site-specific densities for coho in July 2021 were: 57, 38, 75 and 43 fish/100 m in the upper, middle, lower, and outflow channels at Applesprings, respectively, and 5 and 63 fish/100 m in the pond and outflow channel at Bluenose, respectively.

Almost all of the coho juveniles in these off-channel habitats were fry (*n*= 370 out of 373 or 99% in 2020; and *n*= 189 out of 189 or 100% in 2021).

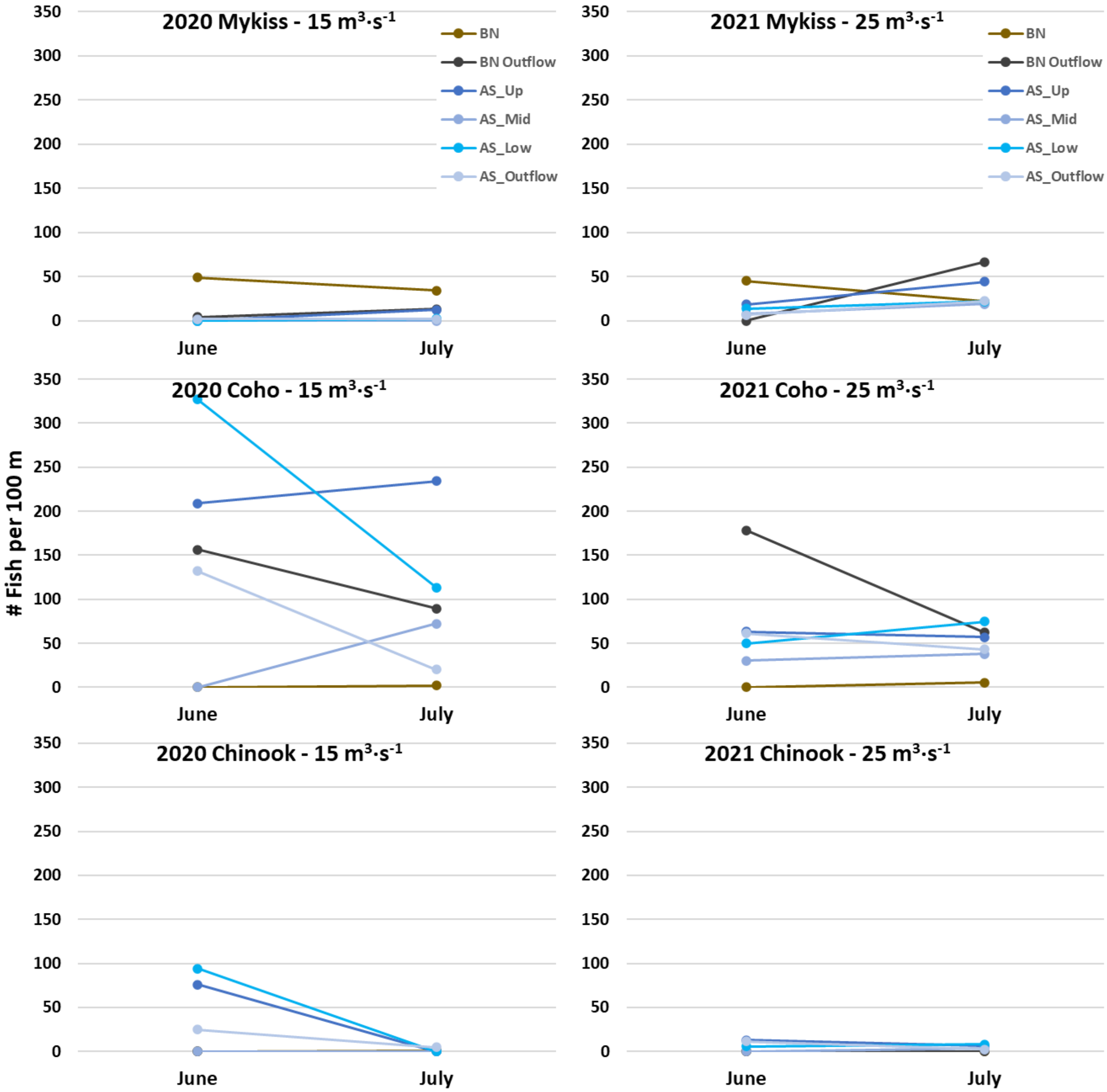


Figure 3.35 Relative numbers of mykiss (top), coho (middle), and Chinook (bottom) juveniles in various sections of two off-channel habitats during peak flow releases from Terzaghi Dam in 2020 (i.e., 15 $m^3 \cdot s^{-1}$; left column) and 2021 (i.e., 25 $m^3 \cdot s^{-1}$; right column). Site names BN = Bluenose and AS = Applesprings. Sampling occurred in mid-June and mid-July during both years to represent spring and summer rearing.

In general, the densities of juvenile coho at the Applesprings habitat were lower at higher peak flows in the mainstem during 2021 than at Trial 2 peak flows in 2020. Average densities at Applesprings were 138 fish/100 m in 2020 vs. 52 fish/100 m in 2021. Average densities at Bluenose were very similar between 2020 and 2021 (i.e., 62 fish/100 m in both years).

Chinook juvenile densities in the off-channel habitats were substantially lower than for coho, and they were almost exclusively captured in the Applesprings habitat during both years (Figure 3.35, lower plot). Site-specific densities in June 2021 were: 13, 0, 6 and 11 fish/100 m in the upper, middle, lower, and outflow channels at Applesprings, respectively, and 0 fish/100 m at both sites in the Bluenose habitat. Site-specific densities for Chinook in July 2021 were: 6, 4, 8 and 2 fish/100 m in the upper, middle, lower, and outflow channels at Applesprings, respectively, and 0 fish/100 m at both sites in the Bluenose habitat. All of the Chinook were fry based on size (*n*= 60 in 2020; and *n*= 15 in 2021).

Similar to the coho juveniles, the densities of Chinook fry at the Applesprings habitat were lower (overall) at higher peak flows in the mainstem during 2021 than at Trial 2 peak flows in 2020. Average densities at Applesprings were 25 fish/100 m in 2020 (driven by higher densities in June that year) vs. 6 fish/100 m in 2021. Average densities at Bluenose were very low during both years (i.e., <1 and 0 fish/100 m for 2020 and 2021, respectively).

Mykiss juveniles showed a slightly different pattern than the other two species (Figure 3.35, top plot). They were more prevalent in the Bluenose habitat, and their overall densities increased slightly between the June and July sessions (i.e., went up or stayed the same at 5 sites and down at 1 site) in both years. Site-specific densities in June 2021 were: 19, 8, 14 and 7 fish/100 m in the upper, middle, lower and outflow channels at Applesprings, respectively, and 45 and 0 fish/100 m in the pond and outflow channels at Bluenose, respectively. Site-specific densities for mykiss in July 2021 were: 44, 19, 22 and 23 fish/100 m in the upper, middle, lower, and outflow channels at Applesprings, respectively, and 22 and 67 fish/100 m in the pond and outflow channel at Bluenose, respectively.

Overall, the densities of juvenile mykiss were higher in the off-channel habitats in 2021 than they were in 2020. Average densities at Applesprings were 2 fish/100 m in 2020 vs. 19 fish/100 m in 2021. Average densities at Bluenose were 25 fish/100 m in 2020 vs. 34 fish/100 m in 2021. Most of the mykiss sampled in the off-channel habitats during these sessions were parr (*n*= 112 out of 118, or 95% in 2020; and *n*= 125 out of 146 or 86% in 2021); however, the slight increase in density at some sites in July was due to the arrival of the new year-class of mykiss fry (*n*= 21) which had emerged and begun colonizing the sites by the time of the July sample. This was similar to the timing of emergence noted for this species during flow trials 1 and 2 (Sneep and Hall 2012a).

Other species captured in the off-channel habitats were: dace sp. (*n*= 31), mountain whitefish (*n*= 9), redside shiner (*n*= 3), bull trout (*n*= 1), and sculpin sp. (*n*= 1), which constituted 11% of the total fish sampled (*n*= 395, all species combined). Densities and trends were not evaluated for these other species since analysis is focussed on the change in habitat use by target species (mykiss, coho and Chinook).

Juvenile Salmonid Density and Biomass in Enhanced Off-channel Habitats

As in previous years (2018-2020), the Bluenose off-channel habitat located in Reach 4 had high densities of mykiss fry (RB-0; Table 3.22), as well as a decent number of mykiss parr (RB-1) in 2021. Densities of juvenile coho (CO-0) were relatively low at Bluenose, and no Chinook juveniles (CH-0) were caught. The Applesprings habitat in Reach 1 had high densities of juvenile coho, moderate densities of mykiss, and low densities of juvenile Chinook in 2021. The Bluenose site had much higher densities of mykiss fry and parr compared to the mainstem in fall of 2020 and 2021, while the Applesprings site had much higher densities of coho fry compared to densities in the mainstem of Reach 1 in 2020 and 2021 (Figure 3.36, top vs. bottom panels).

Table 3.22 Catch, capture probability (pCap), abundance, density, and biomass of juvenile salmonids at enhanced side channel sites in fall of 2021. Note: the suffix "RI" and "PO" in the site names denotes riffle and pool habitats, respectively.

Sp-Age	Site	Catch	pCap (per pass)	Abundance	Density (#/100 m^2)	Biomass (g/100 m^2)
RB-0	Bluenose_RI	25	0.61	27	61	87
	Bluenose_PO	63	0.58	68	39	103
	Applesprings_RI	63	0.60	68	35	93
	Applesprings_PO	4	0.59	5	19	66
RB-1	Bluenose_RI	0				
	Bluenose_PO	67	0.55	74	42	577
	Applesprings_RI	7	0.52	9	6	202
	Applesprings_PO	8	0.53	9	38	878
CO-0	Bluenose_RI	1	0.57	1	3	9
	Bluenose_PO	26	0.61	28	16	81
	Applesprings_RI	69	0.69	72	39	176
	Applesprings_PO	13	0.62	14	59	324
CH-0	Bluenose_RI	0				
	Bluenose_PO	0				
	Applesprings_RI	4	0.46	6	4	41
	Applesprings_PO	0				

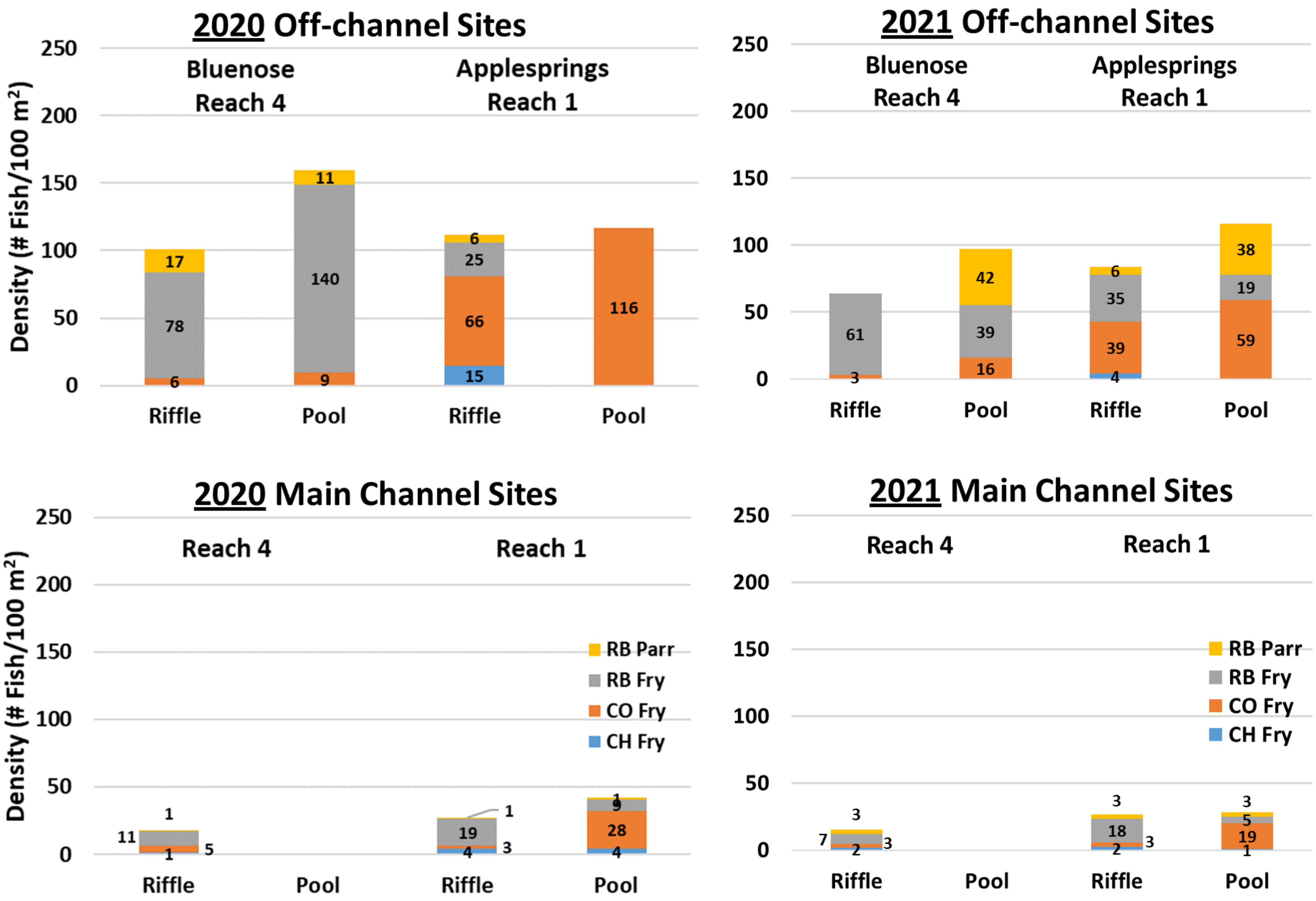


Figure 3.36 Density of juvenile salmonids by reach and habitat type in off-channel habitats (top row) during fall compared to LBR mainstem densities during the same time period (bottom row) for 2020 (left) and 2021 (right).

The densities for each species and age class in these enhanced off-channel sites in 2021 were within the same range of densities documented in these sites in 2018 (i.e., the last high flow year), 2019 and 2020 (the other Post-high flow years), and similar to densities in the mainstem LBR during trials 1 and 2, except for Reach 4 which used to have much higher densities during the flow trials (see Figure 3.17 in Sneep et al. 2020).

The sum of abundance estimates for juvenile salmonids in 2021 at Bluenose (~900 fish) and Applesprings (~10,000) was ~10,900 fish, which was ~9% of the number of fish in the mainstem across reaches 1 – 4 in 2021 (~126,000) (Figure 3.37). This off-channel abundance estimate was on par with 2020 (~11,500 fish), higher than in 2018 (~6,500 fish) and 2019 (~1,100 fish) and represented a fair contribution considering the difference in wetted area (i.e., ~1 ha for the off-channel habitats vs. ~100 ha in the mainstem for reaches 1 to 4). However, this also reveals that an area significantly larger than the Applesprings and Bluenose offchannel habitats would be required to meaningfully contribute to fish production in the Lower Bridge River and compensate for lost production in the main channel caused by high flows.

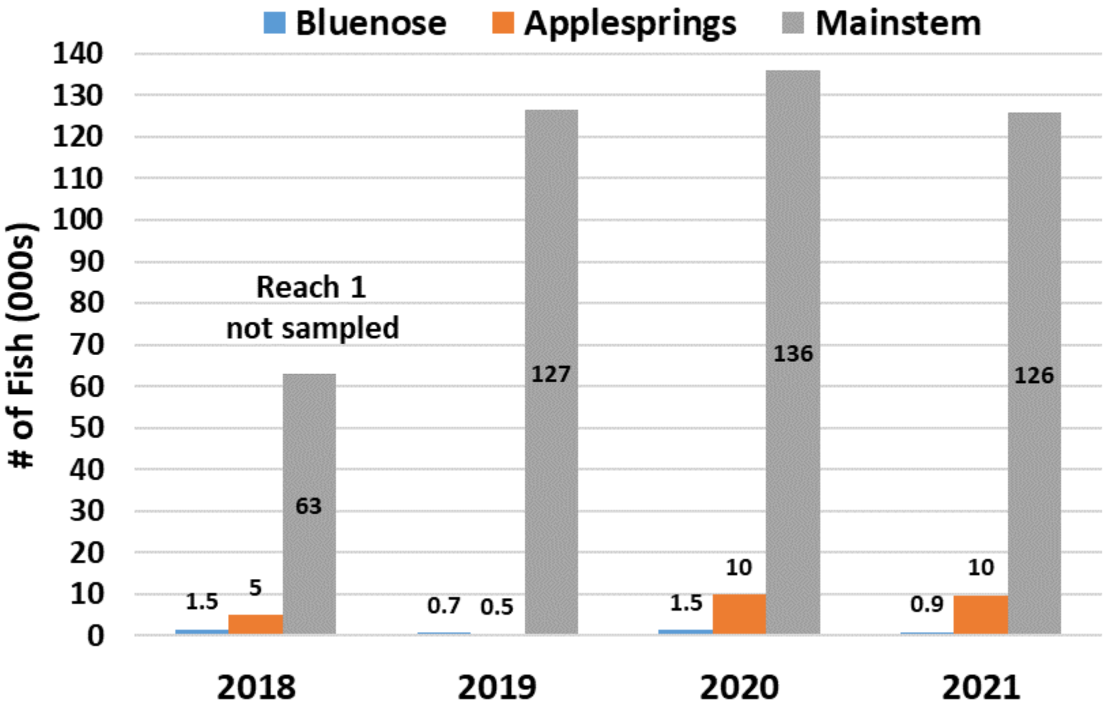


Figure 3.37 Total juvenile salmonid abundance estimates for the two off-channel habitats and the mainstem LBR (reaches 1 to 4 combined) in the four years since stock assessment sampling in the off-channel habitats began. 2018 was the last high flow year and 2019 to 2021 were Post-high flow years. Note: The 2018 mainstem abundance does not include Reach 1 which was sampled for the first time in 2019.

3.5.3. Fish Salvage and Stranding Risk Assessment

In several of the tables and figures throughout this section, comparable ramping information from the other modified operations years (i.e., ≥2016), as well as ramping results from the Trial 2 years (i.e., rampdown range 15 to 1.5 $m^3 \cdot s^{-1}$) have been included along with the 2021 results, for reference.

Ramp downs from peak flows (i.e., between 25.4 and 3.1 $m^3 \cdot s^{-1}$) occurred on ten days between 29 July and 24 August 2021, representing a total flow reduction of 22.3 $m^3 \cdot s^{-1}$ across that period (Figure 3.38 and Table 3.23). This rampdown timing was the same as in Trial 2 and during the other modified operations years (except 2020 when summer rampdowns occurred approx. 2 weeks earlier). Final ramp downs from 3.0 to 1.5 $m^3 \cdot s^{-1}$ (i.e., flow reduction of 1.5 $m^3 \cdot s^{-1}$) were completed across 4 dates between 5 and 20 October 2021, which was similar to the usual timing from previous years but required two more ramping days due to issues with the low-level outlet gates at the dam (Riley Hall, BC Hydro NRS, pers. comm.). For additional information on flow and stage changes for each rampdown event, refer to the tables in Appendix F. Total stage change at the 36.8 km compliance location was 83 cm (22 cm within the High Flow range and 61 cm within the WUP range), and the maximum daily stage change rate implemented was 6.0 cm/hr (Table 3.24) with fish salvage crews present. Some higher hourly ramp rates compared to the Trial 2 years (i.e., between 4.0 and 6.0 cm/h) were implemented in 2017, 2018, 2019 and 2021, with the intent of determining whether there is a correlation between ramping rate and the incidence of fish stranding within the survey area.

Beyond the stage monitoring conducted at the Terzaghi Dam plunge pool during ramp events by the field crews and the continuous stage elevation data recorded by a logger at the compliance location (Rkm 36.8), there were no stage logger data available for other locations downstream in 2021 (see bolded note in Section 2.2.4). As such it was not possible to evaluate the gradient of stage changes across the study area for this year. However, differences from stage monitoring results documented in previous years were not expected given that the flow release hydrograph in 2021 was within the range of flow magnitudes implemented and monitored during other modified operations years. Therefore, the findings noted in previous reports that: a) fish stranding risk is lower in reaches 1 and 2 due to the moderating influence of Yalakom inflows on stage changes; and, b) that fish salvage at discharges ≤15 $m^3 \cdot s^{-1}$ is not required in these reaches, were still considered appropriate.

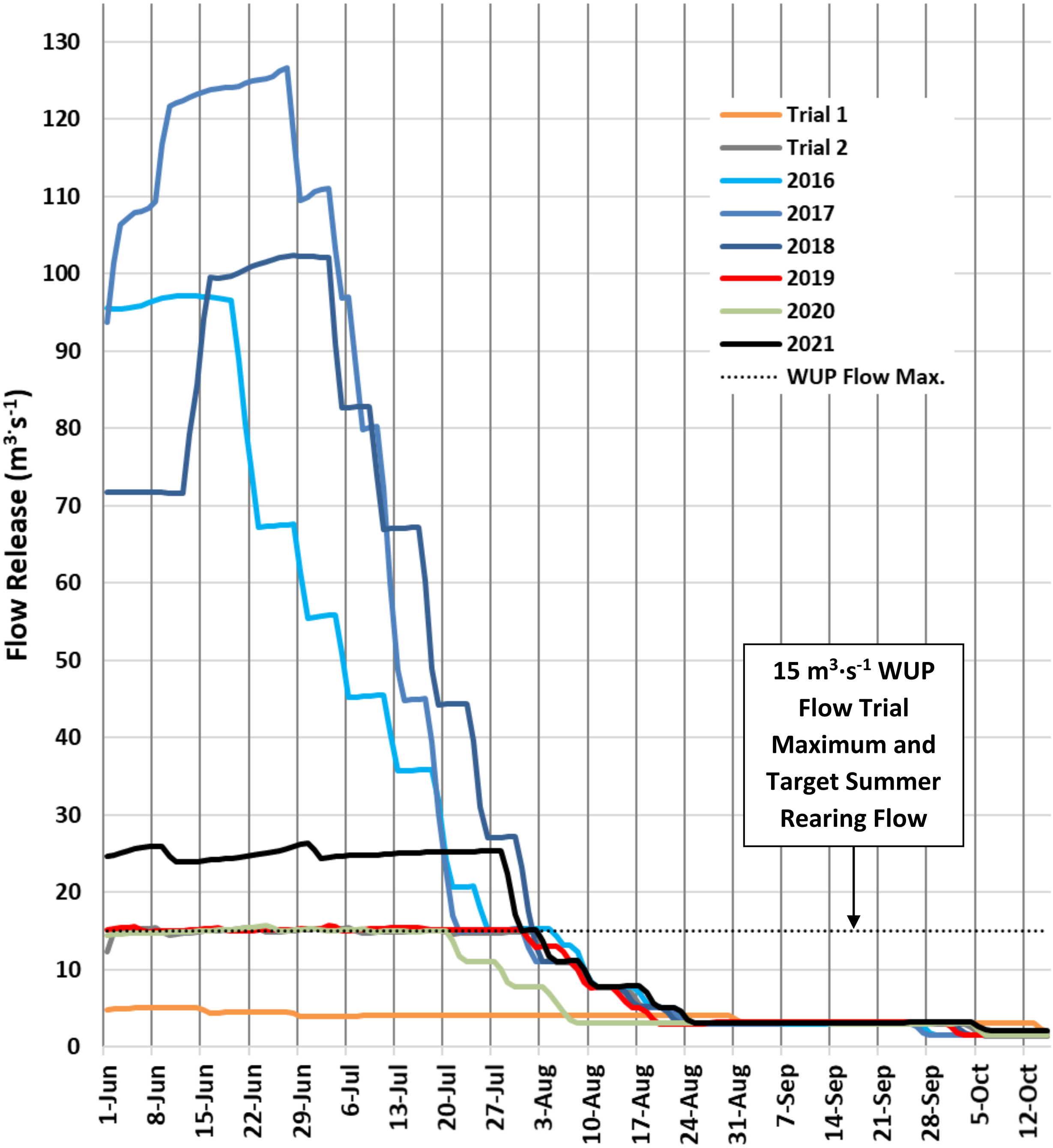


Figure 3.38 **Schedule of flow releases and ramp downs from the peak period to the start of the fall low flow period in 2019, 2020 and 2021. For reference, WUP Trial 1 and 2 flow releases as well as the 2016–2018 high flow years are shown for the same period.**

Table 3.23 Summary of total stage changes at available monitoring locations in reaches 2, 3 and 4 for each rampdown event in 2021.

Year	Date	Event #	Start Flow ($m^3 \cdot s^{-1}$)	End Flow ($m^3 \cdot s^{-1}$)	Flow Change ($m^3 \cdot s^{-1}$)	Total Stage Change (cm)				
						Plunge Pool (Rkm 40.9)	Top of Reach 3 (Rkm 36.8)[a]	Bottom of Reach 3 (Rkm 26.0)[b]	Horseshoe Bend (Rkm 23.6)[b]	Bottom of Reach 2 (Rkm 20.0)[b]
2021	29-Jul	1	25.4	20.0	-5.4	-17	-11	-	-	-
	30-Jul	2	20.0	15.0	-5.0	-19	-11	-	-	-
High Flow Rampdown Summary			**25.4**	**15.0**	**-10.4**	**-35**	**-22**	**-**	**-**	**-**
2021	3 Aug	3	15.1	12.9	-2.2	-10	-5	-	-	-
	4 Aug	4	13.0	11.1	-1.9	-8	-4	-	-	-
	9 Aug	5	11.2	9.3	-1.9	-10	-6	-	-	-
	10 Aug	6	9.3	7.8	-1.5	-9	-5	-	-	-
	18 Aug	7	7.9	6.4	-1.5	-7	-5	-	-	-
	19 Aug	8	6.4	5.1	-1.3	-7	-6	-	-	-
	23 Aug	9	5.1	4.2	-0.9	-12	-5	-	-	-
	24 Aug	10	4.2	3.1	-1.1	-11	-8	-	-	-
	5 Oct	11	3.2	2.3	-0.9	-12	-8	-	-	-
	6 Oct	12	2.3	2.1	-0.2	-3	-1	-	-	-
	19 Oct	13	2.3	2.2	-0.1	-2	-1	-	-	-
	20 Oct	14	2.1	1.3	-0.8	-5	-8	-	-	-
WUP Rampdown Summary			**15.1**	**1.3**	**-13.8**	**-96**	**-61**	**-**	**-**	**-**

[a] This location represents the compliance location for stage changes associated with ramp down events.

[b] Stage monitoring data for this location were not available in 2021 – see bolded note in Section 2.2.4 for explanation.

Table 3.24 Summary of flow ramp down events across the high flow range (>15 $m^3 \cdot s^{-1}$) and "normal" Trial 2 range (≤15 $m^3 \cdot s^{-1}$) during Modified Operations years (2016–2021). Note: Flow releases did not exceed the WUP flow targets in 2019 or 2020 so high flow ramp downs were not required in those years. For more details on individual events refer to the tables provided in Appendix F.

Period	Year	Month(s)	# of Ramping Days	Total Flow Reduction ($m^3 \cdot s^{-1}$) [a]	Total Stage Change (cm) [b]	Maximum Hourly Rate (cm/hr) [c]
High Flow Ramp Events (>15 $m^3 \cdot s^{-1}$)	**2021**	**Jul**	**2**	**-10.4**	**-22**	**-5.0**
	2018	Jul	8	-86.9	-122	-4.0
	2017	Jun – Jul	9	-96.5	-143	-4.1
	2016	Jun – Jul	8	-81.4	-108	-2.3
"Normal" Ramp Events (≤15 $m^3 \cdot s^{-1}$)	**2021**	**Aug, Oct**	**12**	**-13.8**	**-61**	**-6.0**
	2020	Jul/Aug, Oct	10	-13.6	-57	-2.7
	2019	Aug, Oct	10	-13.7	-60	-4.6
	2018	Aug, Oct	9	-13.6	-62	-2.0
	2017	Aug, Sep	9	-13.7	-67	-2.6
	2016	Aug, Sep	10	-13.8	-67	-3.0

[a] Measured at the dam; data provided by BC Hydro.

[a] Measured at the top of Reach 3 (Rkm 36.8; aka the compliance location). Data provided by BC Hydro.

[b] Measured at the Plunge Pool (Rkm 40.9) by manual staff gauge readings.

Coupling the BC Hydro flow release records with the continuous river stage level recorded at 36.8 km (known as the compliance location for tracking ramp rates) enabled characterization of the discharge-stage relationship at that location (Figure 3.39). The plot includes the stage-discharge data points for every year since 2016 (the start of the high flows).

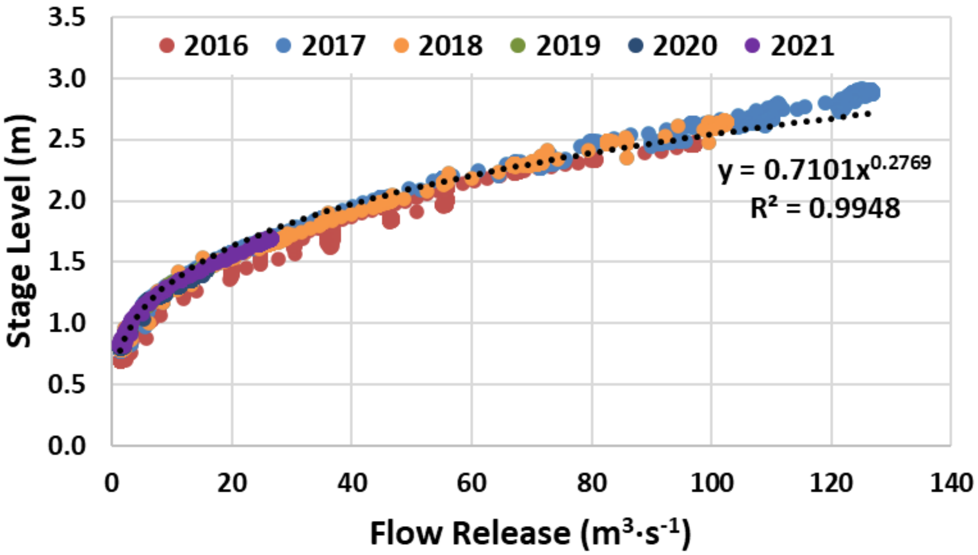
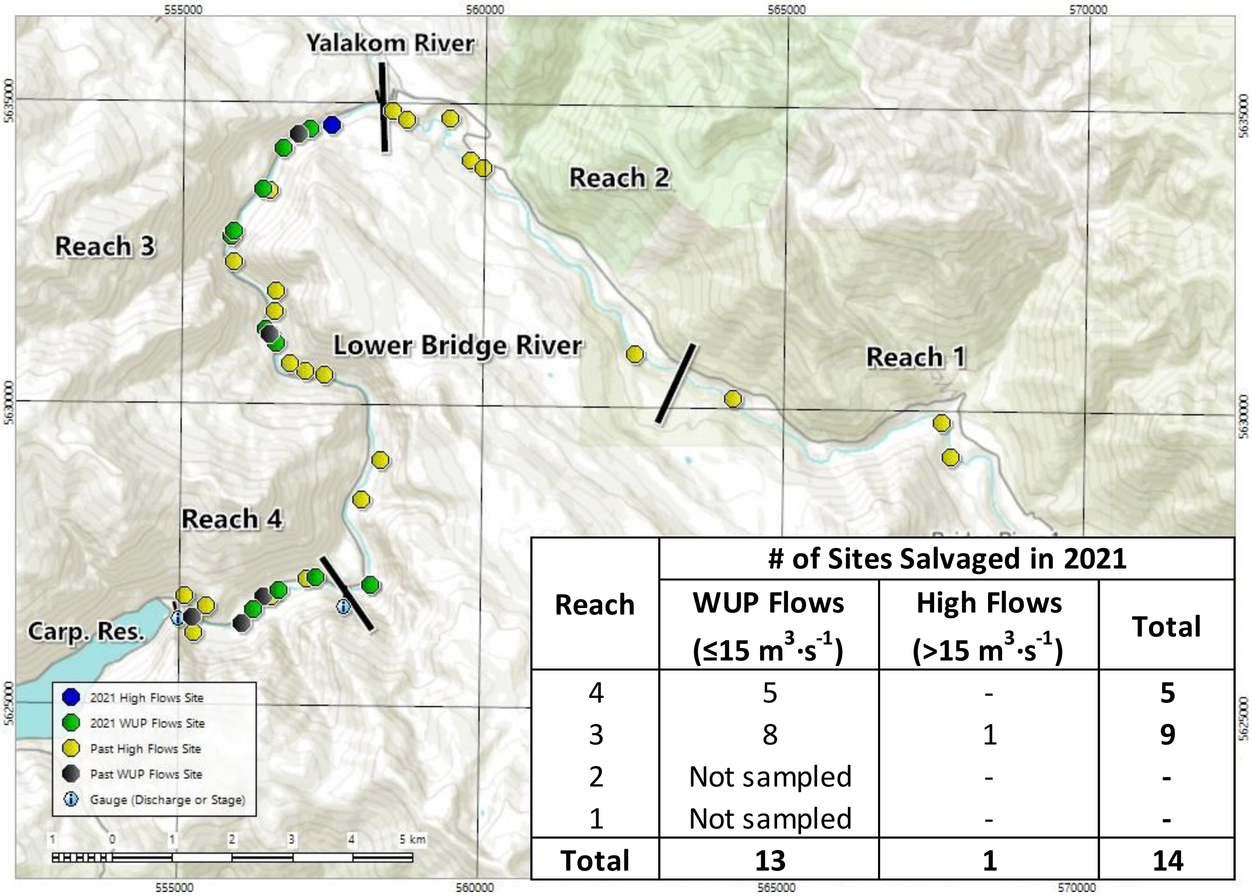


Figure 3.39 **Discharge-stage relationship at 36.8 km (the compliance location) across the range of flows observed across all flow treatments. Separate data points for each Modified Operations monitoring year (2016-2021) are shown.**

The curve drawn through the points has a good fit ($R^2 = 0.995$), such that the associated equation ($y = 0.7101x^{0.2769}$) may be useful for predicting stage changes for particular flow changes within this range. Stage values in 2016, particularly for discharges between approx. 1.5 $m^3 \cdot s^{-1}$ and 55 $m^3 \cdot s^{-1}$ tended to be slightly lower than during the other years, possibly due to some channel changes at the gauging location that occurred with the high peak flows in 2017 or 2018, so the current curve is based on the 2017-2021 data points. The curve may underestimate stage elevations for discharges >100 $m^3 \cdot s^{-1}$. It is clear from the relationship that the greatest degree of stage changes occurs at the lowest discharges (i.e., the initial slope is the steepest). Above ~10 $m^3 \cdot s^{-1}$ the slope begins to decrease, and the discharge-stage relationship becomes close to linear across the higher flows.

As a result of the reconnaissance surveys conducted for the 2021 high flow (modified operations) range between 25 and 15 $m^3 \cdot s^{-1}$, sites with the potential for fish stranding were noted at 4 locations (i.e., 2 sites in Reach 4, 1 in Reach 3, and 1 in Reach 1). However, fish salvaging was only conducted at 1 site during the 2021 high flow ramp downs (i.e., "Fish Counter" site at ~Rkm 26.1 in Reach 3) (Figure 3.40). The number of sites requiring active fish salvaging within the WUP (Trial 2; 15 to 1.5 $m^3 \cdot s^{-1}$) range in 2021 was 13, which was 7 fewer than the 20 locations that had been identified during the previous Trial 2 and High flow years (2011 to 2018; for reaches 3 and 4 only). In some cases during the Post-high flow years, field crews had noted that channel morphology or flow conditions had changed at some locations following the years of high flows, which resulted in the identification of four new sites in 2019 and rendered some others obsolete. An additional 35 locations were identified for the high flow range (>15 $m^3 \cdot s^{-1}$) across all four reaches of the Lower Bridge River during 2016–2018; however, fish salvaging may not have been required at most of these sites in 2021 because the high flow magnitudes were much lower.



Reach	# of Sites Salvaged in 2021		
	WUP Flows ($\leq 15\ m^3 \cdot s^{-1}$)	High Flows ($>15\ m^3 \cdot s^{-1}$)	Total
4	5	-	**5**
3	8	1	**9**
2	Not sampled	-	**-**
1	Not sampled	-	**-**
Total	**13**	**1**	**14**

Figure 3.40 Survey area map for ramp monitoring and fish salvage on the Lower Bridge River showing fish salvage locations for WUP flows in 2021 (green dots), and Modified Operations high flows (blue dots). Identified salvage sites from past years (which were not sampled in 2021) are shown as black and yellow dots for WUP and high flow levels, respectively. Discharge and stage gauging locations are represented by the blue information symbol (i). Solid black lines represent the reach breaks. A table summarizing the number of sites in 2021 is also included (inset).

Including the 2021 data points, fish salvage numbers for the ramp downs across the Trial 2 range ($\leq 15\ m^3 \cdot s^{-1}$) were generally higher relative to the results for the high flow range ($>15\ m^3 \cdot s^{-1}$; Figure 3.41). In previous years (≤2016), crews had noted incidental catches (fish salvaged before their strand-risky habitat had become isolated from the main channel flow or dewatered); however, for consistency with the 2017–2020 results, these incidental catches were not included in the analyses or the results reported here. It should also be noted that the results shown on the plots in this section have not been standardized for abundance (i.e., based on stock assessment results for each year). This was not done for the following reasons: 1) abundance data are only collected in the fall after most of these ramp downs have occurred so we don't know what the pre-ramp abundances were (i.e., we would be assuming that the incidence of fish stranding has no effect on abundance before vs. after ramp events); and, 2) since these data are for modified operations years (2016-2021), the abundances were quite similar for each year (see Figure 3.28) so standardizing by abundance does not change the plots (or conclusions derived from them) in any substantive way.

Inclusive of the results from all available survey years, there appears to be a fairly distinct flow threshold where the fish stranding risk transitions from high risk (>100 fish per 1 $m^3 \cdot s^{-1}$ flow change) to moderate or low risk (10 to 99, and <10 fish per 1 $m^3 \cdot s^{-1}$ flow change, respectively), as defined in the Fish Stranding Protocol for the Lower Bridge River (Sneep 2016). This threshold flow appears to be at ~13 $m^3 \cdot s^{-1}$, similar to the flow value where the transition in the stage-discharge relationship occurs (see Figure 3.39), which was again apparent from the 2021 data. However, it must also be noted that substantially lower abundance of juvenile fish (particularly coho and steelhead fry that are generally the most vulnerable to stranding) were documented for all three high flow years and in 2021 (see Table 3.16). Relative to the Trial 2 averages, abundance of coho and steelhead fry was down by 90% and 80%, respectively, during the high flow years and in 2021. As such, the confounding effect of low abundance (due to displacement out of the survey area or poor survival) on the high flow fish salvage results cannot be ruled out.

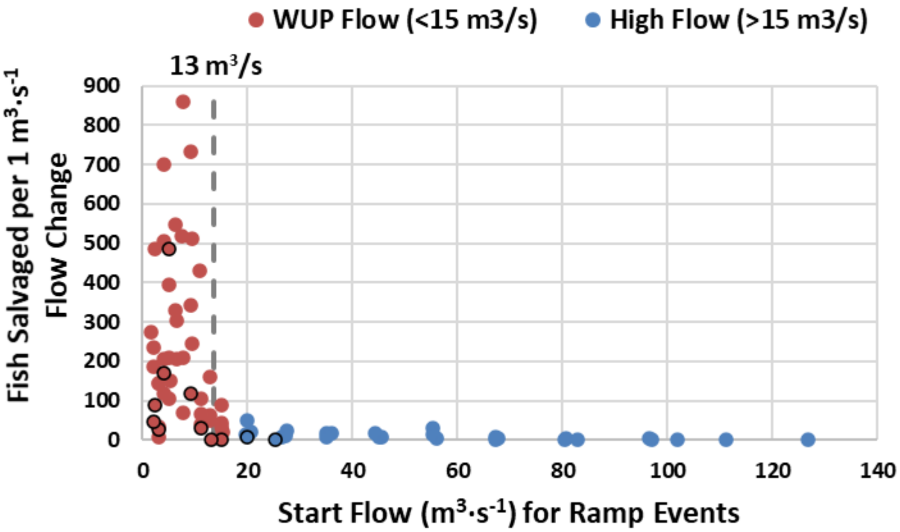


Figure 3.41 Relative differences in number of fish salvaged per increment of flow change for ramp downs from high flows (>15 $m^3 \cdot s^{-1}$) versus Trial 1 and 2 flows (≤15 $m^3 \cdot s^{-1}$). The vertical dashed line represents the approximate flow threshold (~13 $m^3 \cdot s^{-1}$) where the apparent break between high stranding risk and moderate or low stranding risk occurs. Circles with black border represent 2021 data; Plain circles are data from previous years.

Compared to survey results from the previous Trial 1 and Trial 2 years, relatively large areas of fish stranding habitat were documented within the high flow range (2018 total = 66,892 m^2 between 102 and 15 $m^3 \cdot s^{-1}$), primarily due to the addition of stranding site reconnaissance and salvage surveys in reaches 2 and 1. Note: only the area values for the most recent year during the High Flow treatment are presented for comparison because stranding area changed to some degree with each consecutive high flow event. Potential stranding area contribution by reach was 4,887, 9,105, 22,900 and 30,000 m^2 for reaches 4, 3, 2, and 1, respectively. Stranding area identified within the 25 to 15 $m^3 \cdot s^{-1}$ high flow range in 2021 was only 5 m^2 and it was entirely in Reach 3. Under the trial flow range (≤15 $m^3 \cdot s^{-1}$) in 2021 when only reaches 3 and 4 were surveyed, the total stranding area was 4,170 m^2 (compared to 13,992 m^2 for those two reaches at the high flow range in 2018), and stranding area was again more prevalent in Reach 3 than Reach 4 (3,307 and 863 m^2, respectively).

Within the high flow range (>15 $m^3 \cdot s^{-1}$) in 2021, the highest proportion of salvaged fish per stranding habitat area was in Reach 3 (780 fish per 100 m^2; Table 3.25), which was a very high density relative to past results. However, all of these fish (*n*= 39) were salvaged out of a single 5 m^2 isolated pool near the BRGMON-3 fish counter site. Given the very small sample area relative to previous high flow ramp downs, this 2021 result is not considered representative of the relative fish stranding risk among the reaches for the high flow range. Fish were not salvaged out of the other reaches at flows >15 $m^3 \cdot s^{-1}$ in 2021.

Within the WUP flow range (≤15 $m^3 \cdot s^{-1}$) in 2021, fish salvage numbers were much greater (*n*= 936) than at the high flow range but the densities were lower given the much greater area sampled. Stranding densities were higher in Reach 3 followed by Reach 4 (25 and 13 fish per 100 m^2, respectively). These values were at the low end of the range documented for the ≤15 $m^3 \cdot s^{-1}$ flows under the previous flow treatments that were characterized by much higher juvenile salmonid abundance (i.e., Trials 1 & 2 means = 81 (range = 51 to 123) and 63 (range = 48 to 75) fish per 100 m^2 of salvaged area in reaches 3 and 4, respectively; Sneep 2016). Reaches 1 and 2 have not been surveyed within the WUP flow range.

Table 3.25 Summary of fish stranding area and numbers of fish salvaged by reach for the high flow (>15 $m^3 \cdot s^{-1}$) and WUP flow (≤15 $m^3 \cdot s^{-1}$) ranges in 2021. Note: As in past years, there was no data for fish stranding or salvage in reaches 1 and 2 under the WUP flows.

Flow Range	Reach	# of Sites	Area (m^2) (% Contribution)	# of Fish	# of Fish per 100 m^2
2021 High Flows (>15 $m^3 \cdot s^{-1}$)	4		- (0%)	-	-
	3	1	5 (100%)	39	780
	2		- (0%)	-	-
	1		- (0%)	-	-
High Flow Totals		**1**	**5**	**39**	**780**
2021 WUP Flows (≤15 $m^3 \cdot s^{-1}$)	4	5	863 (21%)	114	13
	3	8	3,307 (79%)	822	25
	2	------ No data ------			
	1	------ No data ------			
WUP Flow Totals		**13**	**4,170**	**936**	**22**

With the benefit of fish salvage crews on the ground, some higher ramp rates (up to 6.0 cm/hr) were implemented in 2017, 2018, 2019 and 2021. In prior years, and in 2020, most ramp rates conformed to the ≤2.5 cm/hr threshold specified in the WUP (for when fish salvage crews are not present), even though crews were routinely deployed during all of those events. Based on the sample size available from 2017, 2018 and 2021, the higher ramp rates employed for ramp downs within the high flow range (>15 $m^3 \cdot s^{-1}$) did not increase the incidence of stranding at the flow levels tested (Figure 3.42). This suggests that for flows >15 $m^3 \cdot s^{-1}$ it may be possible to increase the ramp rate above the WUP threshold without unduly increasing the fish stranding risk.

For ramp downs within the Trial 2 range, the faster ramp rates implemented in 2019 and 2021 were associated with greater incidence of stranding than for the high flows but were still within the range of salvage numbers for ramp rates ≤2.5 cm/hr for the other Trial 2 years. Collectively, these results suggest there can be some flexibility for strategically ramping flows down more quickly than would be possible using the WUP rate alone (i.e., to reach more optimal summer rearing flows, for instance). However, it is not possible to rule out the confounding effect of reduced abundance of the most strand-risky fish (i.e., coho and mykiss fry) in 2016 – 2021 relative to the Trial 2 years, as mentioned above. Testing faster ramp rates during years when juvenile salmonid abundance is higher (e.g., similar to Trial 1 or 2 levels) would be necessary to address this uncertainty.

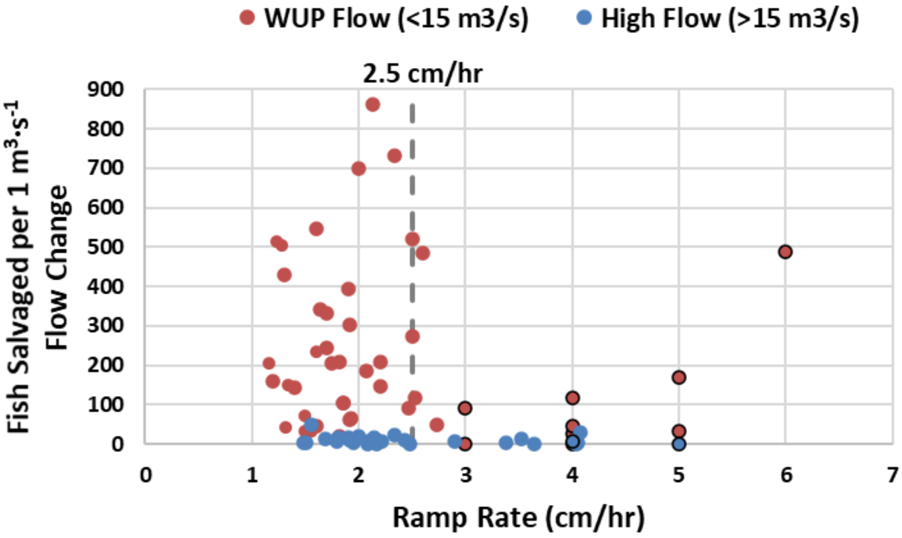


Figure 3.42 Relative incidence of fish stranding per increment of flow change according to different ramping rates under high flow (>15 $m^3 \cdot s^{-1}$; blue circles) and WUP flow (≤15 $m^3 \cdot s^{-1}$; red circles) ranges. The vertical dashed line depicts the ramp rate (2.5 cm/hr) specified in the WUP when fish salvage crews are not present. Circles with black border represent 2021 data; Plain circles are data from past years.

Factoring in the reduced number of locations identified in 2021, the proportions of identified stranding sites on river left (76%) versus river right (24%) were not equal under the trial flows

(≤15 $m^3 \cdot s^{-1}$), even though both banks were accessible to fish salvage crews across a significant part of that range (Table 3.26). Note that these proportions are based on reaches 3 and 4 only as reaches 1 and 2 were not surveyed at flows below 15 $m^3 \cdot s^{-1}$. Across the high flow range (>15 $m^3 \cdot s^{-1}$), the distribution was closer to equal with 44% on river left and 56% on river right based on the site reconnaissance conducted by staff from CNBS (formerly Coldstream Ecology Ltd.) during 2016 – 2018.

Table 3.26 Proportions of sites on the river left bank versus the river right bank for WUP flows (≤15 $m^3 \cdot s^{-1}$; based on reaches 3 and 4 only) and high flows (>15 $m^3 \cdot s^{-1}$; based on new site reconnaissance surveys during the High Flow years).

Flow Range	Left Bank		Right Bank	
	n	%	n	%
Trial Flows (≤15 $m^3 \cdot s^{-1}$) *Reaches 3 & 4 only	13	76%	4	24%
High Flows (>15 $m^3 \cdot s^{-1}$) *New Site Reconn.	10	45%	12	55%
All	**23**	**59%**	**16**	**41%**

As identified for past fish salvage surveys under flow trials 1 and 2 (≤15 $m^3 \cdot s^{-1}$), coho and mykiss were the most frequently encountered species during the High Flow treatment (2016-2018; Sneep et al. 2019), and again during the Post-high flow years (2019-2021). Coho made up 50% of the total catch in 2019, 45% in 2020, and 33% in 2021 (Table 3.27). The contribution of mykiss fry to the salvage totals was 47% in 2019, 49% in 2020, and 67% in 2021. The contribution of other species, including Chinook juveniles, have been low every year (i.e., ≤ 5%). As noted in the Fish Stranding Protocol (Sneep 2016), coho and mykiss fry tend to be the most vulnerable to stranding because the habitat types preferred by this age class of these species (e.g., shallow edge areas and side channels/pools) are also among the habitat types that are most likely to dewater and result in fish stranding when flows are reduced. Fry may also remain in these habitats even as flows are dropping because they are less able to exploit deeper offshore areas where there are typically higher velocities, less cover, and increased risk of predation.

Table 3.27 Summary of numbers (and %) of fish salvaged by species-age class under High flow ramp downs (>15 $m^3 \cdot s^{-1}$) in 2021, and WUP flow ramp downs (≤15 $m^3 \cdot s^{-1}$) in 2019-2021.

Species	High Flows (>15 $m^3 \cdot s^{-1}$)	WUP Flows (≤15 $m^3 \cdot s^{-1}$)		
	2021	2019	2020	2021
Chinook	-	13 (1%)	29 (1%)	13 (1%)
Coho	18 (46%)	1,012 (50%)	1,796 (45%)	299 (32%)
Mykiss	21 (54%)	948 (47%)	1,992 (49%)	616 (66%)
Other spp.[a]	-	48 (2%)	208 (5%)	8 (1%)
All	**39**	**2,021**	**4,025**	**936**

[a] Other species included: bull trout, mountain whitefish, redside shiner, and sculpins.

The least abundant of the target salmonid species in the salvage results were Chinook fry, which were encountered more frequently in Reach 3 (2021 *n*= 12) than in Reach 4 (2021 *n*= 1) but contributed ≤1% to the total number of fish salvaged in every Post-high flow year overall. Chinook fry can occupy some of the same habitats as coho and steelhead fry, but they tend to be larger (because they emerge earlier in the year) so they can exploit habitats further from the river margins that are less likely to dewater. Also, they have been much less abundant in the study area overall since the flow trials began, and particularly in reaches 3 and 4 (see Section 3.4.2).

Other species in the fish salvage catches were: bull trout (2019 *n*= 2; 2020 *n*= 13; 2021 *n*= 2), mountain whitefish (2019 *n*= 2; 2020 *n*= 30; 2021 *n*= 3), redside shiner (2019 *n*= 36; 2020 *n*= 157; 2021 *n*= 0), and sculpin spp. (2019 *n*= 8; 2020 *n*= 8; 2021 *n*= 3). The bull trout and mountain whitefish were almost exclusively salvaged in Reach 3 and the redside shiner and sculpin spp. were most prevalent in the Reach 4 catches. The low numbers of these species in the ramp down results relative to the target salmonid species was likely due to lesser abundance in the survey area, lower proclivity to stranding, or a combination of both. For the specific catch totals by species for each rampdown event, refer to Table F6 in Appendix F.

4. Discussion

4.1. Management Question 1

How does the instream flow regime alter the physical conditions in aquatic and riparian habitats of the Lower Bridge River ecosystem?

The data collected in Year 10 (2021) added another set of results for the Post-High Flow period, which started in 2019. Flows in 2019 and 2020 followed the Trial 2 hydrograph throughout the year, and the 2021 hydrograph was generally the same except flow releases during the peak period (May to July) were approx. 25 $m^3{\cdot}s^{-1}$ instead of the Trial 2 peak of 15 $m^3{\cdot}s^{-1}$. Peak flow releases have been more variable during the modified operations years than during the previous flow trials. One of the goals of monitoring during the Post-High Flow years was to document how physical conditions, under a return to lower flow releases, compared to the previous Trial 2 years (i.e., 2011 to 2015) when flow releases were similar, as well as conditions under the preceding High flow years (2016 to 2018).

During the high spill years in 2016-2018, the volume of the Terzaghi flow releases during the peak flow period defined the physical and water chemistry of the entire Lower Bridge River. Terzaghi discharges during the peak flow period in 2016 – 2018 were 6- to 8-fold higher than the Trial 2 peak and were 3- to 7-fold higher than peak Yalakom inflows. These high flows had impacts on physical conditions within the study area including changes to wetted area, depths, velocities, water temperature, turbidity, bank erosion and substrate deposition (Sneep et al. 2019; Sneep et al. 2018; McHugh et al. 2017). Outside of the peak period, flow releases were the same as Trial 2 and in-season effects on physical conditions during those periods were the same as reported previously for Trial 2 (Soverel and McHugh 2016).

As reported by Ellis et al. 2018, the high flows were of sufficient magnitude to mobilize sediments in the river channel, coarsen the substrate in identified spawning areas (although subsequent monitoring has suggested that spawning habitat area is not limited relative to escapement sizes in the study area; Davey 2019), recruited sediments from active colluvial fans, and impacted embeddedness (increased interstitial pore depth but reduced pore density) within their monitored sites. For several of these measures there was a gradient of effect with greatest impacts closest to the dam, as well as spatial variability of both sediment grain size distributions and applied shear stress from the high flows. These changes associated with the 2016-2018 high flows (coupled with a declining trend in soluble-reactive phosphorus concentrations from upstream sources – see Figure 3.11) have altered physical habitat conditions in the study area relative to the pre-high flow conditions (i.e., during Trial 2) that may affect post-high flow recovery of the aquatic ecosystem and juvenile fish recruitment. This is the primary reason that conditions under the return to lower flows in 2019 to 2021 cannot be considered equivalent to the pre-high flow Trial 2 conditions even though the flow releases were similar. However, with only three years of Post-high flow data from 2019 to 2021 available to-date, and more variable peak flows across the modified operations period (2016 to 2021) than the earlier flow trials, understanding the linkages between these impacts and aquatic ecosystem recovery remain challenging.

The Post-high flow sediment composition was characterized at the BRGMON-1 index monitoring locations (sites A-K in the LBR, and sites Yal_A and Yal_B in the Yalakom River) using the Wolman pebble count method in 2019 and 2021. There were differences in mean particle sizes among the reaches based on those two sets of data. In 2019 there was a diminishing gradient of median particle size from upstream to downstream (i.e., 11.7 ±0.0 cm in Reach 4 to 7.3 ±2.5 cm in Reach 1). However, based on the 2021 pebble count data, an opposite gradient of median sizes was apparent (i.e., 7.6 ±0.0 cm in Reach 4 to 13.2 ±5.0 cm in Reach 1). Smallest D_{50} values were in the Yalakom River in both years (i.e., 3.4 ±0.9 cm in 2019 and 5.2 ±0.7 cm in 2021). Given that flow release magnitudes between 2019 and 2021 were not likely high enough to substantially change the substrate composition at the monitoring locations, the differences in D_{50} values were more likely due to measurement error or methodological differences. Regardless, the data in both cases confirmed that the sizes available at each of the sites were within the range of substrate classes that can support invertebrate colonization, salmon spawning and juvenile salmonid rearing. Further discussion of these results is included in the response to management question (MQ) #2, which follows.

Site-specific discharge estimates highlighted that flow conditions among sites in reaches 3 and 4 differed minimally throughout the year. The magnitude of combined tributary and groundwater inflows were relatively small in these reaches. However, the Yalakom River is a substantial contributor of inflow at the top of Reach 2, particularly during the peak flow period, and this was again evident in 2021. As such, the estimated discharge rates for sites in Reach 2 were substantially higher (by 1.2 to 5.0x depending on the period of the year) than the estimates for the Reach 3 and 4 sites. During years with lower flow releases from Terzaghi Dam (relative to the High Flow years) including 2021, the Yalakom River inflow contributes an increased proportion of the total discharge in the lower reaches which dilutes or masks some of the physical and water chemistry characteristics of the release.

Under the flow release hydrograph implemented in 2021, river stage elevations varied by a total of 0.79 m between the spring peak and the winter low flows at the Rkm 36.8 monitoring location, which was 0.20 m higher than the average stage range under the Trial 2 hydrograph implemented in 2019 and 2020. However, this was well below the total stage change of up to 2.10 m during the High Flow years. Given that stage data were not available for the other monitored locations in 2021 (see explanation in Section 2.2.4), it was not possible to characterize the relative changes in the other reaches. However, it is assumed that the gradient of effect with distance from the dam described in previous Post-high flow reports (i.e., Sneep et al. 2021) similarly applied in 2021.

Under the high flows from 2016 – 2018, the added discharge increased river stage by between 1.08 – 1.43 m above the Trial 2 peak (at the Rkm 36.8 site), but also reduced the proportional area of rearing habitat by increasing velocities beyond levels that juvenile fish could withstand throughout more of the channel. This also may have been the case to some extent at the "higher than Trial 2 peak flows" in 2021. However, it was not possible to measure depths and velocities in mid-channel at these flow magnitudes using conventional field methods. Specific assessment of depths and velocities and changes to habitat area that meets rearing criteria will have to come from analysis of the Telemac 2D model outputs.

The water temperature profiles by reach during the Post-high flow years generally followed the patterns observed in other flow treatments: cooler temperatures in spring and warmer in the fall relative to the Pre-flow period (Trial 0) with a gradient of effect associated with distance from the dam. However, average water temperatures in late spring and summer were warmer than during trials 1 and 2, which was most evident in reaches 3 and 4, but lower than the High Flow treatment for this seasonal period. The primary cause of the warmer water temperatures during the Post-high flow years were attributed to ambient temperature influence. Mean monthly air temperatures (recorded in Lillooet, BC) were warmer in May 2019, September 2020 and June-July 2021 than they were in those months during the previous flow treatment periods.

The warmer temperatures in spring and summer may also be due to an effect of "Modified Operations" on temperature profiles in Downton and Carpenter reservoirs (which are characterized by more frequent deep drawdowns and a reduced maximum fill elevation on Downton), since ambient temperature effects do not entirely explain the differences observed among treatments. An operations-related effect was also observed during the High flow years and to a greater extent (Sneep et al. 2019). Nonetheless, the spring and summer temperatures recorded during the Post-high flow years were still within optimal ranges reported in the literature for steelhead spawning and incubation, and rearing for each salmonid species (Brett 1952, Bjornn and Reiser 1991, Oliver and Fidler 2001).

Turbidity data were continuously recorded by loggers manufactured by RBR from March to November in 2020 and 2021. The pattern that emerged from the 2021 data was relatively low turbidity (<10 NTU) in the mainstem LBR at the end of the winter period, rising to an initial peak in spring (up to 80 NTU in Reach 4 which declined to 60 NTU at the bottom of Reach 3), declining to approx. 20-30 NTU in early summer, rising to a second peak (40-60 NTU) in August, followed by a third peak of similar magnitude in the fall as Carpenter Reservoir becomes isothermic and there is mixing among the depth layers, and then declining again to the winter low (<10 NTU) in November and December. A similar pattern of turbidity was documented in 2020, except there were only two peaks (in spring and fall) that year (see Sneep et al. 2021). Patterns of turbidity within the study area are primarily driven by the following factors: a) snow-melt (in the spring), b) Carpenter reservoir operations (low in the spring, filling and becoming thermally stratified in the summer, and then turning over in the fall), and c) inputs from the tributaries, primarily the Yalakom River, which have high turbidity during the freshet period (May to July) and low turbidity at other times of year (except during periodic rainfall events). These patterns and their relationships with flow releases and reservoir operations will be further fleshed out if additional turbidity monitoring data from subsequent monitoring years becomes available.

4.2. Management Question 2

How do differences in physical conditions in aquatic habitat resulting from the instream flow regime influence community composition and productivity of primary and secondary producers in the Lower Bridge River?

4.2.1. Design

When making decisions about preferred flows for ecological values in the Bridge River, a technical working group suggested that benthic invertebrate diversity and abundance can be a useful proxy for river health (Failing et al. 2012). People believed that benthos provide insight into ecological processes centered not only on food for fish, particularly salmonids that are highly valued (Quinn 2018), but also on overall ecological processes. This belief is well founded in the scientific literature wherein the term, “river health” refers to measurements that show whether a river has sustainable and resilient structure and function (Costanza and Mageau 1999). Benthic invertebrates are particularly good indicators of these attributes because there are many taxa occupying many functional capacities (Cummins and Merritt 1996), they are relatively sedentary (data not confounded by movement), they are easily sampled, and responses to many types of disturbances are known among taxa (Norris and Thoms 1999). Links between benthic invertebrates and combinations of physical and chemical conditions are the basis of bioassessment that is favoured for testing river quality worldwide (e.g., Bailey et al. 2004, 2012; Nichols et al. 2014). The same arguments apply to algal periphyton. They have short life cycles, making them responsive to change in habitat, they are the first organisms to respond to environmental stress, and the first to recover from it (Lowe and Pan 1996, Smucker et al. 2013). In combination, periphytic algae and benthic invertebrates are ideal indicators of change to structure and function of the Lower Bridge River in relation to manipulation of flow.

In the absence of a control reach not affected by different flow releases from the dam we were limited to contrasting Trials, blocked by Reach and Pinks on/off. Testing a Trial effect was similar to testing change in a response variable in a before - after layout that does not control for time. Given the large differences in flow regime between Trials and no known regional change over years other than aspects of climate that would have affected all reaches the same way, this approach was considered reasonable. There is also a precedent. Bradford et al. (2011) used the same approach for testing the effect of flow trials on fish population metrics. We followed this approach.

Pink salmon presence/absence was coded “on” or “off” in the ANOVAs because there were no available data describing run size in each of the odd years. This coding was not particularly sensitive because an odd year with small or almost non-existent numbers of spawners received the same “on” code as did an odd year having large numbers of spawners. Nutrient concentrations, particularly NH_4-N and SRP that are products of salmon decomposition, were reasonable proxies for spawner counts because anomalous and high concentrations originated in the mainstem river, not in tributaries. They only occurred in odd years and the timing coincided with spawning by Pink salmon. This line of evidence showed that high NH_4-N and SRP concentrations must be associated with relatively large numbers of spawning Pink salmon while low concentrations must show presence of few spawners having little influence on soluble nutrient concentrations. Based on the nutrient proxies, Pink run size was relatively large or at least large enough to cause change in NH_4-N and SRP concentrations in 2003, 2005, 2009, 2011, 2013, and 2021. NH_4-N and SRP concentrations in the other 6 odd years were at or close to those of even years, indicating weak run size that did not change soluble nutrient concentrations in the main spawning reaches 2 and 3. To overcome effects of low spawner numbers in half of the odd years, the Pink effect that was found on algal PB, chironomid density, and benthos family richness had to be very large in the six strong run years to be statistically significant in the ANOVAs. It is possible that Pink effects were present among other endpoints but were not detected because of the rather blunt coding for influence of Pink spawning. Unfortunately, there was no other choice due to lack of data showing Pink spawner counts.

Wetted habitat area increases logarithmically with flow in the Bridge River (Figure 3.5). This relationship is additive to the biotic response to flow for calculation of response over areas of whole reaches. A reported areal measure of biomass or density can be multiplied by change in wetted habitat area associated with flow to derive a reach-wide response. Given that all measurements of periphyton and benthic invertebrates occurred at the same low baseflow in the fall among years, there was little value in considering wetted area adjustments. The consistent fall flows produced similar wetted areas among biotic measurements of all Trials and Reaches.

4.2.2. Periphyton

Periphyton samples for taxonomic analysis in 2021 were consistent with earlier data showing benthic algae in the Lower Bridge River mostly included diatoms with smaller abundance of blue greens, and chlorophytes (Sneep et al. 2021). These taxa are ubiquitous among mountain rivers (Wehr et al. 2014, Bowman et al. 2007, Goma et al 2005, Carpenter and Waite 2000, Hieber et al 2001). The small-celled genus *Achnanthidium* that occurred in all previous Bridge River samples has wide ranging environmental tolerances (Ponader and Potapova 2007). Other Bridge River diatoms including *Tabellaria* sp., and *Diatoma* sp. occur in widely varying nutrient conditions (Bothwell 1989) and *Encyonema, Eunotia, Gomphonema, and Nitzschia* are common in extreme physical conditions of alpine and mountain streams like the Bridge River (Rott et al. 2006). With the blue greens and chlorophytes that are also common in cool and fast streams (Bowman et al 2007), these diatoms are thought to be resilient due to fast recovery after scour events (Peterson 1996), formation of phosphatases to sequester phosphorus at low concentrations and grow optimally at higher nutrient levels (Bothwell 1989), and shift assemblage patterns with temperature (DeNicola 1996). All taxa were potentially usable as food among grazing aquatic insects, particularly the EPT and dipterans that were common in the Bridge River (Junker and Cross 2014, Cummins and Merritt 1996).

A flow-algal biomass relationship can have a threshold wherein variation in flow at some low flow does not modify algal biomass, but once flow exceeds some higher value, biomass declines (Davie and Mitrovic 2014, Flinders and Hart 2009). The shape of this response curve can be related to mixtures of different growth forms in which tightly adhered, adnate, prostrate species such as *Achnanthidium* sp. may be highly resistant to sloughing, unless of course substrata movement occurs, while less adhered taxa such as *Nitzschia* sp. may be more easily sloughed. Some form of step response may have occurred in the Bridge River during the spring and summer periods during the progression of rising flows, particularly during the High Flow Years. Afterwards in the fall of each year when sampling occurred, there was time for any flow related decline in biomass to recover from summer disturbance according to algal growth kinetics. In situ accrual to some maximum biomass or temporary equilibrium can take about two months (Grimm and Fisher 1989, Tornes et al. 2015) with shorter times associated with reduced nutrient limitation (Perrin et al 1987, Bothwell 1989). The several weeks at low base flow in the fall when samplers were installed would have provided sufficient time for algal biomass to recover from summertime disturbance. This process shows resilience by the periphyton community. High flows during the spring and summer of the High Flow Years only exerted a short - term change to autotrophy with no impact on periphyton biomass beyond several weeks following disturbance.

In the absence of flow disturbance, nutrient concentrations can determine the extent to which periphytic algae grow to reach some PB (Bothwell 1989). Neither SRP, DIN, or molar N:P ratios had any relationship with PB but the presence of Pink salmon that released nutrients during carcass decomposition in Reach 2 did favour higher PB. The anomalous NH_4-N and SRP concentrations in that reach was a key indicator of this nutrient release. Highest NH_4-N and SRP concentrations coincided with presence of pink spawners in the odd years while lowest concentrations occurred in even years when pinks were absent. NH_4-N is transient in its typically rapid oxidation to NO_3-N in nutrient deficient surface waters, although this nitrification can be slowed at low temperature (Perrin et al. 1984). NH_4-N is the inorganic form of N that is the main nutrient limiting forest growth (Kimmins 1987, LeBauer and Treseder 2008, Bobbink et al 2010, Mahendrappa et al. 1986), which means it is tightly retained in forest soils and rarely gets above 10 $\mu g \cdot L^{-1}$ in pristine streams in the absence of some other source in temperate northwest watersheds. Furthermore, NH_4-N is a preferred N source by phytoplankton (as in Carpenter Reservoir) and periphyton in the Bridge River because less energy is required to take up and metabolize NH_4-N compared to NO_3-N (Lachmann et al. 2018), thus putting more demand on NH_4-N availability by algae. Given these conditions, the high NH_4-N concentrations of 50 – 180 $\mu g \cdot L^{-1}$ that were found during Trials 1 and 2 in Reaches 2 and 3 in the presence of Pink salmon (Figure 3.12) points to nutrient loading from Pink carcasses and it shows the substantial effect Pink salmon spawning can have on the biogeochemistry of mountain streams. The relatively high NH_4-N and SRP concentrations in many of the odd years benefitted biological production via trophic upsurge while lower nutrient concentrations in even years supported less biological production as shown in the BACI analysis for benthic invertebrates. This ecological benefit from salmon carcass decomposition is well known (e.g. Wipfli et al 1998, Johnston et al 2004, Ruegg et al. 2012, Harding et al 2014). Where there is a tight coupling between nutrient release, uptake by algae, and response in algal growth and biomass, a functional relationship between nutrient concentration and algal biomass may not be apparent, as was found in the Bridge River. This disconnect may occur because any release of nutrients from salmon carcasses, particularly where there is a small run size, may be immediately sequestered to support algal biomass. This process may be one reason for the lack of association between nutrient concentrations and algal PB.

Other algal growth-limiting factors to consider are light (Hill 1996) and temperature (Goldman and Carpenter 1974, Bothwell 1988). Water depth can influence light attenuation that affects rate of photosynthesis in periphytic algae (Hill 1996). During sampler incubations in the fall, all water depths at the periphyton samplers were <0.35 m and they did not vary with Trial. The river substratum was visible from above the water surface and light attenuation from glacial turbidity declined during the fall. With this small range of water depths, there would be little to no difference in light limitation of algal growth between Trials. Water temperature was also consistent within reaches between trials (Figure 3.7) and had no relationship to PB (Figure 3.14), so would not be expected to impart a Trial effect on PB.

Grazing by aquatic insect larvae may have contributed to the increase in PB between Trial 1 and the Post-High Flow years, based on a review by Feminella and Hawkins (1995). For grazers to exert control of PB between Trial 1 and the Post-High Flow years, an inverse association between abundance of benthos and mean PB would be expected, assuming that order level benthos counts are a reasonable surrogate for abundance of grazers. The higher PB during Post-High Flow years compared to lower PB in Trial 1 would be expected to occur with lower benthos abundance in the Post-High Flow years compared to that in Trial 1. That outcome was not found for any of the benthos metrics. Densities were the same, which infers grazing effects on periphyton may not have been temporally different enough to explain the difference in PB between Trial 1 and the Post-High Flow years.

4.2.3. Benthic invertebrates

The Bridge River benthos (Tricoptera, Plecoptera, Ephemeroptera, chironomids, other true flies (Diptera), and a range of rarer taxa) can be ingested by salmonids (Hynes 1970, Scott and Crossman 1973, Wipfli and Baxter 2010, Quinn 2018) and were found at densities similar to or higher than those among other comparable rivers (Deegan et al. (1997), Wipfli et al. (1998), Rosario and Resh (2000), Dewson et al. (2007), Rader and Belish (1999).

Unlike periphyton biomass, the Bridge River benthic invertebrate density and diversity declined during the High Flow Years compared to the other flow periods. Values of benthos metrics were unchanged between Trials 1 and 2, as was predicted to some extent in early planning (Failing et al. 2004), but there was a 69% decline in the density among all benthos taxa between Trial 2 and the High Flow Years. There was no event other than the high spring and summer flows that distinguished the High Flow years from the other blocks of years, which supports a conclusion that the high flows contributed to the decline in benthos density and diversity found in the fall following the spring and summer high flows of the High Flow years. This finding shows that flows during the High Flow Years exceeded capacity of habitat to sustain previous densities of benthos. This Trial (time) effect occurred despite a lag of more than two months between the time of high flows in summer and the time of sampling in the fall, which may be considered a low rate of recovery within the year of the flow event. This rate is lower than the two weeks to a month for recovery following disturbance reported by Figueroa et al. (2006) and Mackay (1992) but it is similar to other evidence that more than a year may be needed for full recovery of community diversity following disturbance (Chapelsky et al. 2020).

The high flows of 2016 – 2018 moved bedload, resulting in coarsening of substrata with net loss of small particles in some places but net accumulation in others (Ellis et al 2018). This particle movement may have contributed to lower benthos density and diversity during the High Flow Years compared to the other time periods. The patchy armouring declined with distance from the dam, generally resulting in larger particles in Reach 4 than in Reaches 3 and 2 based on the particle size measurements in 2019 (Table 3.1). Coarse substrata may not effectively trap fine particulate organic matter (FPOM) that comes from periphyton and allochthonous detritus thereby limiting the availability of food for many benthic invertebrates (Bundschuh and McKie 2016).

The measured differences among particle sizes between reaches in 2019, however, were small. The median size of 11.7 mm in Reach 4 comprised medium gravel while the sizes in Reaches 3 and 2 (7.6 mm and 9.1 mm respectively) were fine gravel (Wentworth scale: Bunte and Abt 2001). All sizes were suitable for colonization by benthic invertebrates (Williams and Mundie 1978) or were at the low end of optimum sizes (Quinn and Hickey 1990). They certainly were not larger than ideal sizes and they were suitable to support spawning by anadromous salmon (Davey 2019). The lack of a Reach effect on total benthos and EPT among Trials is evidence that these particle size differences between reaches were not enough to cause change at least among the EPT between different places in the river. Although chironomid density was lower in Reach 3 compared to the other reaches, substrata particle size was not likely a driving factor because Reach 3 particles remained close to optimum sizes for wide ranging taxa, including chironomids.

Loss of large woody debris during the High Flow years may have reduced availability of invertebrate habitat and food associated with microbial films on and within surface complexities of wood. Amounts of wood in the river were not measured between flow trials but observations of scour and disturbance of riparian vegetation in Reaches 4 and 3 that occurred during High Flow years shows that organic matter was moved to some other place in the river at elevations above most water levels and would not be effective in contributing to benthos habitat (Photo 4.1).

Photo 4.1 A 2020 image of moved organic debris resulting from the High Flow years. C. Perrin photo.

Only one of the three High Flow Years was a Pink-on year (2017). Given the low NH_4-N and SRP concentrations in that year that were not much different from the Pink-off years (Figure 3.12), the size of the Pink run must have been small with little contribution to nutrient loading during algal sampling in the fall. This lack of nutrient load from Pinks may have exacerbated loss of benthos during the High Flow years. While high flows of the High Flow years would have scoured and displaced benthic biota as is typical during flood events (Robinson 2012), recovery of that biota may have been strongly limited by low nutrient availability. SRP concentration was particularly low in Reach 2, which shifted N:P ratios in Reach 2 upwards and into a range showing extreme phosphorus deficiency for algal growth.

Another consideration is lotic invertebrate recruitment mediated by drift from upstream. This drift may be limited by interruption of the flow continuum by the dam and reservoir (Jones 2010, Ellis and Jones 2013), commonly called serial discontinuity (Ward and Stanford 1995). With physical removal of benthos during the 2016 – 2018 high flows, this discontinuity may have caused recruitment to take longer for re-establishment of the benthic invertebrate communities following the High Flow years than might be expected in a river that is connected to headwaters. Indeed, the increasing family richness with distance from the dam among all flow trials suggests Reaches 4 and 3 were much like headwater streams in which invertebrate recruitment from tributary inflows and egg laying would gradually add diversity and community complexity with distance from stream source, in this case the dam. We would expect Reach 2 to be less affected by serial discontinuity due to supplemental drift from the Yalakom River but lack of a Reach effect on total benthos and EPT density, despite the change in richness, suggests that supplemental drift from the Yalakom River was not enough to change density between reaches upstream and downstream of the Yalakom-Bridge confluence.

Unlike the EPT, chironomid density was lower in Reach 3 compared to the other reaches. Reach 3 was within a confined canyon where wetted area relative to reach length increased less with rising flow than in the other reaches, which would result in greater water velocities in Reach 3 compared to the other reaches. The Ephemeroptera (mayflies) are particularly well suited to these conditions because many species have fusiform body shapes that enable them to move about on the tops of rocks in rapid water, while others are dorsoventrally flattened to enable them to hug rock surfaces (Edmonds and Waltz, 1996). The Plecoptera (stoneflies) and Tricoptera (caddisflies) do not have these attributes to the same extent but they occurred in relatively small densities compared to the mayflies (Figure 3.18), making them less influential in the statistical test of Reach effects when combined with mayflies in the EPT group. Although chironomids can exist in almost any aquatic habitat, they are mainly burrowers and may be more susceptible to velocity driven drift with rising flow in a confined channel like Reach 3 than the mayflies. This difference of body shape and ability to remain in place may contribute to the mayfly dominated EPT not being affected by physical differences of the reaches while the chironomids were more sensitive. Water velocity in Reach 3 may be the differentiating factor.

Chironomid density did significantly increase in the presence of Pinks, unlike richness and responses by other taxa. Chironomid larvae are generally collector gatherers and filterers that can benefit directly by feeding on small decomposing particles of fish tissue as well as other detrital matter, some of which would be suspended by wildlife feeding on carcasses. The caddisflies have similar feeding habits but their low numbers relative to other families of the EPT may have caused a Pink effect to be masked by different responses of the other taxa. This feeding habit is different from many of the mayflies and stoneflies. The mayflies were collector gatherers, like chironomids, but also scrapers due to body shapes and feeding appendages that facilitate grazing periphyton. Many of the stoneflies were shredders of leaf litter and facultative detritivores (e.g., Nemouridae, Capniidae) or predators of other insects (e.g., Perlidae, Perlodidae, Chloroperlidae). These functional capacities may be less directly influenced by the presence of Pink carcasses than those of the chironomids or caddisflies.

Spawning by Pink salmon every second year fortuitously provided an opportunity to test benthos response to nutrient addition derived from the spawning salmon. The BACI approach for analysis was well founded, having been used in some of the earliest tests of nutrient enrichment of rivers (e.g. Johnston et al 1990). By selecting even years when spawners were absent and years when spawners were abundant based on nutrient concentration proxies, we had a clear demarcation between no nutrient addition and a nutrient addition treatment that was remarkably similar to that used in published nutrient enrichment trials in other rivers (Harvey et al 1998, Johnston et al 1990, Slavik et al. 2004). We also had a good demarcation between the control, Reach 4, that received few spawners based on anecdotal observations in the field and on measured chemistry and the treatment Reach 2 that supported abundant spawners again based on field observations and chemistry. It was reasonable to accept the nutrient proxies for run size because no other process or tributary stream could explain the anomalous NH_4-N and SRP concentrations in the odd years when Pinks were spawning. Using these proxies, it was evident that Pink run size was sufficient to cause detectable nutrient loading to the river in only half of the Pink-on years over the study duration since beginning in 1996. The proxies show Pink run size was small or absent in the other odd years.

The positive response by benthic invertebrates to nutrient addition shows that autotrophic biological production in Reach 2 of the Lower Bridge River is limited by nutrient concentrations. Based on nutrient ratios realized from the salmon spawning and carcass decomposition, production in that reach is mainly limited by nitrogen with or without enrichment from salmon.

This response means that nutrient addition to the Lower Bridge River with the use of fertilizer or other source can potentially increase biological production. A BACI type of analysis done with existing juvenile salmonid abundance data might help to see if the enrichment effect passes beyond benthic invertebrates to fish as has been found in other river fertilization trials (Johnston et al 1990, Slavik et al. 2004). If it does, intentional fertilization may be a useful tool to offset losses associated with high and damaging flow events like those during the High Flow Years. Such a treatment would be most beneficial in Pink-off years when nutrient deficiency of algal growth is most pronounced. Similarly, greatest benefit would be in Reach 2 where the ecological response to nutrient enrichment is now evident.

4.2.4. Other observations

There is uncertainty about cause of declining SRP concentrations and rising molar N:P over time in water released from Carpenter Reservoir (Figure 3.11). Uptake of soluble phosphorus by phytoplankton and P adsorption onto glacial flour (Hodson et al. 2004) in Carpenter Reservoir may be a sink for bioavailable P (Limnotek 2019). These processes have always been present but may be changing over time. There might be changes in the overall transport of P caused by the variation in what parent materials are being eroded as the Bridge glacier recedes at a remarkable rate of 118 m/year (Chernos 2014). This erosion would influence water chemistry in Downton Reservoir, the Middle Bridge River, Carpenter Reservoir, and ultimately the Lower Bridge River (Chernos 2014, Allen and Smith 2007). Downstream transport of P and other nutrients can be modified by management of flows and reservoir water surface elevations (Limnotek 2019) but larger scale processes driven by glacial melt may have influence (Hood and Scott 2008). Further investigation is required to determine if these or other explanations are plausible.

Reach 1 was sampled only during the Post-High Flow years, resulting in a sample size of three based on the design for analysis using years as replicates. Pink-on years were 2019 and 2021 and the one Pink-off year was 2020. In those three years, soluble nutrient concentrations in Reach 1 were less than those in Reach 2 possibly due to nutrient loading from Pink salmon that may spawn more in Reach 2 than in Reach 1 followed by rapid uptake of nutrients during downstream transport. Tributary chemistry does not explain the differences by reach (Table 3.7). Relatively high nutrient concentrations in 2021 in the presence of spawning Pinks in Reach 2 was particularly noticeable (Figure 3.12). Nutrient concentrations in Reach 1 were less than (SRP) or equal to (DIN) those in the Yalakom River with both being less than in Reach 2 during the Post High Flow Years. Coincidentally, benthos densities were three times greater in Reach 2 than in Reach 1 or the Yalakom River during those years (Figure 3.18), possibly due to more Pinks in Reach 2 than in in Reach 1 or the Yalakom River. While the Yalakom River must be an important source for recruitment of aquatic insects to Reaches 2 and 1, enrichment from presence of Pink salmon in odd years appeared to drive greater density among all taxonomic groups in Reach 2 than in Reach 1 or the Yalakom River. Sampling in additional years using the present layout is needed to gain further insight into temporal patterns in Reach 1, again using contrasts with those in Reach 2 and the Yalakom River. Enumeration of Pink salmon spawners by reach would be particularly valuable for examining links between nutrient pulsing from spawners and availability of food for juvenile salmonids.

4.2.5. Conclusions for MQ #2

Ten conclusions related to interactions between flow, nutrients, and lower trophic levels in the Lower Bridge River can be stated from the BRGMON1 project completed to the end of data collection in Year 10 (2021):

1. The lower Bridge River supports a benthic algal and invertebrate community that is common among mountain rivers. Increasing taxonomic richness with increasing distance from the dam showed the river was much like a headwater stream with taxa added by colonization over distance from source. Aquatic invertebrates in the Lower Bridge River were mostly larval stages of aquatic insects and were potentially available as food organisms for juvenile salmonids.
2. Water releases from the Terzaghi Dam during the High Flow Years when annual peak flows exceeded 100m^3·s^{-1} caused a 69% decline in density of benthic invertebrates in the Lower Bridge River compared to densities found at annual peak flows <60 m^3·s^{-1} during all other flow Trials and during the Post High Flow Years. The decline in density at high flows was attributed to scour and exceedance of habitat to support benthic invertebrates commonly found at lower flows.
3. Finding a flow effect on benthic invertebrates at baseflow, in the fall after peak flows in summer during the High Flow Years showed little or no recovery from flow disturbance within a growing season of flow disturbance.
4. Recovery of the benthic invertebrate community from high peak flows was found within a year after disturbance when peak summer flows in the year of measurement were <60 m^3·s^{-1}.
5. Lack of a high flow effect on periphyton biomass was attributed to time for recovery between the times of high flows in June and July and times of biomass measurement in September through November during the High Flow Years.

6. Presence of spawning Pink salmon produced a trophic upsurge based on five lines of evidence:
    a. High nutrient concentrations corresponded with many Pink-on years and low nutrient concentrations corresponded with all Pink-off years.
    b. Ammonium and SRP concentration were indicators of Pink presence/absence because they are products of carcass decomposition and were only found in relatively high concentrations in some of the Pink-on years. Ammonium was a particularly strong indicator and may be a surrogate for size of a Pink Salmon run. This rationale showed relatively strong runs to Reaches 2 and 3 in 2001, 2003, 2005, 2009, 2011, 2013, and 2021. The ammonium proxy indicated poor run size in the other odd years.
    c. Peak biomass of periphyton accruing on substrata was greatest in the presence of Pink salmon spawners and carcasses, particularly in Reach 2 where spawner numbers may be greatest. This Pink salmon effect on algal biomass was not apparent where numbers of spawners and carcasses may be low (Reach 3 and Reach 4).
    d. Chironomid density was greatest in the presence of Pink salmon spawners and carcasses, particularly outside of the confined canyon of Reach 3.
    e. The BACI analysis showed that nutrient enrichment from Pink salmon spawners and decomposing carcasses caused a doubling of benthic invertebrate densities.
7. Algal biomass increased over years in all reaches with biomass in 2021 approximately double what it was in Trial 1 (2000 – 2010). Much of this change may be related to the Pink salmon effect on trophic upsurge when Pink-on years dominated a Trial (e.g. Pinks present in 2 out of the 3 years of the Post High Flow Years) compared to when Pink-on years were less prevalent in a Trial (e.g. half of the Trial 1 years).
8. Potential phosphorus deficiency for biological production was greater in Reach 2 than in the other reaches. This attribute of Reach 2 may be why biological response (periphyton peak biomass, chironomid density, invertebrate family richness) to presence of Pink salmon was greater in that reach compared to the other reaches.
9. Several lines of evidence showed that ranges of temperature and light among Trials and Reaches did not affect variation in patterns of benthic assemblages.

Fertilization of the lower Bridge River can be considered a useful tool to help offset losses of fish food organisms associated with high flow events or years, pending the outcome of analyses to determine if fish growth and size at age would increase with greater abundance of food organisms produced from nutrient enrichment.

4.3. Management Question 3

How do changes in physical conditions and trophic productivity resulting from flow changes together influence the recruitment of fish populations in Lower Bridge River?

The 2021 fish sampling data added another set of results for the Modified Operations years which started in 2016 but were considered part of the Post-high flow treatment (along with the 2019 and 2020 results) in terms of the analyses since salmonid fry (Age-0+) sampled in 2019-2021 were spawned, incubated, and reared entirely under lower flow conditions (i.e., similar to the Trial 2 hydrograph). Conversely, the earlier Modified Operations years (2016 – 2018) were characterized by high flows that far exceeded the Trial 2 peak and resulted in substantially different habitat conditions for fish within the study area during those years (Sneep et al. 2019) and have altered habitat conditions relative to the previous flow trials (see response to MQ #1).

Mean weight data provided an indication of fish size for each species and age class during the fall stock assessment (in September) for each flow treatment, which can be a reflection of growth. Mean weights were also used to transform the juvenile salmonid abundance data into biomass values for each of the target species and age classes. Mean weights of each species and age class were almost always highest (or among the highest) in each reach during the High Flow years (2016 – 2018) and were next highest during the Post-high flow years (2019-2021) compared to the previous flow treatments. However, it should be noted that there was overlap in the standard deviation error bars among some treatments in certain reaches, suggesting that the statistical significance of these differences may be limited in some cases.

There are a few possible reasons why the mean sizes tended to be highest during the High Flow years: 1) despite reduced abundance of benthic invertebrates (Section 3.3; Sneep et al. 2019), the amount of forage available may still have been ample given the significantly reduced density of juvenile fish during the modified operations years and lower competition for the food resources that were available); 2) water temperatures were warmer during the spring and summer rearing period which may have improved growth conditions; or, 3) the high flows likely selected for the largest individuals, as fish compete for habitat areas that are available and the smallest individuals may more likely be displaced downstream or out of the study area. The high mean weights during the Post-high flow years were most likely due to reasons 1) and 2) above, which have continued to some extent, but also due to recovered benthic invertebrate abundance (see Figure 3.18 in Section 3.3) after three years of a return to lower peak flows.

Given the uncertainty about which of these explanations may have been correct, we also plotted mean condition factor (Fulton's K values) by reach and flow trial for each species. The condition factor data showed a different pattern than the mean weight data: Highest condition factors were generally during trials 0, 1, or 2 (according to species/age class) and the lowest were generally during the Mod. Ops. years (i.e., High flows and Post-high flows). Condition factor is a better metric for assessing the relative fitness of fish among the flow trials because it accounts for the relationship between fish length and weight. These results suggest that improved growth (i.e., reasons 1) and 2) above) is probably less likely the cause of the larger mean size of fish during the Mod. Ops years, and that size selection based on flow magnitude (i.e., reason 3) or other undetermined factors may be the more likely explanation.

Overall, juvenile salmonid abundance and biomass were substantially reduced under the three years of high flows (2016-2018), compared to flow trials 1 and 2 and the pre-flow baseline period, and remained low despite a return to lower flows (similar to the Trial 2 hydrograph) during the Post-high flow period (2019-2021). Total abundance of juvenile salmonids (Chinook, coho and mykiss combined) were highest under the flow trial releases (Trial 1 mean = ~312,000 fish; Trial 2 mean = ~284,000 fish), compared to the Pre-flow baseline, High flow and Post-high flow periods (means = ~189,000, ~69,000 and ~84,000 fish, respectively). Overall, the recruitment of juvenile salmonids was reduced by 75–80% under High flows (from 2016 to 2018) relative to trials 1 and 2, when production was greatest overall in each reach. Salmonid abundance data suggested modest recovery in 2019, but no further recovery in 2020, and slight declines under the higher peak flows (~25 $m^3 \cdot s^{-1}$ instead of 15 $m^3 \cdot s^{-1}$) in 2021. Total juvenile abundance for reaches 2, 3 and 4 was approx. 94,000 fish in 2019, 87,000 fish in 2020, and 70,000 in 2021, which was an increase from the 63,000 – 80,000 fish produced during the three preceding High Flow years (or 15,000 more than the High Flow average). However, the Post-High flow abundances were 83,000 – 273,000 fewer fish than the Trial 2 years (2011 – 2015), or 200,000 fewer fish than the Trial 2 average.

While all species and age classes declined during the Mod. Ops. years, the degree of effect varied among them. Under the high flows, the average production of mykiss fry was 20% relative to the two flow trials. Mykiss parr abundance was 30% of both the Trial 1 and 2 estimates. Chinook fry abundance was 30% of Pre-flow numbers, 60% of Trial 1, and equivalent to Trial 2. It is possible that Chinook fry abundance did not further decrease under the high flows (relative to the Trial 2 mean) since their abundance was already depressed due to early emergence effects on their survival or life history caused by the flow release (see Sneep and Evans 2020). Coho fry abundance was 10% of the Trial 1 and 2 numbers. Coho fry went from being the second most abundant species-age class (next to mykiss fry), to the lowest under the high flows. This could have been due to the coincidence of the onset of high flows in May shortly after their emergence time in March or April (modelled; Figure 3.8) when their capacity to hold or select habitats in the high flows would be very limited. This same factor may also have been an issue for the mykiss fry, which would likely emerge during the high flow period (June – July).

Trends in biomass among flow treatments for all species and age classes generally followed those based on abundance (see Figure 3.27) because changes in average weight across flow treatments have been less significant than the changes in abundance. However, the higher mean weights during the High flow and Post-high flow years had a slight moderating effect on the change in biomass (relative to Trial 2) than the change in abundance. This moderating effect was evident for each species and age class during the High flow and Post-high flow treatments and was most notable for mykiss fry and parr since the increases in mean weight were more substantial for this species.

The increase in juvenile abundance from the High flow years to the Post-high flow years was entirely due to improved recruitment of mykiss fry and coho fry (i.e., by approx. 4,000–20,000 and 3,000–17,000 more fish relative to 2016 – 2018, respectively). Post-high flow abundances of mykiss parr (8,000 fish) and Chinook fry (10,000 fish) were each lower than the estimates for these species-age classes from the High flow years. The 2019 mykiss parr had recruited as Age-0+ fish under the last year of the High flows in 2018, so they were not expected to have recovered one year later. The 2020 and 2021 mykiss parr were from the first two year-classes that recruited under the Post-high flow conditions. Like the other species-age classes, they fared poorer than expected given the return to lower flows but were likely limited by the minimal recovery of mykiss fry. In general, the mykiss fry-to-parr survival rate (~30% on average) has remained quite consistent across the different flow treatments.

Chinook fry production remained relatively low (between 11,000 and 15,000 fish) and stable for 10+ years (i.e., since the end of the Trial 1 period). The 2019 Chinook estimate (~12,000 fish) was on par with this range; however, the 2020 and 2021 estimates were lower (~7,000 and 9,000 fish, respectively). At this point the reason for the drop in Chinook production in 2020 and 2021 is not clear, but it may have been impacted by broodstock collection operations or the increased incidence of straying in 2019 and 2020 (affecting spawning or recruitment for the Bridge River population). Overall, relative to the Trial 2 abundances for mykiss fry (162,000 fish), mykiss parr (33,000 fish) coho fry (76,000 fish) and Chinook fry (13,000 fish) the recovery in the first three years following high flows was quite limited (i.e., Post-high flow abundances were 30%, 20%, 20% and 80% of Trial 2 abundances for these species/age classes, respectively; Table 3.16, In terms of recovery post-high flows (i.e., in 2019 to 2021), all species and age classes have fared relatively poorly to-date. Since the return to lower peak flows (i.e., at or near the Trial 2 levels) following the High flow years, mykiss fry and coho fry abundance increased very modestly (+9K mykiss fry and +11K coho fry); however, the estimates for mykiss parr and Chinook fry actually decreased relative to the High flow period (-2K and -3K on average, respectively) (Table 3.16a and Table 3.17). For Biomass during the Post-high flow treatment, only coho fry increased, the other species and age classes all decreased relative to the High Flow years (Table 3.16b). The abundance and biomass estimates were fairly equivalent between 2019 and 2020 for all species/age classes and substantially lower than the Trial 2 estimates, particularly for mykiss (fry and parr) and coho fry (Figure 3.27). For Chinook, the 2019 and 2020 estimates were lower, but the Trial 2 estimates were already low (compared to the Pre-flow estimates) for this species. In 2021, with moderately higher peak flows than Trial 2, the abundance and biomass for mykiss fry and coho fry were lower than in 2019 and 2020, whereas mykiss parr and Chinook fry were in a similar range as those previous two Post-high flow years.

Table 3.17 and Figure 3.28).

By reach during the Post-high flow years, highest juvenile abundances for mykiss (fry and parr) and coho (fry) were in Reach 3. Chinook abundance was highest in reaches 1 and 2. Among the reaches, the modest increase in abundance of juvenile fish in 2019-2021 was almost entirely attributable to Reach 3. Recovery of juvenile fish production in reaches 4 and 2 was negligible in the first three years following high flows, except for coho which increased by approx. 3,000 fish in Reach 4. Based on the redd survey data provided by BRGMON-3, Reach 4 is where the majority of coho spawning occurs (Figure 3.8; White et al. 2021).

For Year 9 and Year 10 reporting, we ran the juvenile salmonid abundance and biomass results for each flow treatment period through a mixed effects model to formally test for flow effects associated with each of the treatments. In the Year 9 analysis, one model run was completed using data that were not stratified by reach (representing flow effects on the study area as a whole) and a second run was completed using reach-stratified data. The unstratified model combined the effects of rewetting Reach 4 with the effect of added flow in reaches 2 and 3 and the stratified model allowed us to tease these effects among reaches apart. The intent of the two different approaches was to determine which one was better at explaining the variation in log density or biomass by the fixed effects (flow for the unstratified model; flow and reach for the stratified model) and random effects (year). For the stratified model, the benefits of improving the fit of the model by adding more parameters (i.e., reach) must outweigh the costs to precision of adding extra parameters. The model outputs (refer to Figures 3.21 and 3.22 and Tables 3.18 and 3.19 in Sneep et al. 2021) characterized the flow effects on log abundance and biomass and confirmed that the reach-stratified model was the better approach (lower DIC scores; higher FE_r^2 values) for each species and age class, except Chinook fry. For Chinook fry, both models were equivalent since their abundance and biomass have diminished among the reaches during each flow treatment.

Since the mixed effects model results indicated there were strong reach-specific flow treatment effects on juvenile salmonid density and biomass in the Year 9 analyses we did not re-run the unstratified model in Year 10. Rather we incorporated analysis of a set of covariates that were selected based on *a priori* relevance to juvenile fish production. However, covariate effects such as phosphorous concentration, benthic invertebrate density, summer water temperature, and spawner density did not improve the predictive ability of the model. This result does not imply these factors are not important determinants of juvenile density and biomass, because covariate effects were evaluated in a model that already accounted for categorical reach effects and reach-specific flow treatment effects (which had strong predictive power). Thus, covariate effects attempted to explain variation in densities or biomass across reaches and years beyond what could be explained by categorical reach and flow treatment effects. It is also important to recognize that the extent of variation in covariate values was sometimes limited, making it difficult for the model to estimate a covariate effect. For example, Chinook escapement has been fairly consistent and low across the Trial 2, High Flow and Post-High Flow treatments. This results in limited contrast in spawner abundance among years within these treatments. Under these conditions the model cannot separate whether low spawner density or flow conditions were the main causes for low juvenile densities under these treatments.

Furthermore, the *a priori* assumption of relevance may have been strained in some cases where we were limited by the information available. For benthic invertebrate abundances (as a measure of trophic productivity) the time series of data available were collected in the fall. Spring and summer data were available for the Pre-flow period and the first half of Trial 1 but were discontinued after 2005. For all Age-0 groups of juvenile salmonids, those fall trophic measurements (collected in November) do not necessarily align with conditions that would have influenced their fall abundance (i.e., at the time of stock assessment sampling in September). Summer invertebrate densities would probably be a better metric, but that was only available for 10 of the 26 years of monitoring.

For the covariates including annual spawner escapement sizes, the fish counter data were not available for all years (i.e., since the start of Trial 2 only) so the full time series of data available for Chinook and coho were AUC-derived estimates based on visual surveys. The estimates from these surveys assume comparable post-flow release observer efficiency (o.e.) under all flow treatments (since o.e. wasn't assessed prior to Trial 2). The river became chronically more turbid under the release flows (based on anecdotal observations), so this assumption may artificially inflate the estimates during the Pre-flow period (when flows in the study area were much clearer) for these species. For mykiss, the estimates for steelhead escapements were only available since the start of Trial 2, and several years had missing or incomplete estimates due to the effects of peak flows >15 $m^3{\cdot}s^{-1}$ on counter operation or visual assessment methods (i.e., during the High Flow years and 2021). For these reasons we used the time series of estimates for other nearby populations as a surrogate (assuming abundance trends in the LBR were similar; absolute abundance values were not necessary for this analysis) and, as mentioned above, estimates of spawner numbers for resident rainbow trout were not available in any study year.

The adult salmon escapement estimates provided by the BRGMON-3 program were also used to evaluate stock-recruitment relationships according to flow release treatments as another means of evaluating whether spawner stock size was a potential limiting factor on recruitment. An apparent shift in escapement-fry stock-recruitment curves for Chinook and coho across the different flow treatments reflected the changes in fry abundance seen in the juvenile abundance analysis. However, because the curves associated with each treatment were different, and there was uncertainty in estimating egg-to-fry survival rates, there was limited information for defining the initial slope of the curves (which is essential for understanding the number of spawners required to "fully seed" the available habitat).

We fitted new curves for the Post-high flow period (2019-2021; Figure 3.33 and Figure 3.34) that suggested improved recruitment of juveniles per spawner for coho relative to the High Flow treatment and highest recruitment per spawner of Chinook among all treatments. However, like for the High Flow period, these curves were based on only three datapoints and most were clustered near the origin of the x-axis (i.e., relatively low escapements). As such, there was again limited information for defining the initial slope which was why the asymptote of the Post-high flow curve for Chinook was higher than the curves for any other treatment. As such, this is a reflection of the limited number and spread of available datapoints for this flow treatment rather than an indication of improved recruitment.

It remains important to point out that the escapement estimates for Chinook in at least a couple of the Post-high flow years were likely biased low (possibly by ≥50%) due to the effect of fish fence operations on the spawner surveys in those years (White et al. 2021). The lowest juvenile Chinook recruitment values have also coincided with the years of broodstock fence operation. Another factor potentially affecting spawning (and subsequent recruitment in 2020 and 2021) was the incidence of straying by Chinook spawners from other populations due to the migration obstruction on the Fraser River (upstream of the Bridge River confluence) caused by the Big Bar slide. The Chinook broodstock collection fence did not impact coho spawners due to the difference in run timing for these species in the LBR. Also, the Big Bar slide should not have impacted the incidence of straying for coho as much as Chinook since the flows at the slide site were more passable during the coho migration window (DFO unpublished data), although data on coho straying were not available for this report.

Due to the limitations in the number and spread of datapoints available for each flow treatment, the stock-recruitment analysis continues to be a bit of a crude tool for assessing whether spawner stock size was a factor in limiting juvenile recruitment among years and flow treatments in the Lower Bridge River (for the reasons described in Section 3.4.3). Because of these reasons, we included spawner escapement metrics as a covariate in the mixed effects model (as described above and in Section 2.8.5) to explicitly test whether escapement sizes among years and treatments alter the flow effect size and certainty for these species. However, as noted above, the inclusion of escapement metrics (i.e., spawner densities) did not improve the predictive capability of the model. This was likely because, as the stock-recruitment curves show, juvenile abundance does not tend to increase with increasing spawner abundance above relatively low escapement thresholds.

Salmonid abundance data were collected in Reach 1 starting at the beginning of the Post-high flow treatment (i.e., 2019-2021). Results for this period highlighted that all of the target species and age classes were present in the reach, of which mykiss fry were the most abundant (mean = ~22,000 fish), followed by coho fry (mean = ~11,000 fish), Chinook fry (mean = ~8,000 fish), and then mykiss parr (mean = ~4,000 fish). This pattern of relative abundance among the species in Reach 1 was quite consistent each year and the absolute abundances for each species and age class (except coho fry) increased in Reach 1 each year. The Reach 1 total for all species and age classes (mean = ~46,000) during the Post-high flow years was just over 1/3 of all fish captured in the study area (mean = ~128,000). Lineal densities in Reach 1 were on par with Reach 4 for mykiss fry, reaches 4 and 2 for mykiss parr, Reach 2 for coho fry, and reaches 3 and 2 for Chinook fry, at this stage of post-high flow recovery.

While the number of year replicates are still very limited for Reach 1 relative to the other study reaches, the lineal density results for the Post-high flow period may suggest that: a) densities in Reach 1 were less affected by the High flow years; b) recovery could be occurring more quickly in this reach than the upstream reaches; or c) that fish were additionally recruited to this reach from upstream (i.e., reaches 2, 3 and 4) or downstream (i.e., Fraser River) sources. Without the historical time series of data for Reach 1, however, it is not possible to know how current abundances in this reach compare with the earlier flow trials or how relative differences among the reaches may or may not have changed.

Given the expected increase in straying to the LBR by salmon from other populations due to the migration obstruction on the Fraser River caused by the Big Bar slide, samples for DNA analysis were collected from coho and Chinook juveniles in 2020 so they could be analyzed for genetic stock identification (see Sneep et al. 2021 for more information). The samples from juvenile Chinook (n= 117) were sent to the Molecular Genetics Lab at Pacific Biological Station for this analysis. From the original sample size of 117, three individuals failed to genotype and eight were identified as non-Chinook, reducing the viable sample size to 106 (Wetklo and Sutherland 2021). The following conclusion from their analyses comes directly from their summary report:

> *"Two analytical approaches have been applied to describe the ancestry of Bridge River juvenile Chinook sampled in 2020. The majority of the juveniles were the progeny of pure form Bridge parental pairings. We estimate that at least 13% of the juveniles were admixed and attributed to the successful spawning of mid and upper Fraser River Chinook salmon strays in Bridge River."* (Wetklo and Sutherland 2021).

4.4. Management Question 4

What is the appropriate 'shape' of the descending limb of the 6 $m^3 \cdot s^{-1}$ hydrograph, particularly from 15 $m^3 \cdot s^{-1}$ to 3 $m^3 \cdot s^{-1}$?

Results from ramp down and fish salvage monitoring within the WUP (Trial 2) range in 2021 were very similar to the results reported for previous years. As such, they did not provide significant *new* insights on the optimal 'shape' of the descending limb of the hydrograph from 15 $m^3 \cdot s^{-1}$ to 3 $m^3 \cdot s^{-1}$ beyond what has been reported for this flow range previously (Sneep et al. 2021; McHugh and Soverel 2017; Sneep 2016). Ramping across this range in 2021 conformed well to the timing and shape implemented during the previous years of Trial 2 flows (Figure 3.38). However, the results did reaffirm that 13 $m^3 \cdot s^{-1}$ is the approximate flow threshold below which fish stranding risk tends to increase. As such, implementing the WUP rates (≤2.5 cm/hr and maximum 15 cm/day) is likely warranted across most or all of this range. Above the 13 $m^3 \cdot s^{-1}$ threshold, there is flexibility to implement faster ramp rates (up to 6.0 cm/hr was tested in 2021) to reduce flows more quickly without increasing fish stranding risk significantly (based on results for the modified operations years to-date: 2016 – 2021).

Since high flows (defined as flow releases >15 $m^3 \cdot s^{-1}$) were implemented in 2021, responses to the "*Fish Salvage and Stranding Risk Assessment*" portion of the Modified Operations management questions are provided in Section 4.7.2, below. The results for the ramp down assessment and fish salvage efforts for the WUP range in 2021 are also integrated into that portion of the Discussion.

4.5. Management Question 5

Do increased water temperatures and early emergence associated with Terzaghi Dam flow releases affect the survival of juvenile Chinook salmon in the Lower Bridge River?

The following response to this question comes from the Chinook Emergence Timing and Life History Review Memo Report that was prepared for St'at'imc Eco-Resources and BC Hydro (Sneep and Evans 2022). This Management Question was one of the focusses of that document, and, as such, there was additional context and detail provided within it (beyond what is in this report). It was based on the current set of information available from BRGMON-1, as well as other studies, at the time of writing. Please refer to that Memo Report, which serves as a supplement to the BRGMON-1 reporting, for further results and details pertaining to this management question.

> "*Based on the results of monitoring, the answer to this question is a qualified "Yes". Since the onset of the release and its associated effects on incubation conditions and emergence timing, there has been a corresponding decrease in abundance for both juveniles (recruits) and, subsequently, returning spawners in the study area relative to the Pre-flow treatment. Given each of the factors described in the* [Discussion section of the memo report]*, it is unlikely that the observed reduction in abundance was directly caused by instream flow changes (i.e., flow rate, depths, velocities) associated with the flow trials. Coho and steelhead/rainbow trout abundances increased under the Trial 1 and 2 hydrographs, but then declined sharply under the Trial 3 high flows. Conversely, Chinook abundance declined early on and then has remained quite stable and low across each of the flow trials.*
>
> *The main reason the answer is qualified is the juvenile abundance results are based on the numbers of fish that remain in the study area (i.e., "stayers") following emergence. Given the diversity of life history strategies exhibited by Chinook (and documented by the LBR spawner scale samples) it is certainly possible that a proportion of Chinook recruits produced in the LBR migrate soon after emergence and rear outside of the study area (i.e., "movers"). The available data do not provide insight into the magnitude of this proportion, or what the differences in survival rate and contribution to the adult population for stayers vs. movers might be. Recommendations for additional monitoring approaches to fill these information*

> *gaps are provided in Section* ***Error! Reference source not found.*** [of the memo report].”

4.6. Management Question 6

What freshwater rearing habitats are used by Lower Bridge River juvenile Chinook salmon and is rearing habitat use influenced by Terzaghi Dam flow releases?

The following response to this question comes from the Chinook Emergence Timing and Life History Review Memo Report that was prepared for St’at’imc Eco-Resources and BC Hydro (Sneep and Evans 2022). This Management Question was one of the focusses of that document, and, as such, there was additional context and detail provided within it (beyond what is in this report). It was based on the current set of information available from BRGMON-1, as well as other studies, at the time of writing. Please refer to that Memo Report, which serves as a supplement to the BRGMON-1 reporting, for further results and details pertaining to this management question.

> “*At the site level, there was no strong evidence for a change in the habitat criteria (i.e., depths and velocities) selected for rearing by juvenile Chinook at the time of the annual stock assessment in September; however, the range of depths and velocities included in the sample is also constrained by the limitations of the sampling method. At the reach level, there has been a change in juvenile Chinook distribution. Prior to the flow release, the majority of rearing occurred at sites in Reach 3. Under the flow trials, the contribution of Reach 3 diminished and Reach 2 increased. This shift may reflect a downstream migration tendency for Chinook juveniles which are still primarily produced in Reach 3 (based on BRGMON-3 streamwalk and redd survey data). Abundances in Reach 4 have always been low. Considering that conditions (e.g., food availability) may be poor for fish that emerge early under the flow release, they may move downstream in search of better conditions. Additional monitoring would be required (as described in Section* ***Error! Reference source not found.*** [of the memo report]*) to determine the extent to which this is occurring and whether there is a survival consequence for early emergers.*
>
> *Between mainstem habitats and the off-channel habitats that have been sampled, Chinook have been much more abundant in the mainstem of the LBR. Chinook abundance has been very low in two constructed off-channel rearing habitats (i.e., “Bluenose” in Reach 4 and “Applesprings” in Reach 1) that were sampled in September 2018* [to 2021] *(see* [Section 3.5.2 of this Year 10 report for more information on these results]*). However, Chinook yearlings were periodically sampled out of a series of off-channel beaver ponds in the Horseshoe Bend (Reach 2; Rkm 23.6) in early spring, suggesting the possibility of overwintering use of off-channel habitats, as has been observed on the Nicola River (Swales and Levings 1989). Given the lack of*

> *quantitative sampling in this case, however, this observation remains somewhat anecdotal.*
>
> *So, based on the results of monitoring, the answer to this question is a qualified "Yes"; rearing habitat use (in the form of juvenile Chinook distribution at the time of the fall stock assessment) has been influenced by the flow release. The reason this answer must be qualified is because there is also the possibility that a proportion of LBR Chinook are rearing outside of the BRGMON-1 study area and therefore have not been detected by the monitoring to-date. If that is the case, we do not know what proportion of the total juvenile production in the LBR is "missing" from the fall recruitment estimate, or if that proportion has changed among the flow treatments. Such a change would be further evidence of a shift in habitat use (e.g., between the LBR to the Fraser River or other tributaries); however, data to support this are not presently available. While rearing outside of the study area may be an important survival adaptation for the LBR Chinook population (if successful), it should be acknowledged that an original goal for the flow release was to maintain or increase productive capacity for juvenile salmonids within the LBR study area itself.*"

4.7. Modified Operations Management Questions

4.7.1. High Flow Monitoring

Do flow releases from Terzaghi Dam under the modified flow regime affect water quality or cause erosion in the Lower Bridge River? If so, what are the potential effects on fish and what mitigation options are available?

The high flows in 2021 resulted in elevated water temperatures, turbidity and %TGP levels (relative to background or values recorded during previous Trial 2 hydrograph years) in reaches 3 and 4, and caused some erosion along the wetted edge at various sites that were monitored in reaches 2 – 4.

Air and water temperatures generally increased in the LBR across the 2021 monitoring dates (as per typical seasonal trends) and water temperatures increased from upstream to downstream sites on each survey date. As noted in the results, there was a large discrepancy between the spot water temperatures recorded on the survey dates and the logger data that was recorded hourly throughout the year. The information referred to here is based on the logger data since there is a more continuous record and for consistency with temperature records from previous years. As noted in response to the management question in Section 4.1, the upper range of water temperatures was warmer (by 1-2°C) than daily average temperatures for this period during trials 1 and 2. However, these temperatures were also approx. 1-2°C cooler than during the previous High Flow years (2016-2018) when flow release magnitudes were much higher. Temperatures were within the optimal range for rearing by chinook, coho and steelhead, as cited in the literature, in all cases (McCullough et al. 2001; Oliver & Fidler 2001; Myrick & Cech 2000). These warmer spring-summer temperatures during the modified operations years may have been one of the factors contributing to the larger mean size of the juveniles from target species by the time of the annual stock assessment sampling in September.

Based on spot monitoring during the peak flow period in 2021, turbidities were not particularly elevated at any of the sites on the survey dates (i.e., spot turbidity ranged between 10 and 21 NTU in the mainstem LBR, and between 2 and 8 NTU in the Yalakom River) compared to the previous high flow years when spot turbidity ranged up to ~56 NTU and ~40 NTU at peak flows in 2017 and 2018, respectively (turbidity was not monitored in 2016). However, the turbidity data recorded by the loggers deployed at the upstream and downstream extents of each reach and in the Yalakom River indicated that there were actually three turbidity peaks in the LBR during 2021: The first peak was in May and June (up to ~80 NTU at the top of Reach 4), the second was in August (up to ~60 NTU), and the third was in September (also up to ~60 NTU). In all cases, the peak declined with distance downstream.

During high flow release periods, elevated turbidity levels can be caused by: 1) the generally turbid water drawn from the bottom of Carpenter Reservoir; 2) flooding of edge areas above the Trial 2 peak that are less frequently wetted; and 3) seasonally higher turbidity from tributaries that are in freshet condition. All three of these factors were likely at play in 2021; however, the timing of peaks at all sites above the Yalakom confluence were similar to that in water released from Carpenter (measured at the Plunge Pool), which showed that in-reservoir processes were a significant factor in reaches 4 and 3 during 2021. The logger data also indicated that there were some high turbidity events in the Yalakom in 2021, including a very high spike during July (i.e., up to ~300 NTU) likely related to run-off caused by record high air temperatures that month. Tributary inflows were an additional factor influencing turbidity levels, particularly in reaches 2 and 1 (below the Yalakom inflow). The results collected in 2021 also highlight that the data recorded by loggers at regular intervals (i.e., ≤ hourly) throughout the year tend to be much more informative than periodic spot measurements.

Total dissolved gas (TDG) % saturation levels were relatively stable across the high flow period in 2021 with a gradient of diminishing value with distance downstream of the dam (i.e., 105-106% at the Plunge Pool (Rkm 40.9); 101-103% at the mid-point of Reach 3 (Rkm 30.4); and 100-102% at the bottom of Reach 3 (Rkm 25.5). TDG levels dropped to 100-102% at all sites by the end of August and were consistently 100-101% on all survey dates in the Yalakom. To-date, the mechanism causing increased TDG at high flows from the low-level outlet at Terzaghi Dam has not been determined, but the observed saturation levels have been below the thresholds for triggering a mitigation response according to Table 1 of BC Hydro's Total Dissolved Gas Management Strategy (i.e., BC Hydro 2014).

According to the strategy, the effects of elevated TDG on fish result from an interaction of several physical factors (e.g., maximum and sustained versus background TDG saturation levels; depth and distance of effect downstream of the source; duration; temperature, accessible depths available to fish in the river, etc.) coupled with potential compensatory mechanisms available to fish (e.g., physiological, behavioural, species- and life history-specific characteristics, etc.). Assessment of the possible interaction of each of these factors for determining potential detrimental effects on fish were not feasible with the scope of information available from the 2016-2018 or 2021 high flow monitoring and were not considered necessary given the levels of supersaturation reached. Going forward, it is recommended that monitoring follow the BC Hydro Total Dissolved Gas Management Strategy: Implementation Plan when the measured TDG values reach or exceed the 110% to 115% range (BC Hydro 2014).

Within the 110-115% TDG range, if fish may be limited to shallow water (≤0.5 m) habitats in some areas, and the duration of exposure will exceed 10 days, the management strategy recommends "Low Effort Monitoring." Low effort monitoring is described as shore-based visual surveys with a focus on locating/identifying dead or floating fish in shallow water habitats, which should be carried out during the period of threshold exceedance or soon after. The visual assessments should also be accompanied by total gas pressure measurements taken at various locations (including spot measurements in the hyporheic zone if incubating eggs may be present). This level of effort is very similar to the kokanee entrainment surveys and water quality measurements (see High Flow Monitoring methods and results) that have been implemented in each high flow year to-date. Any prescription beyond that would require the input of BC Hydro, St'at'imc, and the regulatory agencies.

Bank erosion and substrate recruitment was documented to occur at 12 of 18 monitored sites in reaches 2, 3 and 4 during 2021, covering a total area of ~464 m^2 (est.). Many of these were the same locations where erosion and substrate recruitment was observed in 2018; however, the total erosion area was substantially higher that year (~5,400 m^2 est.) when peak flow magnitude went up to 102 $m^3 \cdot s^{-1}$. In 2021, there were two medium-sized areas (<500 m^2) in Reach 2 ("Below Horseshoe" and "Horseshoe Bend"), and the remainder were considered small (<50 m^2; *n*= 10) sites in reaches 3 and 4. The other six assessed sites did not show evidence of bank erosion under the 2021 high flows.

The majority of the observed erosion was caused by the interaction of the expanded wetted edge of the river (at high flows) with the base of active alluvial slide areas adjacent to the river, and most of it occurred at or below the high flow waterline. As such, these sites contain natural materials and are areas of ongoing substrate recruitment to the river, even at low flows, which is an important source given the general armouring of substrates in channels below impoundments. The rate of recruitment and distribution of substrate materials in the river would be correlated with flow magnitude (Ellis et al. 2018). On an ongoing basis, substrates from these slides will continue to fill in the material mobilized at the toe of the bank. Based on the scope of monitoring in 2021, crews were not able to assess substrate deposition within the river channel.

The specific conditions contributing to the incidence of kokanee entrainment from Carpenter Reservoir into the Lower Bridge River remain somewhat elusive. Flow release magnitude is certainly a potential driver (i.e., higher flows drawn from the reservoir may increase the risk of entrainment). Totals of 83 and 48 injured or deceased kokanee were observed during the high flow period (up to 97 and 127 $m^3 \cdot s^{-1}$) in 2016 and 2017, respectively; however, 0 were observed in 2018 (with peak flows up to 102 $m^3 \cdot s^{-1}$) and 2021 (peak flows up to 25 $m^3 \cdot s^{-1}$). Carpenter reservoir operations have also varied among those years (i.e., different minimum and maximum levels, and fill and drawdown rates) and kokanee population size may also have declined since the start of modified operations (Annika Putt, BRGMON-4 Project Biologist, pers. comm.). The method employed for documenting entrainment was also limited: when kokanee *are* observed it serves as an indication that some degree of entrainment has occurred. However, it is not effective at documenting the actual rates of entrainment or confirming when entrainment has not occurred.

Beyond the indication of some entrainment from the kokanee surveys, we don’t have any data to suggest a direct adverse effect of the measured temperatures, turbidity levels, %TDG saturation or bank erosion across the high flow period on fish in the Lower Bridge River – especially relative to the effects of release temperatures during the incubation period and the effects of the high flows on rearing habitat availability, and juvenile salmonid abundance and biomass, which we are more certain about.

4.7.2. Fish Salvage and Stranding Risk Assessment

How does the risk of fish stranding during LBR ramp downs vary with discharge?

According to the fish salvage results for ramp downs from high flows (>15 $m^3 \cdot s^{-1}$) in 2016-2018, the fish stranding risk was consistently low (or occasionally moderate), per 1 $m^3 \cdot s^{-1}$ increment of flow change, above a threshold of ~13 $m^3 \cdot s^{-1}$ based on the criteria defined in the fish stranding protocol (Sneep 2016). Conversely, below the 13 $m^3 \cdot s^{-1}$ threshold, the fish stranding risk was more consistently moderate or high. The 2021 results reaffirmed these observations. As such, implementing the WUP rates (≤2.5 cm/hr) is likely warranted across most or all of this range. Above the 13 $m^3 \cdot s^{-1}$ threshold, there is flexibility to implement faster ramp rates (up to 4.1 cm/hr was tested in 2017, 4.0 cm/hr in 2018, 4.6 cm/hr in 2019, and 5.0 cm/hr in 2021; Table 3.24) to reduce flows more quickly without increasing fish stranding risk significantly (based on results for 2016–2021). Reducing flows more quickly (especially from high discharges >15 $m^3 \cdot s^{-1}$), can provide the opportunity to reach more optimal levels for summer rearing (i.e., the Trial 2 peak or lower) in less time, or over fewer days. Furthermore, field crews have reported that, because ramp down events can be completed more quickly, final gate changes can be implemented earlier in the day such that the river stabilizes at the new stage level before the end of the day, which facilitates the effectiveness of salvage efforts.

An important caveat that must be noted for the Modified Operations results, however, is that juvenile salmonid numbers were shown to be substantially reduced by the effects of the high flows overall (i.e., due to poor survival or displacement out of the study area) from 2016–2018, and only moderate recovery in the Post-high flow years (2019–2021), relative to the densities documented under trials 1 and 2. Although, given the effects of the high flows on physical habitat parameters, benthos production, and fish abundance (as noted in the sections above), this may be the case any time flow magnitudes in the range of the 2016 – 2018 discharges occur. For these reasons, the incidence of fish stranding and the effects of faster ramp rates on stranding risk should continue to be monitored for flows >15 $m^3 \cdot s^{-1}$ in order to build up a larger sample size of data and improve confidence in the results.

How does the risk of fish stranding during LBR ramp downs vary by reach?

Under flow ramp downs <15 $m^3 \cdot s^{-1}$ in previous years, differences in the number of fish salvaged among reaches were substantial: On average, the number of stranded fish in Reach 4 (mean = ~3,000) was nearly 1.5-fold higher than the number in Reach 3 (mean = ~2,000), and the amount of identified stranding area was nearly equivalent among them (4,865 and 4,540 m^2, respectively; Sneep 2016) despite the fact that Reach 3 is nearly four times longer than Reach 4 (~12 km vs ~3 km, respectively). Patterns in stranding risk among reaches were also apparent in the Post-high flow data, although they were different than the Trial 2 results, probably due to the significant physical changes to habitat that were caused by the high flows and the minimal recovery of juvenile salmonid abundances, particularly in Reach 4. From 2019-2021, the amount of fish stranding area was substantially greater in Reach 3 (mean = 5,878 m^2) versus Reach 4 (1,350 m^2), and the numbers of fish salvaged were also higher in Reach 3 (mean = 1,852 vs 475); However, the fish stranding densities were slightly higher in Reach 4 (33 fish per 100 m^2 of strand area), relative to Reach 3 (30 fish per 100 m^2). Note: under the high flow range (>15 $m^3 \cdot s^{-1}$) in 2021, among all four reaches, fish were only salvaged at one location (i.e., near the fish fence site; Rkm 26.1) in Reach 3.

Despite differences in sample size (i.e., # of years) for ramping and fish salvage data between modified operations years and the Trial 1 and 2 flows, there is little uncertainty that juvenile fish distribution and relative stranding risk varies among the reaches of the Lower Bridge River. Given the low abundance of juvenile salmonids in 2016–2021 overall, it would be worthwhile to characterize the relative stranding risk among the reaches at different high flow magnitudes (when fish abundance may be greater). However, based on assessment of stage changes in Reach 2 within the Trial 2 range (see Figure 3.4 and Table 3.21 in the Year 9 report (Sneep et al. 2021) since stage level data beyond the Rkm 36.8 compliance location were not available for 2021), total daily stage changes per event in that reach were approx. ⅓ to ½ the magnitude of changes at the top of Reach 3, and hourly changes were likely lower as well due to the mitigating influence of the Yalakom River and other tributary inflows. This is one of the primary reasons that fish salvage efforts were focussed on reaches 4 and 3 and not on reaches 2 and 1 during implementation of the Trial 1 and 2 hydrographs, as well as flow ramp downs within that range in recent years. Considered together: the reduced stage changes, moderated ramping rate due to attenuated inflows, and generally low fish stranding risk documented for reaches 2 and 1 to-date, mutually support that fish stranding risk below the Yalakom confluence is lower than it is in the reaches above. However, it should be noted that fish stranding data in reaches 1 and 2 were available for ramp downs from flows >15 $m^3 \cdot s^{-1}$ but not within the 15 to 1.5 $m^3 \cdot s^{-1}$ range.

How does the risk of fish stranding during LBR ramp downs vary with ramping rate and stage change?

Ramping rates implemented in 2021 were between 2.0 and 6.0 cm/hr (mean stage reduction per hour at the Plunge Pool – recorded by a staff gauge monitor from CNBS). This represented the fourth modified operations year that rates above the ≤2.5 cm/hr WUP-referenced rate were implemented (rates in 2017, 2018 and 2019 were up to 4.1, 4.0 and 4.6 cm/hr, respectively). As before, fish salvage crews were on the ground to monitor the results, but generally avoided proactively moving fish out of strand-risky habitats in advance of isolation or dewatering (i.e., "incidental" catches) such that catch data would better reflect actual numbers of stranded fish[1]. Despite the variation in ramping rates across the high flow range, the incidence of fish stranding did not change substantively relative to the identified risk for flows ≤15 $m^3 \cdot s^{-1}$. The observed stranding risk remained low (<10 per 1 $m^3 \cdot s^{-1}$) to moderate (10 to 99 fish per 1 $m^3 \cdot s^{-1}$), as defined in the fish stranding protocol (Sneep 2016), across each of the implemented rates at high flows.

Some of the higher ramp rates in 2021 (i.e., up to 6.0 cm/hr) were also implemented within the WUP range. While the resulting fish stranding risk didn't exceed the highest stranding rates for previous ramp downs within the 15 to 1.5 $m^3 \cdot s^{-1}$ flow range, there was some evidence of a positive correlation between ramp rate and strand risk, particularly between 4.0 and 6.0 cm/hr (see Figure 3.42). Fish stranding risk increased from approx. 100 to 500 fish per 1 $m^3 \cdot s^{-1}$ across this ramp rate range under WUP flows.

Since the onset of modified operations, the sample size for stranding monitoring at ramping rates >2.5 cm/hr has grown. However, as was noted for the MQ above, juvenile salmonid abundances in 2016-2021 were low overall, which could have confounded the incidence of stranding despite the higher rates implemented in several of those years. Nonetheless, the results to-date suggest that stranding risk is lower at flow releases >13 $m^3 \cdot s^{-1}$ (see above). As such, this should provide opportunity to continue testing higher rates across the high flow range going forward without unduly risking higher fish mortality. Increasing the number of ramp down events completed at higher ramp rates will be necessary to continue reducing uncertainty about the specific effects of higher ramp rates across the different high flow levels (and potentially at different fish abundance levels in the future). The results from 2021 also suggest that caution should be exercised when considering or implementing faster ramp rates within the WUP range.

---

[1] In 2018, this approach was applied except when water temperatures in isolating habitats became too warm. In these cases, fish were removed as 'incidentals' to mitigate the risk of increased mortality.

How does the risk of fish stranding during LBR ramp downs vary by river bank?

The distribution of sites between river left and river right was not equal for flows within the Trial 2 range. Based on the updated salvage survey data (including 2021 results), the distribution of sites was 76% on river left and 24% on river right for ramp downs at flows ≤15 $m^3 \cdot s^{-1}$. Across the high flow range, based on site reconnaissance surveys conducted in 2016-2018 and 2021, the distribution was 45% and 55%, respectively. Upon initial purview, differences in distribution of sites according to side of the river may seem unexpected, given that there is no known reason based in an understanding of channel morphological processes that more strand-risky habitats would naturally form on one side of the river versus the other across the length of these reaches. Rather, it's possible the reason could have more to do with human-caused effects than natural ones.

Other than at the very bottom of Reach 2 (i.e., at Camoo; Rkm 20.0) and the bottom of Reach 1 up to the Applesprings off-channel habitat, road access along the entire length of the Lower Bridge River is along the river left side. The proportion of identified stranding sites on river left is likely influenced by this access and its associated human-caused effects, including: dam construction-, habitat enhancement- (i.e., spawning platforms, off-channel habitats), fish research-, river access-, and gold mining-related activities (to name a few).

At least some of the stranding sites that were likely created or altered by these activities include: the plunge pool, Eagle lake, Bluenose, Russell Springs, fish counter, Hippy pool, Horseshoe bend, and Camoo sites on river left; and the plunge pool, grizzly bar, and Camoo sites on river right. Given that the river was generally in a low flow, pre-release condition for 40 years following dam construction, most of these human-affected sites tend to occur within the lower flow range (≤15 $m^3 \cdot s^{-1}$). At higher flows (>15 $m^3 \cdot s^{-1}$), the distribution of sites appears to become more balanced on either side of the river – closer to what we would expect in the absence of human-caused interference.

Are there opportunities to minimize or mitigate the risk of fish stranding during ramp downs in the Lower Bridge River?

The primary opportunity (or most conservative approach) for minimizing or mitigating the risk of fish stranding is by implementing the ramping rates referenced in the WUP (i.e., ≤2.5 cm/hr) and having fish salvage crews actively salvaging fish in each of the reaches downstream of the dam. This approach has been employed successfully in the Lower Bridge River for documenting the relative incidence of stranding and mitigating mortalities since the continuous flow release began. At these ramp rates, fish may have more opportunity to move out of strand-risky habitats with the changing flow level (similar to what occurs in unregulated systems), relative to faster rates, and fish salvage crews can more easily keep on top of salvaging fish from habitats as they become isolated (and before they dewater). Although, it must be acknowledged that fish stranding does occur on unregulated systems also, and it will never be possible to completely mitigate stranding with ramping rates alone. While being the most conservative from a fish stranding perspective, this approach is also the most time- and labour-intensive as the duration and number of ramp events are higher.

In some cases, such as during the modified operations period when flow releases above the WUP (Trial 2) range were more frequently implemented, there can be additional rationale for ramping the flows down faster in order to reach more optimal summer rearing flows (i.e., ≤15 $m^3{\cdot}s^{-1}$) more quickly following peak flows. With the data for ramp downs from high flows available from 2016-2018 and 2021, there is some evidence for when faster ramping rates can be applied without unduly increasing fish stranding risk. As described in the management question responses above, this could apply to ramping rates up to 5.0 cm/hr at discharges >13 $m^3{\cdot}s^{-1}$ based on the information currently available. However, due to the factors noted in the sections above (low fish abundance during the Modified Operations years to-date; lower sample size at higher ramping rates), the application of these rates should be accompanied by ramp monitoring and fish salvaging (as was done in 2016–2021) to further characterize the fish stranding risk at flows >15 $m^3{\cdot}s^{-1}$ when fish abundances may be greater and expand the dataset from which conclusions are drawn.

4.7.3. Mitigation Effectiveness Monitoring

How does juvenile salmonid habitat availability in the Lower Bridge River change with discharge under the modified flow regime?

Based on the results of the pilot-level sampling in 2018 (additional data collection to address this management question has not occurred since then), there is insufficient information at this point to answer this management question (i.e., from the field data alone). Refer to the Year 7 (2018) report for more information (Sneep et al. 2019). Depths and velocities were highly variable across the different mainstem sites (both those pre-selected as high quality habitat and low quality habitat), and tended to be more consistent in the off-channel sites which are more sheltered from the main channel flows. The high quality mainstem sites tended to span a slightly broader range of both depth and velocity across the high flows than the low quality sites, suggesting they were somewhat more directly affected by changes in discharge across the high flow period. There was some evidence that changes in the availability of suitable habitat areas and conditions (i.e., depths & velocities) at some locations were offset by the changes at other sites, such that there was no substantial net loss among the pre-selected sites across the high flow range surveyed in 2018. However, this is highly tenuous given the degree of variability in the data and the limited sample size (i.e., number of sites and flows tested).

There were some sites (both high quality and low quality) that provided suitable depths and velocities for rearing (based on LBR HSI curves for coho, chinook and mykiss) across the full range of high flows. This is probably because, as the flows increase and the wetted area of the river expands, the habitat with suitable rearing criteria just moves with the wetted edge to some extent. In other words, the edge depths and velocities do not change to the same degree that mid-channel depths and velocities do. A better approach to addressing this question may be to use BC Hydro's Telemac2D model for predicting rearing habitat area across the high flow range and using the field measurements to check or calibrate the predictions. However, this was outside of the current scope of work for BRGMON-1.

How does habitat use by juvenile salmonids change with discharge under the modified flow regime?

Spring (mid June) and summer (mid July) sampling in the off-channel habitats (Bluenose in Reach 4 and Applesprings in Reach 1) in 2020 and 2021 further documented use of these sites by each of the target species during these two monthly periods. Results from 2020 were intended to serve as a "baseline" for lower peak flows (i.e., the Trial 2 peak – 15 $m^3 \cdot s^{-1}$) against which results collected during a high(er) flow release year (i.e., 2021 - 25 $m^3 \cdot s^{-1}$) could be compared for addressing this modified operations management question. A potential shift in habitat use (from mainstem to off-channel) would be determined by detecting an increase in the lineal density of fish in the off-channel habitats that coincides with the timing of the flow increase from Terzaghi Dam and is different than the patterns in lineal densities documented under baseline conditions.

Coho fry were the most prevalent species at the Applesprings site and in the Bluenose outflow channel in 2020 and 2021. Their relative abundance (on average) was equivalent across those two years in the Bluenose habitat and higher in 2020 than 2021 at Applesprings. Chinook fry densities were much lower than coho at the Applesprings site and were absent at Bluenose during all surveys. There was a drop in their relative abundance from June to July, which was most apparent in the 2020 results. This change between spring and summer periods may have corresponded with the timing of outmigration for at least some portion of Chinook juveniles in the Bridge River (see the responses to MQ's #5 and 6 in Sections 4.5 and 4.6 for some additional context on this point). Mykiss (which were predominantly parr during the spring and summer surveys since they occurred during the timing of emergence for the new year class of fry) were more prevalent in the Bluenose site and their relative abundance was slightly higher in 2021 than 2020, although not by a lot. There was a small increase in abundance between June and July at some sites in both years, which reflected supplementation by some recently emerged mykiss fry during this seasonal period.

Under both the Trial 2 peak flows in 2020, and the higher peak flows in 2021, there was no evidence of migration into these off-channel habitats in response to main channel flows (i.e., for refuge) for any of the target species and age classes (based on the level of information these sampling results can provide). However, analysis of these data, as well as what we can learn from them, is limited due to the factors highlighted in the first paragraph of Section 3.5.2. Even if differences or changes in lineal densities occur between years, or sessions within a year, there are a wide range of factors to be considered for discerning the potential causes (including life history events such as emergence or outmigration, or factors which affect capture efficiencies, to name a few) which are not possible based on the current methods.

Depletion sampling at riffle and pool sites in the two off-channel habitats was also completed in fall 2021, as it was each year from 2018-2020. Results from these surveys have indicated that mykiss (fry and parr) dominate the Bluenose habitat and coho fry dominate the Applesprings habitat in the fall. Chinook fry were absent at Bluenose and present only at the riffle sites in Applesprings at relatively low density. In general, densities of juvenile salmonids in these off-channel habitats were high (i.e., on par with mainstem densities in reaches 4 and 2 during the Trial 2 years), particularly in 2018 (the last high flow year), 2020 and 2021, and much higher than current densities in reaches 4 and 1 at this stage of post-high flow recovery in the mainstem.

These depletion sampling results have indicated that: a) the two selected offchannel habitats can support high densities of juvenile salmonids, b) each habitat has consistently had higher suitability for some species over others, and c) the overall densities have varied among years even with the same flows in the mainstem (i.e., 2019 and 2020). Also, given that densities in these habitats were very similar between 2018 (a high peak flow year) and 2020 (a lower peak flow year), there is currently no evidence to suggest that fish move into these off-channel habitats in greater numbers during high flow years (e.g., seeking refuge); however, the number of year-replicates for assessing this is still quite low (*n*= 1 high flow year (2018), 2 low flow years (2019, 2020), and 1 moderate flow year (2021)).

Beyond a potential role as refuge habitats, the overall contributions from these off-channel habitats cannot compensate for the reduced production observed in the mainstem during years with high peak flows (e.g., 2016-2018) due to their limited size and availability in the LBR study area. Total mainstem production under Trial 2 was 284,000 fish (not including Reach 1 which was not sampled during that trial), so the mean estimate of 7,900 fish for the two existing off-channel sites during the Post-high flow years would represent ~3% of this Trial 2 abundance. To make-up for reduced mainstem production documented since the High flow years, a total of 200,000 additional fish are required (i.e., 284,000 Trial 2 average - 84,000 Post-high flow average). Assuming that the *maximum* densities of juvenile salmonids documented in the two off-channel areas from 2018 to 2021 are representative of what these habitats can produce, a total area of 119,000 m^2 (or approximately 103 Bluenose-sized sites) and 130,000 m^2 (equivalent to 15 Applesprings sites) would be required to recover the lost mykiss and coho production in the mainstem. Off-channel recovery of lost Chinook production in the mainstem seems unlikely given the consistently low use of the existing off-channel habitats documented for this species to-date.

5. Recommendations

The following recommendations stem from the analysis and reporting of results for addressing the management questions up to, and including, Year 10 (2021):

1. Measurement of Pink salmon run size is needed to examine the importance of interactions between flow and trophic upsurge on fish food organisms and overall health of the Lower Bridge River.
2. A workshop is needed to review direct and empirical evidence supporting potential benefits of fertilization of the Lower Bridge River among all interest groups. The premise would be to use fertilization as a tool to offset losses of biological production caused by high flows like those occurring during the High Flow Years of 2016-2018. The outcome of this workshop would lead to a decision on whether to implement a fertilization trial including measurement of ecological and fish responses.

6. List of References

Albers, S.J. and E.L. Petticrew. 2012. Ecosystem response to a salmon disturbance regime: Implications for downstream nutrient fluxes in aquatic systems. Limnology and Oceanography. 57: 113-123.

Allen, S.M. and D.J. Smith. 2007. Late Holocene glacial activity of Bridge Glacier, British Columbia Coast Mountains. Canadian Journal of Earth Science. 44: 1753-1773.

Anderson, R.O. and R.M. Neumann. 1996. Length, weight, and associated structural indices. Pages 447–482 in B.R. Murphy and D.W. Willis, eds. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, MD.

APHA (American Public Health Association). 2011. Standard methods for the examination of water and wastewater. Available at: http://www.standardmethods.org American Public Health Association, American Water Works Association, and Water Environment Federation.

Ardon, M., L.H. Zeglin, R.M. Utz, S.D. Cooper, W.K. Dodds, R.J. Bixby, A.S. Burgett, J.F. Shah, N.A. Griffiths, T. K. Harms, S.L. Johnson, J.B. Jones, J.S. Kominoski, W.H. McDowell, A.D. Rosemond, M.T. Trentman, D.Van Horn, and A. Ward. 2020. Experimental nitrogen and phosphorus enrichment stimulates multiple trophic levels of algal and detrital-based food webs: a global meta-analysis from streams and rivers. Biological Reviews. doi: 10.1111/brv.12673.

Arscott, D.B., J.K. Jackson, & E.B. Kratzer. 2006. Role of rarity and taxonomic resolution in a regional and spatial analysis of stream macroinvertebrates. *Journal of the North American Benthological Society*, *25*(4), 977–997.

Bailey, R.C., G. Scrimgeour, D. Cote, D. Kehler, S. Linke, and Y. Cao. 2012. Bioassessment of stream ecosystems enduring a decade of simulated degradation: lessons for the real world. Can. Journal of Fish. Aquat. Sci. 69: 784-796.

Bailey, R.C., R.H. Norris, and T.B. Reynoldson. 2004. *Bioassessment of Freshwater Ecosystems: Using the Reference Condition Approach*. Kluwer Academic Publishers, Massachusetts, USA.

Bailey, R.C., R.H. Norris, & T.B. Reynoldson. 2001. Taxonomic resolution of benthic macroinvertebrate communities in bioassessments. *Journal of the North American Benthological Society*, *20*(2), 280–286.

Barbour, M.T. and J. Gerritsen. 1996. Subsampling of benthic samples: a defense of the fixed count method. *Journal of the North American Benthological Society*, 15 (3): 386-391.

BC Hydro. 2020. Scope of Services – 2020-2021 BRGMON-1 – Lower Bridge River Aquatic Monitoring and Reporting; WUP AND High Flow Monitoring. Prepared by Matt Casselman and Ira Hofer, BC Hydro Fish and Aquatic Issues Team, June 2020.

BC Hydro. 2018. BRGMON-1 Lower Bridge River Aquatic Monitoring – Monitoring Program Terms of Reference Revision 1. Bridge-Seton Water Use Plan. November 30, 2018.

BC Hydro. 2012. Bridge-Seton Water Use Plan Monitoring Program Terms of Reference – BRGMON-1 Lower Bridge River Aquatic Monitoring. Prepared for the Comptroller of Water Rights, Province of British Columbia, January 23, 2012.

BC Hydro. 2011. Bridge River power development water use plan: revised for acceptance for the Comptroller of Water Rights. pp. 1–79.

Biggs, B.J.F. 2000. Eutrophication of streams and rivers: dissoved nutrient-chlorophyll relationships for benthic algae. Journal of the North American Benthological Society. 19: 17-31.

Bobbink, R. K. Hicks, J. Galloway, T. Spranger, R. Alkemade, M. Ashmore, M. Bustamante, S. Cinderby, E. Davison, F. Dentener, B. Emmett, J.W. Erisman, M. Fenn, F. Gillian, A. Nordin, L. Pardo, and W. Devries. 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. Ecological Applications. 20: 30-59.

Borchardt, M.A. 1996. Nutrients. In: R.J. Stevenson, M.L. Bothwell, and R.L. Lowe. (Ed). Algal Ecology. Academic Press. New York.

Bothwell, M.L. 1989. Phosphorus-limited growth dynamics of lotic periphyton diatom communities: areal biomass and cellular growth rate responses. *Canadian Journal of Fisheries and Aquatic Sciences*, 46:1293-1301.

Boulton, A.J. 1999. An overview of river health assessment: philosophies, practice, problems and prognosis. Freshwater Biology. 41: 469-479.

Bowman, M.F., P.A. Chambers, and D.W. Schindler. 2007. Constraints on benthic algal response to nutrient addition in oligotrophic mountain rivers. River Research and Applications. 23: 858-876.

Bradford, M.J. 1995. Comparative review of Pacific salmon survival rates. *Can. J. Fish. Aquat. Sci. 52:* 1327–1338.

Bradford, M.J., J. Korman, and P.S. Higgins. 2005. Using confidence intervals to estimate the response of salmon populations (*Oncorhynchus spp.*) to experimental habitat alterations. *Can. J. Fish. Aquat. Sci.* 62: 2716-2726.

Bradford, M.J., & P.S. Higgins. 2001. Habitat-, season-, and size-specific variation in diel activity patterns of juvenile chinook salmon (*Oncorhynchus tshawytscha*) and steelhead trout (*Oncorhynchus mykiss*). *Can. J. Fish. Aquat. Sci. 58*(2), 365–374. http://doi.org/10.1139/cjfas-58-2-365

Bradford, M.J., P.S. Higgins, J. Korman, and J. Sneep. 2011. Test of environmental flow release in a British Columbia river: does more water mean more fish? *Freshwater Biology* 56:2119-2134.

Bundschuh, M. and B.G. McKie. 2016. An ecological and ecotoxicological perspective on fine particulate organic matter in streams. Freshwater Biology. 61: 2063-2074.

Bunte, K. and S.R. Abt. 2001. Sampling surface and subsurface particle-size distributions in wadable gravel and cobble bed streams for analyses in sediment transport, hydraulics, and streambed monitoring. US Dept of Agriculture, Forest Service, Rocky Mountain Research Station General technical report RMRS-GTR-74.

Carpenter, K.D. and I.R. Waite. 2000. Relations of habitat-specific algal assemblages to land use and water chemistry in the Willamette Basin, Oregon. Environmental Monitoring and Assessment 64: 247-257.

Chapelski, A.J., M.M. Guzzo, L.E. Hrenchuk, and P.J. Blanchfield. 2020. Invertebrate colonization of a newly constructed diversion channel in the Canadian Shield. Canadian Journal of Fisheries and Aquatic Science. 77: 1477-1486.

Chernos, M. 2014. The relative importance of calving and surface ablation at a lacustrine terminating glacier. MSc. Thesis. University of British Columbia.

Chessman, B., S. Williams, & C. Besley. 2007. Bioassessment of streams with macroinvertebrates: effect of sampled habitat and taxonomic resolution. *Journal of the North American Benthological Society*, *26*(3), 546–565.

Costanza, R. and M. Mageau. 1999. What is a healthy ecosystem? Aquatic Ecology. 33: 105-115.

Crossin, G. T., S.G. Hinch, A.P. Farrell, M.P. Whelly, & M.C. Healey. 2003. Pink salmon (*Oncorhynchus gorbuscha*) migratory energetics: response to migratory difficulty and comparisons with sockeye salmon (*Oncorhynchus nerka*). *Canadian Journal of Zoology*, *81*(12), 1986–1995. http://doi.org/10.1139/z03-193

Cummins, K.W. and R.W. Merritt. 1996. Ecology and Distribution of Aquatic Insects. In: R.W. Merritt and K.W. Cummins (ed). An Introduction to the Aquatic Insects of North America. 3rd Edition. Kendall/Hunt Publishing.

Davey, C. 2019. Lower Bridge River Pilot Spawning Gravel Supplementation – Assessment of Spawning Habitat Availability in the Lower Bridge River. Technical Memorandum prepared by Kerr Wood Leidal Ltd. For BC Hydro and St'at'imc Government Services, July 9 2019. 22 p.

Davie, A.W. and S.M. Mitrovic. 2014. Benthic algal biomass and assemblage changes following environmental flow releases and unregulated tributary flows downstream of a major storage. Marne and Freshwater Research. 65: 1059-1071.

Decker, A.S., M.J. Bradford, & P.S. Higgins. 2008. Rate of biotic colonization following flow restoration below a diversion dam in the Bridge River, British Columbia. *River Research and Applications*, *24*(6), 876–883. http://doi.org/10.1002/rra.1076

Deegan, L.A., B.J. Peterson, H.Golden, C.C. McIvor and M.C. Miller. 1997. Effects of fish density and river fertilization on algal standing stocks, invertebrate communities, and fish production in an arctic river. Can. J. Fish. Aquat. Sci. 54: 269-283.

DeNicola, D.M. 1996. Periphyton responses to temperature at different ecological levels. In. R.J. Stevenson, M.L. Bothwell, and R.L. Lowe (Ed). Algal Ecology. Academic Press.

Dewson, Z.S., A.B.W. James, and R.G. Death. 2007. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. *Journal of North American Benthological Society* 26(3): 401-415.

Earl, S.R., H.M. Valett, and J.R. Webster. 2006. Nitrogen saturation in stream ecosystems. Ecology. 87: 3140-3151.

Eaton, B.C., R.D. Moore, and L.G. MacKenzie. 2019. Percentile-based grain size distribution analysis tools (GSDtools) – estimating confidence limits and hypothesis tests for comparing two samples. Earth Surface Dynamics 7: 789-806.

Edmunds, G.F. and R.D. Waltz. 1996. Ephemeroptera. In: R.D. Merritt and K.W. Cummins (Ed). An Introduction to the Aquatic Insects of North America. 3rd Edition. Kendall Hunt Publishing. Dubuque. Iowa.

Ellis, E., C. Davey, A. Taleghani, B. Whitehouse, and B. Eaton. 2018. Lower Bridge River Sediment and Erosion Monitoring, 2017. Prepared by Kerr Wood Leidal Associates Ltd. for St'át'imc Eco-Resources and BC Hydro. 67p.

Ellis, L.E. and N.E. Jones. 2013. Longitudinal trends in regulated rivers: a review and synthesis within the context of the serial discontinuity concept. Environmental Reviews. 21: 136-148.

Ensign, S.H. and M.W. Doyle. 2006. Nutrient spiraling in streams and river networks. Journal of Geophysical Research. 111: 1-13.

Failing, L., G. Horn, and P. Higgins. 2004. Using expert judgment and stakeholder values to evaluate adaptive management options. *Ecology and Society* **9**:13.

Failing, L., R. Gregory, and P. Higgins. 2012. Science, uncertainty and values in ecological restoration: a case study in structured decision-making and adaptive management. *Restoration Ecology* 21:422-430.

Feminella, J.W. and C.P. Hawkins. 1995. Interactions between stream herbivores and periphyton: A quantitative analysis of past experiments. Journal of the North American Benthological Society. 14: 465-509.

Figueroa, R., V. Ruiz, X. Niell, E. Araya, and A. Palma. 2006. Invertebrate colonization patterns in a Mediterranean Chilean stream. *Hydrobiologia*. 571: 409-417.

Flinders, C.A. and D.D. Hart. 2009. Effects of pulsed flows on nuisance periphyton growths in rivers: A mesocosm study. River Research and Applications. 25: 1320-1330.

Francoeur S.N. and B.J.F. Biggs. 2006. Short-term effects of elevated velocity and sediment abrasion on benthic algal communities. Hydrobiologia. 561: 59-69.

Gelman, A. and I. Pardoe. 2006. Bayesian measures of explained variance and pooling in multilevel (hierarchical) models. Technometrics 48: 241-251.

Gelman, A., Carlin, J.B., Stern, H.S., and Rubin, D.B. 2004. Bayesian data analysis, 2nd Edition. Chapman & Hall/CRC, Boca Raton, FL. 668 p.

Goldman, J.C. and E. J. Carpenter. 1974. A kinetic approach to the effect of temperature on algal growth. *Limnology and Oceanography* 19(5): 756 – 766.

Goma, J., F. Rimet, J. Cambra, L. Hoffmann, and L. Ector. 2005. Diatom communities and water quality assessment in mountain rivers of the upper Segre basin (La Cerdanya, Oriental Pyrenees). Hydrobiologia. 551: 209-225.

Grant, S.C.H., M. Townsend, B. White, and M. Lapointe. 2014. Fraser River pink salmon (Oncyhynchus gorbuscha) data review: Inputs for biological status and escapement goals. Report prepared for Pacific Salmon Commission by Fisheries and Oceans Canada and Pacific Salmon Commission. 90pp.

Grimm, N.B. and S.G. Fisher. 1989. Stability of periphyton and macroinvertebrates to disturbance by flash floods in a desert stream. Journal of the North American Benthological Society. 8: 293-307.

Groves, P., J.A. Chandler, & T. Richter. 2008. Comparison of Temperature Data Collected from Artificial Chinook Salmon Redds and Surface Water in the Snake River. North American Journal of Fisheries Management. 28. 766-780. 10.1577/M07-045.1.

Guildford, S.J. and R.E. Hecky. 2000. Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: Is there a common relationship? *Limnol. Oceanogr.* 45: 1213-1223.

Hall, A.A., S.B. Rood, & P.S. Higgins. 2011. Resizing a river: a downscaled, seasonal flow regime promotes riparian restoration. *Restoration Ecology*, *19*(3), 351–359. http://doi.org/10.1111/j.1526-100X.2009.00581.x

Harding, J.N. and J.D. Reynolds. 2014. Opposing forces: Evaluating multiple ecological roles of Pacific salmon in coastal stream ecosystems. Ecosphere. 5(12):157. http://dx.doi.org/10.1890/ES14-00207.1.

Harding, J.N., J.M.S. Harding, and J.D. Reynolds. 2014. Movers and shakers: nutrient subsidies and benthic disturbance predict biofilm biomass and stable isotope signatures in coastal streams. Freshwater Biology. doi:10.1111/fwb.12351.

Harvey, C.J., B.J. Peterson, W. B. Bowden, A.E. Hershey, M.C. Miller, L.A. Deegan, and J.C. Finlay. 1998. Biological responses to fertilization of Oksrukuyik Creek, a tundra stream. Journal of the North American Benthological Society. 17: 190-209.

Healey, F.P. 1985. Interacting effects of light and nutrient limitation on the growth rate of Synechococcus linearis (Cyanophyceae). *J. Phycol.* 21:134-146.

Heggenes, J., K. Alfredsen, J.E. Brittain, A. Adeva Bustos, A. Huusko, and M. Stickler. 2017. Temperature changes and biological responses to hydropower-regulated northern stream systems. University College of Southeast Norway. Publication series No. 21.

Hieber, M., C.T. Robinson, S.R. Rushforth, and U. Uehlinger. 2001. Algal communities associated with different alpine stream types. Arctic, Antarctic, and Alpine Research 33: 447-456.

Higgins, P.S. and J. Korman. 2000. Abundance, growth, standing stock, and components of variation of juvenile salmonids in the Bridge River: An analysis to define "baseline conditions" and optimal sampling designs. Report prepared by BC Hydro and Ecometric Research for Bridge River Water Use Planning Fisheries Technical Committee. 45p.

Higgins, P.S., & M.J. Bradford. 1996. Evaluation of a large-scale fish salvage to reduce the impacts of controlled flow reduction in a regulated river. *North American Journal of Fisheries Management*, *16*(3), 666–673. http://doi.org/10.1577/1548-8675(1996)016<0666:EOALSF>2.3.CO;2

Hill, W.R. 1996. Effects of light. In. R.J. Stevenson, M.L. Bothwell, and R.L. Lowe (Ed). Algal Ecology. Academic Press.

Hodson, A. J., P. N. Mumford, and D. Lister. 2004. Suspended sediment and phosphorus in proglacial rivers: bioavailability and potential impacts upon the P status of ice-marginal receiving waters. Hydrol. Process. **18**: 2409‑2422, doi:10.1002/hyp.1471

Holling, C. S. 1978. Adaptive environmental assessment and management. John Wiley and Sons, New York.

Holm-Hansen, O., C.J. Lorenzen, R.W. Holmes, and J.D.H. Strickland. 1965. Fluorometric determination of chlorophyll. *J. Cons. Perm. Int. Explor. Mer*. 30:3-15.

Holt, C.R., D. Pfitzer, C. Scalley, B.A. Caldwell, P. I. Capece, and D.P. Batzer. 2015. Longitudinal variation in macroinvertebrate assemblages below a large-scale hydroelectric dam. Hydrobiologia. 755: 13-26.

Hood, E. and D. Scott. 2008 Riverine organic matter and nutrients in southeast Alaska affected by glacial coverage. Nature Geoscience Letters. 1: 583-587.

Hynes, H.B.N. 1970. The ecology of running waters. University of Toronto Press, Toronto, Ontario.

Johnston, N.T., E.A. MacIsaac, P.J. Tschaplinski, and K.J. Hall. 2004. Effects of the abundance of spawning sockeye salmon (Oncorhynchus nerka) on nutrients and algal biomass in forested streams. Canadian Journal of Fisheries and Aquatic Science. 61: 384-403.

Johnston, N.T., C.J. Perrin, P.A. Slaney, and B.R. Ward. 1990. Increased juvenile salmonid growth by whole-river fertilization. Canadian Journal of Fisheries and Aquatic Science. 47: 862-872.

Jones, N.E. 2010. Incorporating lakes within the river discontinuum: longitudinal changes in ecological characteristics in stream – lake networks. Canadian Journal of Fisheries and Aquatic Sciences. 67: 1350-1362.

Jones, O.R. and J. Wang. 2010. COLONY: a program for parentage and sibship inference from multilocus genotype data. Molecular Ecology Resources 10: 551-555.

Junker, J.R. and W.F. Cross. 2014. Seasonality in the trophic basis of a temperate stream invertebrate assemblage: Importance of temperature and food quality. Limnology and Oceanography. 59: 507-518.

Kennedy, B.P., K.H. Nislow, and C.L. Folt. 2008. Habitat-mediated foraging limitations drive survival bottlenecks for juvenile salmon. Ecology. 89: 2529-2541.

Kennedy, T.A., J.D. Muehlbauer, C.B. Yackulic, D.A. Lytle, S.w. Miller, K.L. Dibble, E.W. Kortenhoeven, A.N. Metcalfe, and C.V. Baxter. 2016. Flow management for hydropower extirpates aquatic insects, undermining river food webs. BioScience. 66: 561-575.

Kimmins, J.P. 1987. Forest Ecology. Macmillan Publishing. New York.

King, N. and A. Clarke. 2015. The Life-History of Rainbow Trout in the Bridge River as Determined by Otolith Microchemistry. Unpublished report prepared for Coldstream Ecology, Ltd. 15pp + App.

King, R.S. and C.J. Richardson. 2001. Evaluating subsampling approaches and macroinvertebrate taxonomic resolution for wetland bioassessment. *Journal of the North American Benthological Society*, 21(1):150-171.

Korman, J. and A. Tompkins. 2014. Estimating regional distributions of freshwater stock productivity, carrying capacity, and sustainable harvest rates for Coho salmon using a hierarchical Bayesian modelling approach. CSAS Res. Doc. 2014/089.

Lachmann, S.C., T. Mettler – Altmann, and A. Wacker. 2019. Nitrate or ammonium: Influences of nitrogen source on the physiology of a green alga. Ecology and Evolution. 9: 1070-1082.

LeBauer, D.S. and K.K. Treseder. 2008. Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. Ecology. 89: 371-379.

Limnotek. 2019. Carpenter Reservoir Productivity Model Validation and Refinement (BRGMON10): Final report. Report prepared for BC Hydro. 93p. plus appendices.

Lowe, R.L. and Y. Pan. 1996. Benthic algal communities as biological indicators. Pp. 705-733 In: Algal Ecology. R.J. Stevenson, M.L. Bothwell, R.L Lowe (Ed). Academic Press.

Mackay, R.J. 1992. Colonization by lotic macroinvertebrates: A review of processes and patterns. *Can. J. Fish. Aquat. Sci*. 49: 617-628.

Mahendrappa, M.K., N.W. Foster, G.F. Weetman, and H.H. Krause. 1986. Nutrient cycling and availability in forest soils. Canadian Journal of Soil Science. 66: 547-572.

Matthew, P.L. and R.W.J. Stewart. 1985. Summary of juvenile salmonid downstream trapping surveys; Spring, 1984. Report for Community Economic Development, Salmonid Enhancement Program, by Central Interior Tribal Fisheries, Kamloops, B.C., 69 p.

McCullough, D., Spalding, S., Sturdevant, D., & Hicks, M. 2001. Summary of Technical Literature Examining the Physiological Effects of Temperature on Salmonids. US Environmental Protection Agency, (May), 1–119.

McHugh, A.E., and N.O. Soverel. 2017. BRGMON-1 Lower Bridge River Aquatic Monitoring. Annual Report – 2015. Unpublished report by Coldstream Ecology, Ltd., Lillooet, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 45 pp. + Apps.

McHugh, A.E., and N.O. Soverel. 2015. Lower Bridge River Aquatic Monitoring. Year 2014 Data Report. Bridge Seton Water Use Plan. Prepared for St’át’imc Eco Resources, Ltd. and BC Hydro for submission to the Deputy Comptroller of Water Rights, August 2015.

McHugh, A.E., and N.O. Soverel. 2014. Lower Bridge River Aquatic Monitoring. Year 2013 Data Report. Bridge Seton Water Use Plan. Prepared for St'at'imc Eco Resources, Ltd. and BC Hydro for submission to the Deputy Comptroller of Water Rights, August 2014.

McHugh, A.E., and N.O. Soverel. 2013. Lower Bridge River Aquatic Monitoring. Year 2012 Data Report. Bridge Seton Water Use Plan. Prepared for St'at'imc Eco Resources, Ltd. and BC Hydro for submission to the Deputy Comptroller of Water Rights, August 2013.

McHugh, A.E., Soverel, N.O., and D. O’Farrell. 2017. Lower Bridge River (LBR) Aquatic Monitoring (2016 High Flow). Unpublished report by Coldstream Ecology, Ltd., Lillooet, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 48 pp. + Apps.

McKnight, D.M., R.L. Runkel, C.M. Tate, J.H. Duff, and D.L. Moorhead. 2004. Inorganic N and P dynamics of Antarctic glacial meltwater streams as controlled by hyporheic exchange and benthic autotrophic communities. Journal of the North American Benthological Society. 23: 171-188.

Merritt, R.W. and K.W. Cummins. 1996. An Introduction to Aquatic Insects of North America. 3rd ed. Kendall/Hunt Pub. Dubuque, Iowa.

Merritt, R.W., K.W. Cummins, & M.B. Berg. 2008. An introduction to the aquatic insects of North America. (K.W. Cummins & M.B. Berg, eds.) (4 ed.). Dubuque, Iowa: Kendall/Hunt.

Murray, C.B., T.D. Beacham, & J.D. McPhail. 1990. Influence of parental stock and incubation temperature on the early development of coho salmon (Oncorhynchus kisutch) in British Columbia. *Canadian Journal of Zoology* 68 347-358.

Myrick, C. A., & Cech, J. J. 2000. Temperature influences on California rainbow trout physiological performance. *Fish Physiology and Biochemistry*, 22(3), 245–254. https://doi.org/10.1023/A:1007805322097

Nelson, C.E., D.M. Bennett, and B.J. Cardinale. 2013. Consistency and sensitivity of stream periphyton community structural and functional responses to nutrient enrichment. Ecological Applications. 23: 159-173.

Nichols, S.J., T.B. Reynoldson, and E.T. Harrison. 2014. Evaluating AUSRIVAS predictive model performance for detecting simulated eutrophication effects on invertebrate assemblages. Freshwater Science. 33: 1212-1224.

Norris, R.H. and M.C. Thoms. 1999. What is river health? Freshwater Biology 41: 197-209.

Northcote, T.G., & D.Y. Atagi. 1997. Pacific salmon abundance trends in the Fraser River watershed compared with other British Columbia Systems. In D. J. Stouder, P. A. Bisson, & R. J. Naiman (eds.), *Pacific Salmon and their ecosystems status and future options* (pp. 199–218). Seattle.

Nusch, E.A. 1980. A comparison of different methods for chlorophyll and pheopigment analysis. *Arch. Hydrobiol. Beih. Ergebn. Limnol*. 14: 14-36.

O'Donnell B. 1988. Indian and Non-native use of the Bridge River: a historical perspective. Fisheries and Oceans Canada Native Affairs Division Publication No. 7.

O'Farrell, D. and A.E. McHugh. 2018. 2018 Lower Bridge River High Flow Monitoring. Unpublished report by Coldstream Ecology Ltd, Lillooet, B.C. for St'at'imc Eco-Resources Ltd., Lillooet, BC and BC Hydro, Burnaby, BC. 18pp + Apps.

Oliver, G.G. and L.E. Fidler. 2001. Towards a Water Quality Guideline for Temperature in the Province of British Columbia: 3.0 Effects of Temperature on Aquatic Biota. *Ministry of Environment, Lands and Parks, Water Management Branch*.

Perrin, C.J., M.L. Bothwell and P.A. Slaney. 1987. Experimental enrichment of a coastal stream in British Columbia: effects of organic and inorganic additions on autotrophic periphyton production. Can. J. Fish. Aquat. Sci. 44: 1247-1256.

Perrin, C.J. and J. S. Richardson. 1997. N and P limitation of benthos abundance in the Nechako River, British Columbia. Canadian Journal of Fisheries and Aquatic Science. 54: 2574-2583.

Perrin, C.J., K.S. Shortreed, and J.G. Stockner. 1984. An integration of forest and lake fertilization: transport and transformations of fertilizer elements. Canadian Journal of Fisheries and Aquatic Sciences. 41(2):253-262.

Peterson, C.G. 1996. Response of benthic algal communities to natural physical disturbance. In. R.J. Stevenson, M.L. Bothwell, and R.L. Lowe (Ed). Algal Ecology. Academic Press.

Ponader, K.C. and M.G. Potapova. 2007 Diatoms from the genus Achnanthidium in flowing waters of the Appalachian Mountains (North America): Ecology, distribution and taxonomic notes. Limnologica 37: 227-241.

Pritchard, J.K., M. Stephens and P Donnelly. 2000. Inference of Population Structure Using Multilocus Genotype Data. Genetics 155: 945–959.

Quinn, J.M. and C.W. Hickey. 1990. Characterization and classification of benthic invertebrate communities in 88 New Zealand rivers in relation to environmental factors. New Zealand Journal of Marine and Freshwater Research. 24: 387-409.

Quinn, T.P. 2018. The Behaviour and Ecology of Pacific Salmon and Trout. UBC Press.

Rader, R.B., and Belish, T.A. 1999. Influence of mild to severe flow alterations on invertebrates in three mountain streams. Regulated Rivers: Research and Management. 15: 353-363.

Recchia, A. 2010. R-squared measures for two-level hierarchical linear models using SAS. J. Stat. Soft. 32.

Reynoldson, T.B., D.M. Rosenberg, & V.H. Resh. 2001. Comparison of models predicting invertebrate assemblages for biomonitoring in the Fraser River catchment, British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences*, *58*(7), 1395–1410. http://doi.org/10.1139/cjfas-58-7-1395

Rhee, G.-Y. 1978. Effects of N:P atomic ratios and nitrate limitation on algal growth, cell composition, and nitrate uptake. *Limnol. Oceanogr.* 23:10-25.

Rhee, G.-Y. and I.J. Gotham. 1980. Optimum N:P ratios and coexistence of planktonic algae. *J. Phycol.* 16:486-489.

Rinke, K., C.T. Robinson, and U. Uehlinger. 2001. A note on abiotic factors that constrain periphyton growth in alpine glacier streams. Internat. Rev. Hydrobiol. 86: 361-366.

Robinson, C.T. 2012. Long term changes in community assembly, resistance, and resilience following experimental floods. Ecological Applications. 22: 1949-1961.

Rosario, R., and V. Resh. 2000. Invertebrates in intermittent and perennial streams: is the hyporheic zone a refuge from drying? Journal of the North American Benthological Society. 19: 680-696.

Rott, E., M. Cantonati, L. Fureder, and P. Pfister. 2006. Benthic algae in high altitude streams of the Alps – a neglected component of the aquatic biota. Hydrobiologia. 562: 195-216.

Ruegg, J. D.T. Chaloner, P.S. Levi, J.L. Yank, S.D. Tiegs, and G.A. Lamberti. 2012. Environmental variability and the ecological effects of spawning Pacific salmon on stream biofilm. Freshwater Biology 57: 129-142.

Scrine, J., M. Jochum, J.S.Olafsson, and E.J. O'Gorman. 2017. Interactive effects of temperature and habitat complexity on freshwater communities. *Ecology and Evolution.* 7:9333–9346.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Fisheries Research Board of Canada. Bulletin 184.

Smokorowski, K.E. and R.G. Randall. 2017. Cautions on using the Before – After – Control – Impact design in environmental effects monitoring programs. Facets. 2: 212-232.

Smucker, N.J., M. Becker, N.E. Detenbeck, and A.C. Morrison. 2013. Using algal metrics and biomass to evaluate multiple ways of defining concentration-based nutrient criteria for streams and ecological relevance. Ecological Indicators. 32: 51-61.

Sneep, J., C. Perrin, S. Bennett, and J. Korman. 2021. BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 9 (2020) Results. Report prepared for St'at'imc Eco-Resources and BC Hydro. April 2021.

Sneep, J., and M. Evans. 2022. BRGMON-1 Lower Bridge River Aquatic Monitoring: Chinook Salmon Emergence Timing and Life History Review. Report prepared for St'at'imc Eco-Resources and BC Hydro. February 2022.

Sneep, J., C. Perrin, S. Bennett, and J. Korman. 2020. BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 8 (2019) Results. Report prepared for St'at'imc Eco-Resources and BC Hydro. April 2020.

Sneep, J., C. Perrin, S. Bennett, and J. Korman. 2019. BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 7 (2018) Results. Report prepared for St'at'imc Eco-Resources and BC Hydro. July 2019.

Sneep, J., J. Korman, C. Perrin, S. Bennett, and J. Harding. 2018. BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 6 (2017) Results. Report prepared for St'at'imc Eco-Resources and BC Hydro. November 2018.

Sneep, J. 2016. Fish Stranding Protocol for BC Hydro's Bridge-Seton Generation Area. Prepared for BC Hydro, March 2016.

Sneep, J. and S. Hall. 2012a. Lower Bridge River Aquatic Monitoring, Year 2011 Data Report. Prepared for BC Hydro and the Deputy Comptroller of Water Rights. August 2012.

Sneep, J. and S. Hall. 2012b. Lower Bridge River 2011 Flow Ramping Report. Prepared for BC Hydro, Bridge River Generation. March 2012.

Soverel, N.O., and A.E. McHugh. 2016. Lower Bridge River Aquatic Monitoring. Year 2015 Data Report. Bridge Seton Water Use Plan. Prepared for St'át'imc Eco-Resources, Ltd. and BC Hydro for submission to the Deputy Comptroller of Water Rights, August 2016.

Spiegelhalter, D.J., Best, N.G., Carlin, B.P., and A. Van Der Linde. 2002. Bayesian measures of model complexity and fit. J. R. Statist. Soc. B. 64 (4): 583-639

Stevenson, R.J. 1996. An introduction to algal ecology in freshwater habitats. In: R.J. Stevenson, M.L. Bothwell, and R.L. Lowe. Editors: Algal Ecology. 3-30. Academic Press. San Diego.

Stewart-Oaten, A., W.W. Murdoch, and K.R. Parker. 1986. Environmental impact assessment: "Pseudoreplication" in time? Ecology 67: 929-940.

Stumm, W. and J.J. Morgan. 1996. Aquatic Chemistry: An introduction emphasizing chemical equilibria in natural waters. John Wiley and Sons. New York.

Thorp, J.H. and A.P. Covich. 2001. Ecology and classification of North American freshwater invertebrates. 2nd Ed. Academic Press. USA.

Tisdale Environmental Consulting. 2011a. Lower Bridge River Flow Monitoring, 2010. Report prepared for BC Hydro and Power Authority, May 2011.

Tisdale Environmental Consulting. 2011b. Lower Bridge River Flow Monitoring, 2009. Report prepared for BC Hydro and Power Authority, March 2011.

Tornes, E., V. Acuna, C. N. Dahm, and S. Sabater. 2015 Flood disturbance effects on benthic diatom assemblage structure in a semiarid river network. Journal of Phycology. 51: 133-143.

Townsend, S.A., E.A. Garcia, and M.M. Douglas. 2012. The response of benthic algal biomass to nutrient addition over a range of current speeds in an oligotrophic river. Freshwater Science. 31: 1233-1243.

Vinson, M.R. and C.P. Hawkins. 1996. Effects of subsampling area and subsampling procedure on comparisons of taxa richness among streams. JNABS 15(3): 392-399.

Walsh, C.J. 1997. A multivariate method for determining optimal subsampling size in the analysis of macroinvertebrate samples. Marine and Freshwater Research 48(3): 241-248.

Walters, C. J. 1986. Adaptive management of renewable resources. Blackburn Press, Caldwell, New Jersey.

Ward, J.V., and Stanford, J.A. 1995. The serial discontinuity concept: extendingthe model to floodplain rivers. Regulated Rivers: Research and Management, **10**: 159–168. doi:10.1002/rrr.3450100211.

Wehr, J.D., R.G. Sheath, and J. P. Kociolek. Eds. 2014. Freshwater algae of North America: Ecology and Classification. 2nd Edition. Elsevier Press Inc. New York.

Wellnitz, T. and N.L. Poff. 2006. Herbivory, current velocity and algal regrowth: how does periphyton grow when the grazers have gone? Freshwater Biology. 51: 2114-2123.

Wetklo, M. and B. Sutherland. 2021. Genetic stock identification of Bridge River 2020 juvenile Chinook – summary memo. Prepared for St'at'imc Eco-Resources and BC Hydro by the Molecular Genetics Lab at Pacific Biological Station, 22 March 2021.

White, C., A. Putt, D. Ramos-Espinoza, M. Chung and K. Cooke. 2021. Lower Bridge River Adult Salmon and Steelhead Enumeration, 2020. Report prepared for St'at'imc Eco-Resources and BC Hydro. 89 pp + appendices.

Williams, D.D. and J.H. Mundie. 1978. Substrate size selection by stream invertebrates and the influence of sand. Limnology and Oceanography 23: 1030-1033.

Wipfli, M.S. and C.V. Baxter. 2010. Linking ecosystems, food webs, and fish production: subsidies in salmonid watersheds. *Fisheries*. 35: 373-387.

Wipfli, M.S., Hudson, J., and Caouette, J.P. 1998. Influence of salmon carcasses on stream productivity: response of biofilm and benthic macroinvertebrates in southeastern Alaska, USA. Canadian Journal of Fisheries and Aquatic Sciences **55**: 1503–1511.

Wolman. M. 1954. A method of sampling coarse bed material. American Geophysical Union Transactions. 35:951-956.

Appendix A – Locations of sampling sites in the Lower Bridge River.

Index Monitoring Site Locations

Reach	Index Site	Approx. River Km	UTM Coordinates	
			Easting	Northing
4	A	39.9	555649	5626314
3	B	36.5	558176	5627005
	C	33.3	558109	5629483
	D	30.4	556469	5631133
	E	26.4	556969	5634487
2	F	23.6	559356	5634485
	G	20.0	562537	5630967
1	H	11.3	567796	5629231
	I	7.5	570496	5627006
	J	3.9	572675	5624878
	K	1.2	574432	5623131
Yalakom River	Yal_A	3.8	555989	5637089
	Yal_B	0.1	558281	5635123

[a] UTM zone is 10U.

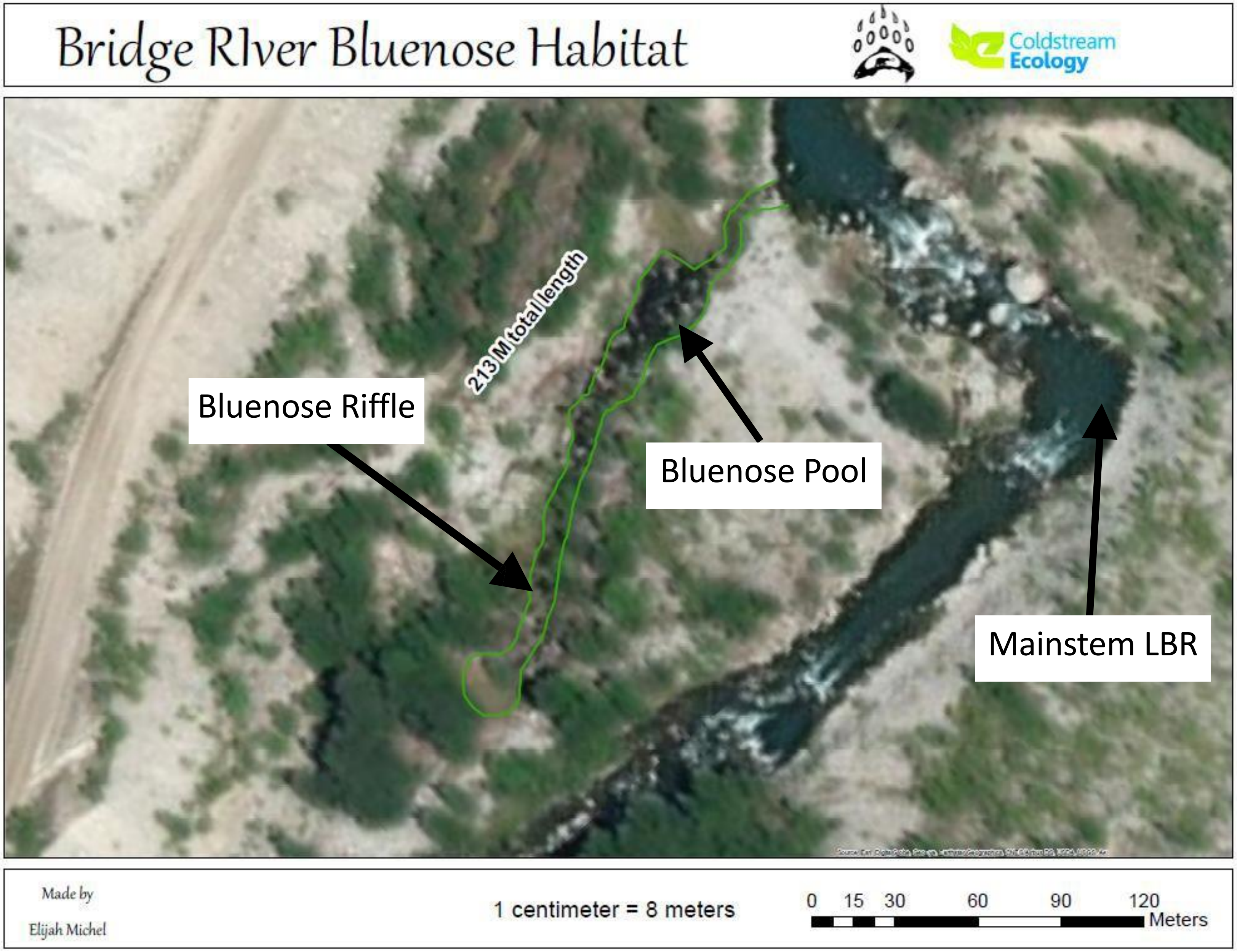
Off-channel Fish Sampling Locations

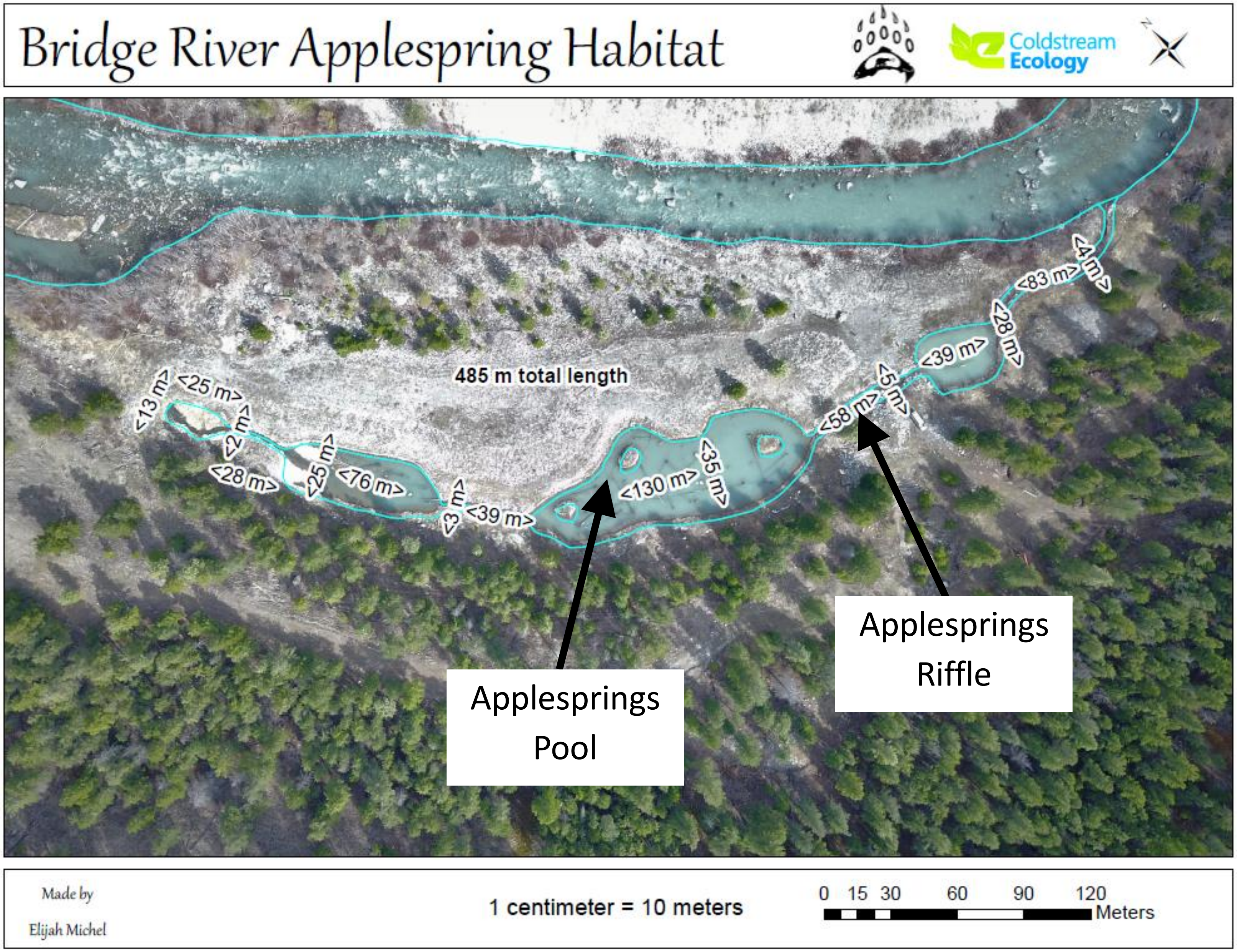
Field Component	Reach	Site	Approx. River Km	Bank	UTM Coordinates	
					Easting	Northing
Juvenile Salmonid Habitat Use (Mitigation Effectiveness Monitoring)	4	Bluenose Outflow	39.2	L	556654	5626903
		Bluenose Pond & Upper Intake	39.2	L	556651	5626901
	1	Applesprings Outflow	11.3	R	567992	5629097
		Applesprings Upper Sidechannel	11.3	R	567798	5629205
		Applesprings Middle Sidechannel	11.3	R	567832	5629153
		Applesprings Lower Sidechannel	11.3	R	567928	5629103
Off-channel Stock Assessment	4	Bluenose Riffle	39.2	L	556600	5626876
		Bluenose Pool	39.2	L	556626	5626888
	1	Applesprings Riffle	11.3	R	567955	5629095
		Applesprings Pool	11.3	R	567813	5629192

Juvenile Stock Assessment Site Locations (LBR Mainstem)

Reach	Site	Approx. River Km	Bank	UTM Coordinates	
				Easting	Northing
4	40500	40.5	L	555590	5626147
	40200	40.2	L	555649	5626314
	40100	40.1	R	555717	5626271
	39401	39.4	R	556482	5626807
	39400	39.4	R	556482	5626807
	39201	39.2	L	556759	5626389
	39200	39.2	L	556761	5626653
	37300	37.3	L	557900	5626750
	37200	37.2	R	558000	5626725
	37150	37.2	L	558075	5626750
	37001	37.0	L	558200	5626750
	37000	37.0	L	558225	5626775
3	35941	35.9	L	558174	5627021
	35940	35.9	R	558191	5627003
	33824	33.8	L	558250	5629725
	33800	33.8	L	558250	5629850
	32440	32.4	L	557800	5630250
	32432	32.4	L	557766	5630280
	32284	32.3	L	557725	5630314
	32211	32.2	L	557625	5630301
	32206	32.2	L	557627	5630344
	30721	30.7	L	556550	5631375
	30700	30.7	R	556550	5631375
	29300	29.3	L	556010	5632098
	29010	29.0	R	555733	5632232
	29000	29.0	R	555724	5632258
	28533	28.5	L	555900	5632500
	27600	27.6	R	556242	5633466
	27500	27.5	L	556291	5633555
	27450	27.5	L	556324	5633622
	26100	26.1	L	557174	5634605
	26000	26.0	L	557356	5634661

Reach	Site	Approx. River Km	Bank	UTM Coordinates	
				Easting	Northing
2	24841	24.8	L	558400	5634600
	24802	24.8	R	558450	5634575
	23900	23.9	L	559250	5634450
	23800	23.8	L	559350	5634550
	23602	23.6	L	559450	5634475
	23601	23.6	R	559450	5634475
	23301	23.3	L	559500	5634325
	22801	22.8	L	559743	5633911
	22800	22.8	L	559732	5633929
	21801	21.8	L	560494	5633459
	21800	21.8	R	560422	5633455
	21601	21.6	L	560748	5633331
	21600	21.6	R	560648	5633291
	21401	21.4	L	560849	5633130
	21400	21.4	R	560849	5633130
	20001	20.0	L	562500	5630900
	20000	20.0	R	562500	5630900
1	11350	11.4	R	567798	5629277
	11301	11.3	R	567817	5629276
	11300	11.3	R	567850	5629246
	7250	7.3	R	569867	5627075
	7201	7.2	L	569932	5627092
	7200	7.2	R	569888	5627048
	3999	4.0	L	572566	5625155
	3950	4.0	R	572636	5625094
	3901	3.9	L	572632	5625019
	1250	1.3	L	574437	5623140
	1200	1.2	L	574360	5623091
	1125	1.1	L	574377	5623127


Bridge RIver Bluenose Habitat
Coldstream Ecology
213 M total length
Bluenose Riffle
Bluenose Pool
Mainstem LBR
Made by
Elijah Michel
1 centimeter = 8 meters
0
15
30
60
90
120
Meters


Bridge River Applespring Habitat
Coldstream Ecology
485 m total length
<13 m>
<25 m>
<2 m>
<28 m>
<25 m>
<76 m>
<3 m>
<39 m>
<130 m>
<35 m>
<58 m>
<5 m>
<39 m>
<28 m>
<83 m>
<4 m>
Applesprings Pool
Applesprings Riffle
Made by
Elijah Michel
1 centimeter = 10 meters
0 15 30 60 90 120
Meters

Appendix B – Description of Hierarchical Bayesian Model Estimating Juvenile Salmonid Abundance and Biomass in the Lower Bridge River

Our hierarchical Bayesian Model (HBM) is similar to model I of Wyatt (2002 and 2003). The model consists of two levels or hierarchies. Site-specific estimates of detection probability (also referred to as catchability) and densities at the lowest level of the hierarchy are considered random variables that come from hyper-distributions of catchability and density at the higher level. The HBM jointly estimates both site- and hyper-parameters. The process component of the model assumes that variation in fish abundance across sites can be modeled using a Poisson/log-normal mixture (Royle and Dorazio 2008). That is, abundance at-a-site is Poisson-distributed with a site-specific log-normally distributed mean. The observation component of the model assumes that variation in detection probability across sites can be modeled using a beta distribution, and that electrofishing catches across sites and passes vary according to a binomial distribution which depends on site-specific detection probability and abundance.

In the following description "fish" refers to one species-age group combination. Greek letters denote model parameters that are estimated. Capitalized Arabic letters denote derived variables that are computed as a function of parameters. Lower case Arabic letters are either subscripts, data, or prior parameter values.

We assumed that the number of fish captured, c, by electrofishing in year y at site i on pass j followed a binomial distribution (*dbin*) described by the detection probability (or catchability) θ, and the number of fish in the sampling arena, N:

(1)

We assumed that detection probability was constant across passes but could vary among sites. The number of fish remaining in the sampling area after pass j was the difference between the number present prior to pass j and the catch on pass j:

(2)

These two equations describe the binomial model on which removal estimators are based (e.g., Moran 1951, Otis et al. 1978). Inter-site variation in detection probability was assumed to follow a beta hyper-distribution (*dbeta*), with year-specific parameters:

(3)

Inter-site variation in fish density (λ) in log space was assumed to follow a normal (*dnorm*) hyper-distribution:

(4)

Here μ and τ are the mean and precision of the normal probability distribution ($\tau = \sigma_\lambda^{-1}$) specifying the hyper-distribution of log density for each reach and year. The number of fish present at site *i* prior to the first electrofishing pass ($N_{y,i,1}$) followed a poisson distribution with an expected value determined by the product of site area, *a*, and fish density drawn from the hyper-distribution (Equation 4):

(5)

To compute the total abundance of fish in a reach we also needed an estimate the number of fish in the areas of the river that we did not sample. As most of our sampling was conducted along the shorelines, we partitioned the wetted area of the river into one of 3 categories: the shoreline area that was sampled, the shoreline area that was not sampled, and the centre of the channel that in most cases was not sampled. The total abundance in reach *r* and year *y*, $Ntot_{y,r}$, was the sum of the estimates from sampled shoreline sites within the reach, *Nss*, the estimate for the unsampled shoreline, *Nus*, and abundance in the unsampled centre channel area (*Nuc*) for that reach and year:

(6)

The number of fish in the sampled shoreline was the sum of abundances of all sites within the reach:

(7)

Abundance in the unsampled shoreline (*Nus*) was computed as the product of the transformed mean density from the log-normal density hyper distribution (μ_{λ}) with log-normal bias correction ($0.5\tau^{-1}_{\lambda}$), and the area of the unsampled shoreline in the reach. The area of the unsampled shoreline is the area of the shoreline zone (the product of twice the length of the reach (l) and the average width of sampled area, w, less the total area that was sampled in the reach:

(8))

The number of fish in the centre of the channel (*Nuc*) was computed based on the abundance in the shoreline zone (*Nss*+*Nus*) and estimates of the proportion of the total population that was in the shoreline zone (ρ).

(9)

The parameter ρ is calculated for each reach, *r*, and flow period, *f*, and depends on the average width of electrofishing sites in each reach relative to the distribution of fish from shore determined from the field study described earlier. We assumed that the number of fish in the micro-habitat study ($h_{f,r}$) between the shoreline and the average width of electrofishing sites ($w_{y,r}$) in any year-reach strata was a binomially distributed random variable that depended on $\rho_{y,r}$ and the total number of fish observed in the micro-habitat study for that strata ($m_{f,r}$).

(10)

In Reach 3 during the baseline period the total wetted width was sampled. Hence $w_{y,r}$ is the average wetted width of the reach so the total wetted area of the reach is l_3w_3 and the multiplier 2 in equation 8 is not used. Also $\rho = 1$ in Equation 9 and consequently *Nuc*=0.

We estimated the effect of the flow release in each reach as the difference in the estimated average abundance between the treatment and baseline years (Δ_r) for age-0 fish as:

$$\Delta_r = \frac{\sum_{y=2001}^{2008} N_{y,r}}{8} - \frac{\sum_{y=1996}^{1999} N_{y,r}}{4}$$

(11)

Data for the year 2000 were not used as the change in flow occurred midway through the growing season and it is unclear how age-0 fish would be affected. The overall effect of flow in the study area Δ, which includes the contribution from the re-wetted Reach 4, is the difference in the average abundance of three Reaches (2-4) during the treatment period and the average abundance for Reaches 2 and 3 for the baseline period:

$$\Delta = \frac{\sum_{y=2001}^{2008} \sum_{r=2}^{4} N_{y,r}}{8} - \frac{\sum_{y=1996}^{1999} \sum_{r=2}^{3} N_{y,r}}{4}$$

(12)

For age-1 trout we considered fish sampled in September 2000 to be part of the baseline period as they would have experienced the increased flows for only a month just before sampling, representing <10% of their life as free-swimming fish. We did not use data for 2001 for the treatment period as these fish would have experienced baseline flows during their first 2-3 months after emergence from spawning gravels, which may have affected survival during this important early life stage. The summation indices in Equations 11 and 12 were adjusted accordingly for this age group.

Posterior distributions of model parameters were estimated using WinBUGS (Spiegelhalter et al. 1999) called from the R2WinBUGS (Sturtz et al. 2005) library from R (R Development Core Team 2009). Prior distributions for hyper-parameters and related transformations are given in Table 1. Posterior distributions were based on taking every second sample from a total of 5000 simulations after excluding the first 2000 to remove the effects of initial values.

The HBM was able to converge in all years using uninformative priors for both age-0 rainbow trout and age-0 chinook salmon (Table 1). For age-1 rainbow trout and age-0 coho salmon, depletion data were sparse for Reach 2 (there were small total catches at many sites within the reach). In these cases, the estimated abundance and detection probability at each site were highly confounded as the model was not able to distinguish estimates of high abundance and low detection probability with the converse. This uncertainty resulted in very low estimates of the precision of the hyper-distribution in log fish density across sites (τ_λ in Equation 4). To avoid unrealistically low estimates of precision, which in turn would lead to overestimates of abundance in the unsampled shoreline zone because of the bias correction term (Equation 8) we used a more informative distribution for these 2 species-age groups (Table 2). Following recommendations by Gelman (2006), the half-Cauchy or folded *t*-distribution prior was used to constrain σ_λ and achieve convergence.

The HBM had difficulty reaching convergence based on data from recent years due to low catches for some species and age groups (e.g. age-0 chinook). Site-specific estimates of capture probability, which drive estimates of the hyper-distribution of capture probability, depend on the magnitude of the reduction in catches across passes.There is no information about capture probabilty at a site if no fish of a given species-age class are captured, and very little information when the catch is very low. If this pattern occurs at many sites, the hyper-distribution of capture probability will be poorly defined and more information on capture probability in the prior distribution is required to obtain reliable estimates of capture probability and abundance.

In the original application of the HBM we used an uninformative prior for the mean capture probability across sites centered at 0.5 (beta distribution with parameters beta(1,1)), and a minimally informative prior for the standard deviation in capture probabilities across sites (half-cauchy distribution with scale parameters 0 and 0.3, see Gelman 2006). To obtain more reliable estimates, we used a more informative prior on the mean capture probability across sites. The prior was still centered at 0.5 (beta(50,50)), but has a uniform prior on the precison (inverse of variance) of capture probability across sites (unif(10,500)) which constrained the maximum extent of variation in capture probability aross sites. To be consistent, we applied the revised priors to all species and age classes.

In cases where capture probability was well defined in all years because the species-age class was abundant and widely distributed across sites (e.g. Rb-0), model estimates based on uninformative and minimally informative priors were very similar. Uncertainty in capture probability (Fig. A1) and abundance (Fig. A2) estimates was slightly lower when the more informative priors were used. In cases where catch was low and fish were absent from many sites (Ch-0 in years > 2003, Co-0 1996-2000), the more informative priors led to reduced variation in capture probability estimates across years. In the case of juvenile chinook salmon, the original priors resulted in a decline in capture probability over time (Fig. A1, bottom-right panel). That pattern was suspect because it was inconsistent with the stable trends for other species-age classes (Rb-0, Co-0) where capture probability was well defined. Both electrofishing methods and flows at the time of sampling were stable during this period, which should lead to stable capture probabilities. The revised priors stabilized and increased Ch-0 capture probability across years (Fig. A1) such that they were more consistent with trends from species-ages that were well determined. For the other species, revised capture probabilities tended to be higher when catches were low. This in turn resulted in a decrease in estimated abundance in many years and a large reduction in the uncertainty in annual abundance estimates.

To better understand the effects of low catch and occupancy on estimates of abundance from the HBM, we simulated a set of catch depletions across 50 sites based on a zero-inflated log-normal distribution of fish densities. We then applied the HBM to the simulated data and compared estimates of abundance and capture probability to the values used drive the simulation. We found that capture probability was underestimated and abundance was overestimated, and the extent of bias increased with the degree of zero-inflation in simulated fish densities. For example, when we assumed that 30% of the sample sites were unoccupied and mean density was low, abundance was overestimated by 50%. This occurred because the HBM assumes a log-normal distribution in fish density across sites and does not explicitly account for zero-inflation. When the true distribution of densities is a zero-inflated, a better fit is obtained by lowering the capture probability because this increases the likelihood for sites with low or zero catch. This in turn results in an overestimate of abundance. Increasing information on capture probability in prior distributions reduces the tendency of the model to underestimate capture probability and therefore reduces the extent of positive bias in abundance. We attempted to revise the structure of the HBM to directly estimate the extent of zero-inflation, but this additional parameter was not estimable because the degree of zero-inflation and the magnitude of capture probability were confounded. That is, the model could not distinguish between cases where capture probabiltiy was high and a large fraction of sites were unoccupied, and the opposte pattern. Although directly accounting for zero-inflation in animal distributions can be accomodated in a mark-recapture framework (Conroy et al. 2008), confounding between capture probability and abundance precludes its use in depletion-based studies.

References

Conroy, M.J., Runge, J.P., Barker, R.J., Schofield, M.R., and C.J. Fonnesbeck. 2008. Efficient estimate of abundance for patchily distributed populations via two-phase, adaptive sampling. Ecology 89:3362-3370.

Gelman, A. 2006. Prior distributions for variance parameters in hierarchical models. Bayesian analysis 1: 515-533.

Royale, J.A., and R.M. Dorazio. 2008. Hierarchical modelling and inference in ecology. Academic Press, Amsterdam. 444 pp.

Spiegelhalter, D.J., Thomas A., Best, N.G., and Lunn, D. 1999. WinBUGS User Manual: Version 1.4. Cambridge: MRC Biostatistics Unit, 60 pp.

Sturtz, S., Legges U., and Gelman A. 2005. R2WinBGS: a package for running WinBUGS from R. Journal of Statistical Software 3: 1-16.

Wyatt, R.J. 2002. Estimating riverine fish population size from single- and multiple-pass removal sampling using a hierarchical mode. Can J. Fish. Aquat. Sci. 59:695-706.

Wyatt, R.J. 2003. Mapping the abundance of riverine fish populations: integrating hierarchical Bayesian models with a geographic information system (GIS). Can. J. Fish. Aquat. Sci. 60: 997-1007.

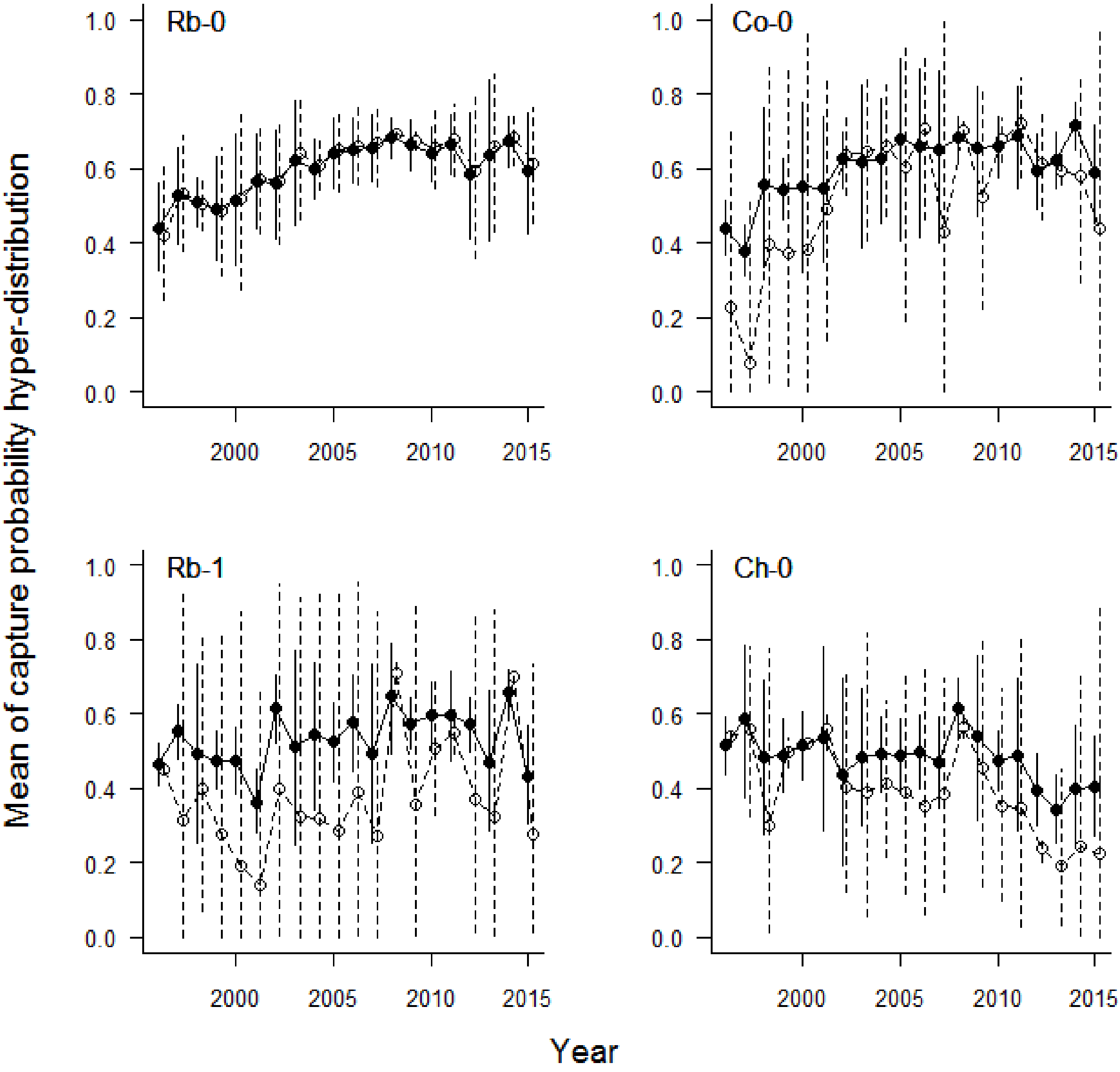


Figure B1. Annual estimates of the mean (with 90% credible interval) of the capture probability hyper-distribution (distribution of capture probability across sites) based on the HBM with more restrictive priors for the capture probability hyper-distribution (solid symbols). Also shown are estimates based on uninformative capture probability priors used in Bradford et al. (2011, open symbols).

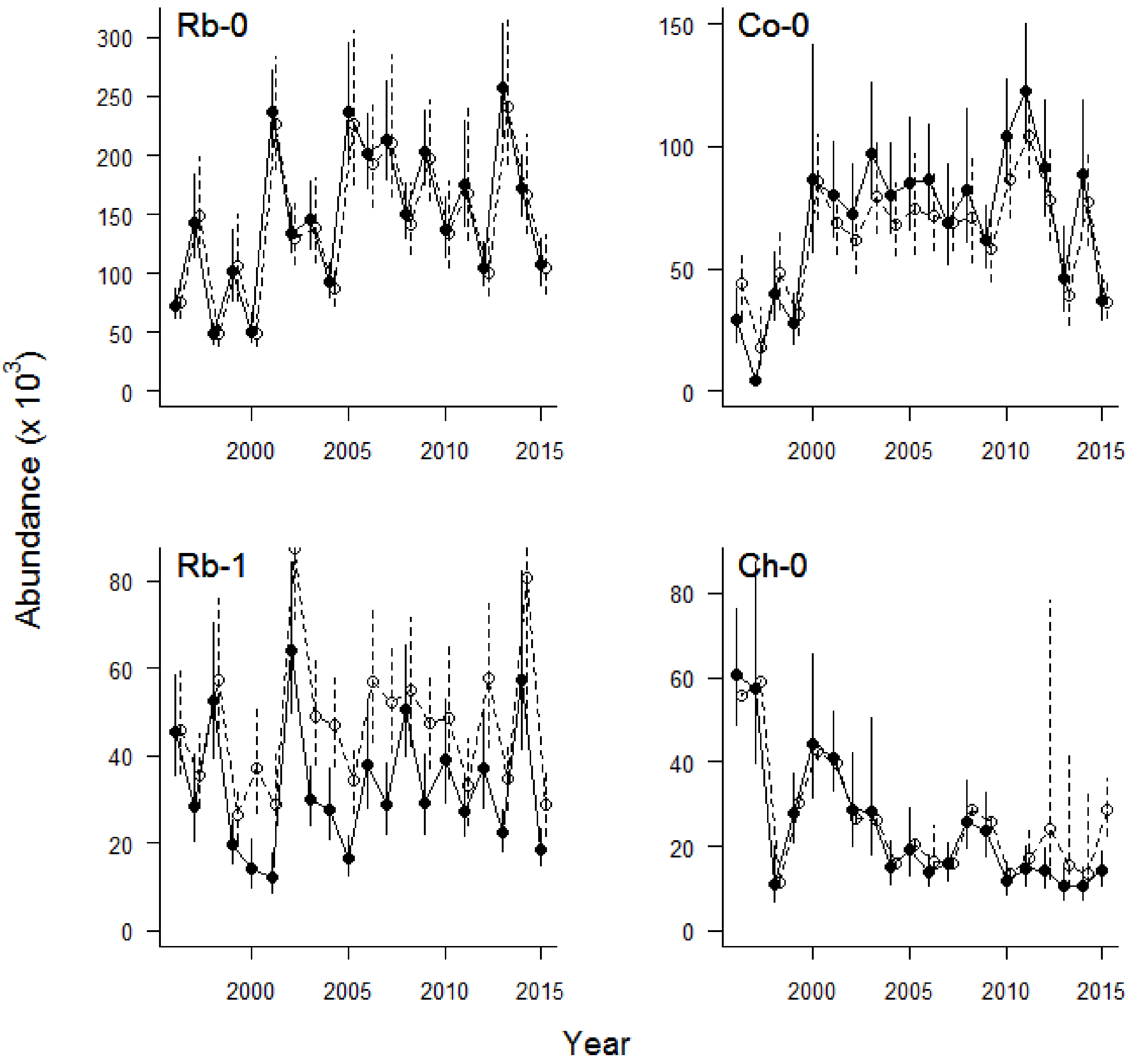


Figure B2. Annual estimates of abundance (all reaches combined) based on the HBM with a more restrictive prior (solid symbols). Also shown are estimates based on the uninformative priors used in Bradford et al. (2011, open symbols).

Appendix C – Mixed Effects Model Code and Covariate Results

BUGS code for mixed effects model. Text in green are comments, and bolded text represents data, and blue text represent probability distributions.

```
BetaC~dnorm(0,1.0E-03)  #covariate effect

#Priors on fixed effects of reach and flow, year-specific random effect variance, and process variance
for(ir in 1:2) {BetaR[ir]~dnorm(0,1.0E-03)}     #fixed reach effect
BetaR[Nreaches]<- -100                          #0 fish in reach 4 at 0 cms in log space

for(ir in 1:Nreaches){
      for(itr in 1:Ntreats){
            BetaF[ir,itr]~dnorm(0,1.0E-03)      #fixed treatment effects
      }
}
yrSD~dunif(0.001,10);yrPrec<-pow(yrSD,-2)
proSD~dunif(0.001,10);  proPrec<-pow(proSD,-2)  #process error

for (i in 1:Nyrs) {BetaY[i]~dnorm(0,yrPrec)}    #random year effect

#Loop over samples (reach*year)
for(i in 1:LastTreat0){ #samples under treatment 0 that don’t depend on treatment effect
      pro_dev[i]~dnorm(0,proPrec)
      pred[i] <- BetaR[Reach[i]] + BetaY[Year[i]] + BetaC*X[i] + pro_dev[i]   #pred log den or biomass
      obs_mu[i]~dnorm(pred[i],obs_prec[i])      #likelihood comparing prediction to observation
      RE[i]<-BetaY[Year[i]] + pro_dev[i]        #random effects needed for multi-level r2 calculation
}

for(i in (LastTreat0+1):Nsamps){#samples effected by treatment flows that include treatment effect BetaF
      pro_dev[i]~dnorm(0,proPrec)
      pred[i] <- BetaR[Reach[i]] + BetaF[Reach[i],Ftreat[i]] + BetaY[Year[i]] + BetaC*X[i] + pro_dev[i]
      obs_mu[i]~dnorm(pred[i],obs_prec[i])
      RE[i]<-BetaY[Year[i]] + pro_dev[i]
}
```

BUGS code for mixed effects model continued.

```
#Multi-level r2 computations (proportion of variation in pred[] explained by fixed effects and Pearson r2)
muRE<-sum(RE[])/Nsamps; mupred<-sum(pred[])/Nsamps; muobs_mu<-sum(obs_mu[])/Nsamps
for(i in 1:Nsamps){
      ssRE[i]<-pow(RE[i]-muRE,2)                  #sums of squares on random effects
      ssTOT[i]<-pow(pred[i]-mupred,2)             #sums of squares on fixed+random effects
      data_res[i]<-pow(obs_mu[i]-pred[i],2)       #observed - predicted (squared residual)
      data_ssTOT[i]<-pow(obs_mu[i]-muobs_mu,2)
}

FE_r2<-1.0-sum(ssRE[])/sum(ssTOT[]) #Proportion of total sums of squares explained by fixed effects
data_r2<-1.0-sum(data_res[])/sum(data_ssTOT[])  #standard Pearson r2 calculation

#Calculate density (fish/m) or biomass (g/m) (DenBio) and abundance ('000s) or total biomass (kg) (N) by
#reach and treatment (including @ 0 cms)
for(ir in 1:Nreaches){
      DenBio[ir,1]<-exp(BetaR[ir])        #0 cms
      N[ir,1]<-DenBio[ir,1]*TotLength[ir]*0.001

      for(itr in 2:(Ntreats+1)){
            DenBio[ir,itr]<-exp(BetaR[ir]+BetaF[ir,itr-1])
            N[ir,itr]<-DenBio[ir,itr]*TotLength[ir]*0.001
      }
}
for(itr in 1:(Ntreats+1)){Ntot[itr]<-sum(N[1:Nreaches,itr])} #total abundance across reaches
```

Table C1 **Results from the mixed effects model for selected covariates on Density (fish/m) and Biomass (g/m) by species-age.**

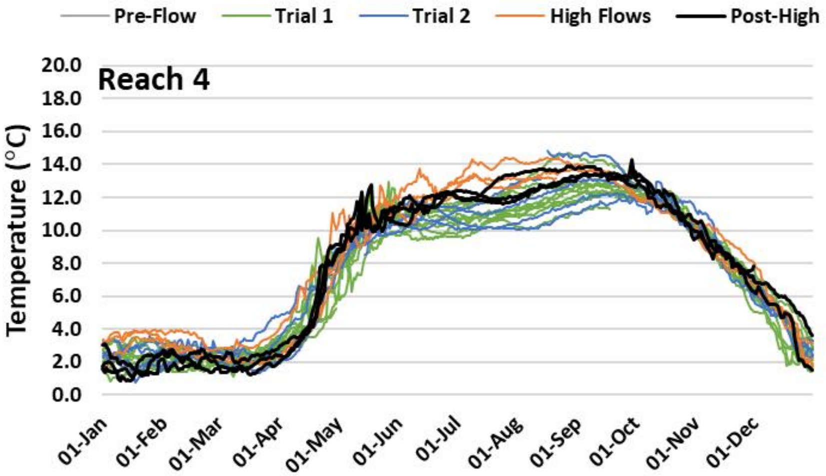
Species-Age	Parameter	Co-variate	Effect Size		Proportion Explained		ΔDIC	DIC	SD
			Mean	CV	Fixed Effects	Data			
Mykiss fry (RB-0)	Density (fish/m)	Null	NA	NA	0.80	0.96	0.56	-12.613	NA
		muSRP	0.03	2.08	0.55	0.96	1.11	-12.066	0.07
		muBugs	-0.07	1.77	0.73	0.96	1.15	-12.023	0.12
		muSummerTemp	0.03	10.82	-2.62	0.96	2.20	-10.975	0.28
		Chil_SH_Sp.km_1	-0.05	1.64	0.79	0.96	0.00	-13.174	0.08
		Chil_SH_Sp.km_2	0.09	2.55	0.65	0.96	1.14	-12.034	0.23
		Thom_SH_Sp.km_1	-0.07	1.38	0.77	0.96	0.10	-13.072	0.10
		Thom_SH_Sp.km_2	-0.06	10.16	0.34	0.96	0.91	-12.267	0.58
	Biomass (g/m)	Null	NA	NA	0.72	0.94	1.50	-15.513	NA
		muSRP	-0.27	2.73	-3.92	0.95	4.35	-12.669	0.74
		muBugs	-0.16	0.34	0.74	0.94	0.00	-17.014	0.06
		muSummerTemp	0.16	4.16	-9.99	0.95	3.63	-13.382	0.67
		Chil_SH_Sp.km_1	-0.03	2.20	0.70	0.95	1.89	-15.128	0.07
		Chil_SH_Sp.km_2	-0.03	11.95	0.49	0.94	2.05	-14.968	0.34
		Thom_SH_Sp.km_1	-0.07	1.17	-0.66	0.95	2.56	-14.452	0.08
		Thom_SH_Sp.km_2	-0.44	2.55	-2.16	0.94	2.25	-14.76	1.12
Mykiss parr (RB-1)	Density (fish/m)	Null	NA	NA	0.87	0.95	0.00	37.951	NA
		muSRP	-0.06	1.13	0.87	0.95	0.93	38.879	0.07
		muBugs	-0.02	4.10	0.87	0.95	0.63	38.584	0.08
		muSummerTemp	0.01	14.73	0.86	0.95	0.65	38.6	0.08
		Chil_SH_Sp.km_1	0.11	0.74	0.87	0.96	0.14	38.096	0.09
		Chil_SH_Sp.km_2	0.09	1.12	0.87	0.95	0.60	38.554	0.10
		Thom_SH_Sp.km_1	0.09	1.00	0.87	0.95	1.28	39.228	0.09
		Thom_SH_Sp.km_2	0.05	2.01	0.86	0.95	0.31	38.266	0.11
	Biomass (g/m)	Null	NA	NA	0.90	0.95	0.11	35.624	NA
		muSRP	-0.05	1.07	0.90	0.95	1.20	36.712	0.06
		muBugs	-0.09	0.79	0.90	0.95	0.35	35.863	0.07
		muSummerTemp	0.01	5.42	0.89	0.95	0.87	36.383	0.07
		Chil_SH_Sp.km_1	0.15	0.48	0.91	0.95	0.00	35.514	0.07
		Chil_SH_Sp.km_2	0.09	0.97	0.90	0.95	0.61	36.129	0.08
		Thom_SH_Sp.km_1	0.15	0.52	0.90	0.95	0.25	35.764	0.08
		Thom_SH_Sp.km_2	0.08	1.12	0.90	0.95	1.16	36.672	0.09
Coho fry (CO-0)	Density (fish/m)	Null	NA	NA	0.83	0.92	0.16	52.341	NA
		muSRP	-0.05	1.67	0.83	0.92	0.28	52.467	0.08
		muBugs	-0.05	1.65	0.82	0.92	0.75	52.932	0.09
		muSummerTemp	-0.17	0.55	0.86	0.92	2.15	54.335	0.09
		CO_Sp.km_1	-0.01	18.75	0.83	0.92	0.38	52.566	0.10
		CO_Sp.km_2	0.15	0.83	0.83	0.92	0.00	52.182	0.13
	Biomass (g/m)	Null	NA	NA	0.82	0.92	0.32	51.097	NA
		muSRP	-0.04	1.98	0.81	0.92	0.02	50.796	0.08
		muBugs	-0.11	0.74	0.81	0.92	0.00	50.775	0.08
		muSummerTemp	-0.14	0.66	0.84	0.92	1.36	52.133	0.10
		CO_Sp.km_1	-0.01	12.48	0.81	0.92	0.33	51.102	0.10
		CO_Sp.km_2	0.10	1.35	0.81	0.92	0.67	51.444	0.13

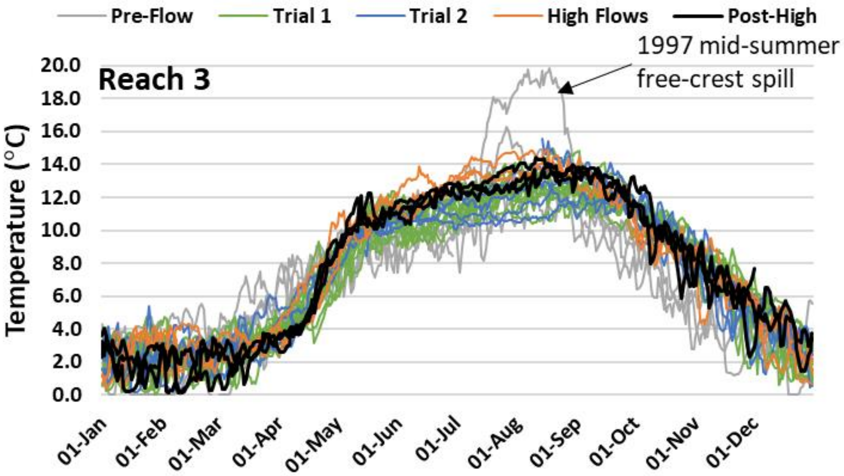
Table C1 Continued.

Species-Age	Parameter	Co-variate	Effect Size		Proportion Explained		∆DIC	DIC	SD
			Mean	CV	Fixed Effects	Data			
Chinook fry (CH-0)	Density (fish/m)	Null	NA	NA	0.71	0.91	0.71	53.499	NA
		muSRP	0.11	0.70	0.71	0.91	0.80	53.59	0.08
		muBugs	0.02	4.90	0.71	0.91	1.78	54.565	0.09
		muSummerTemp	0.09	1.08	0.71	0.91	0.89	53.675	0.09
		muCH_IncubDays	0.09	3.31	0.71	0.91	1.75	54.542	0.30
		CH_Sp.km_1	-0.16	0.56	0.71	0.91	0.00	52.788	0.09
		CH_Sp.km_2	0.00	58.91	0.70	0.91	1.45	54.237	0.11
	Biomass (g/m)	Null	NA	NA	0.63	0.86	1.04	53.163	NA
		muSRP	0.12	0.61	0.63	0.86	0.00	52.119	0.07
		muBugs	-0.02	3.83	0.62	0.86	1.40	53.515	0.08
		muSummerTemp	0.07	1.17	0.63	0.86	1.25	53.371	0.08
		muCH_IncubDays	0.18	1.61	0.62	0.86	0.53	52.65	0.30
		CH_Sp.km_1	-0.14	0.61	0.62	0.86	0.08	52.2	0.08
		CH_Sp.km_2	-0.01	15.78	0.61	0.86	1.29	53.409	0.10

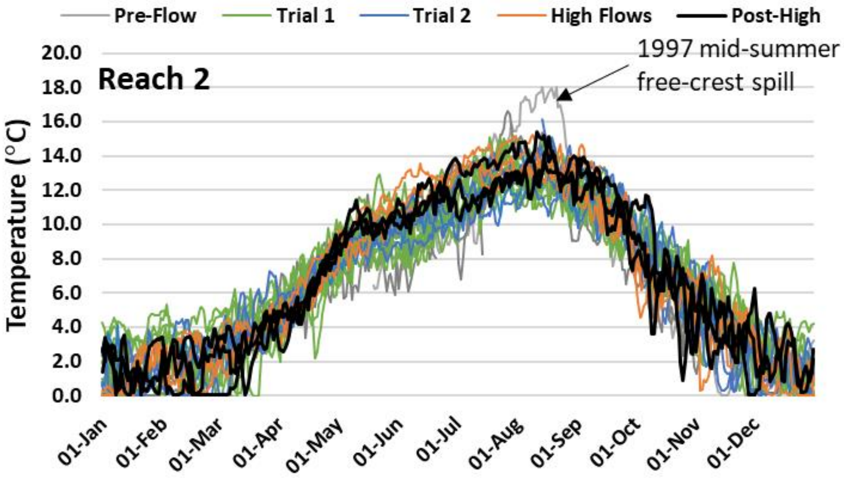
Appendix D – Mean Water Temperatures in the Lower Bridge River (by Reach) and the Yalakom River for each Study Year

Note: Each year is coloured according to flow trial.

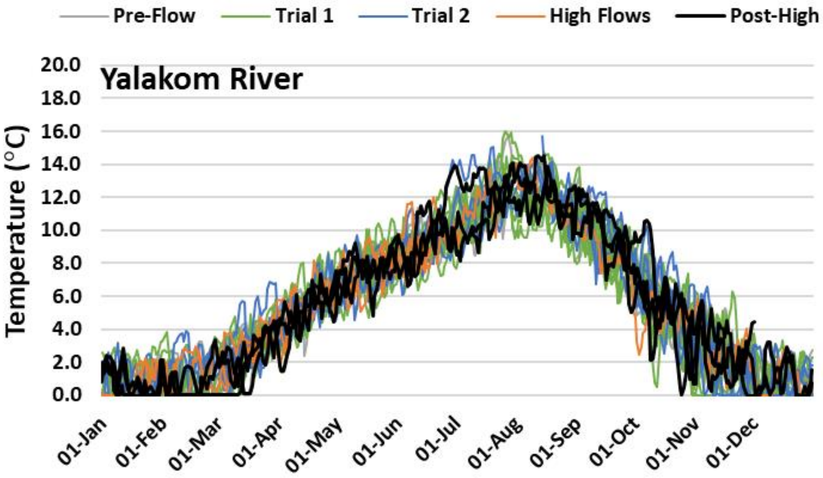




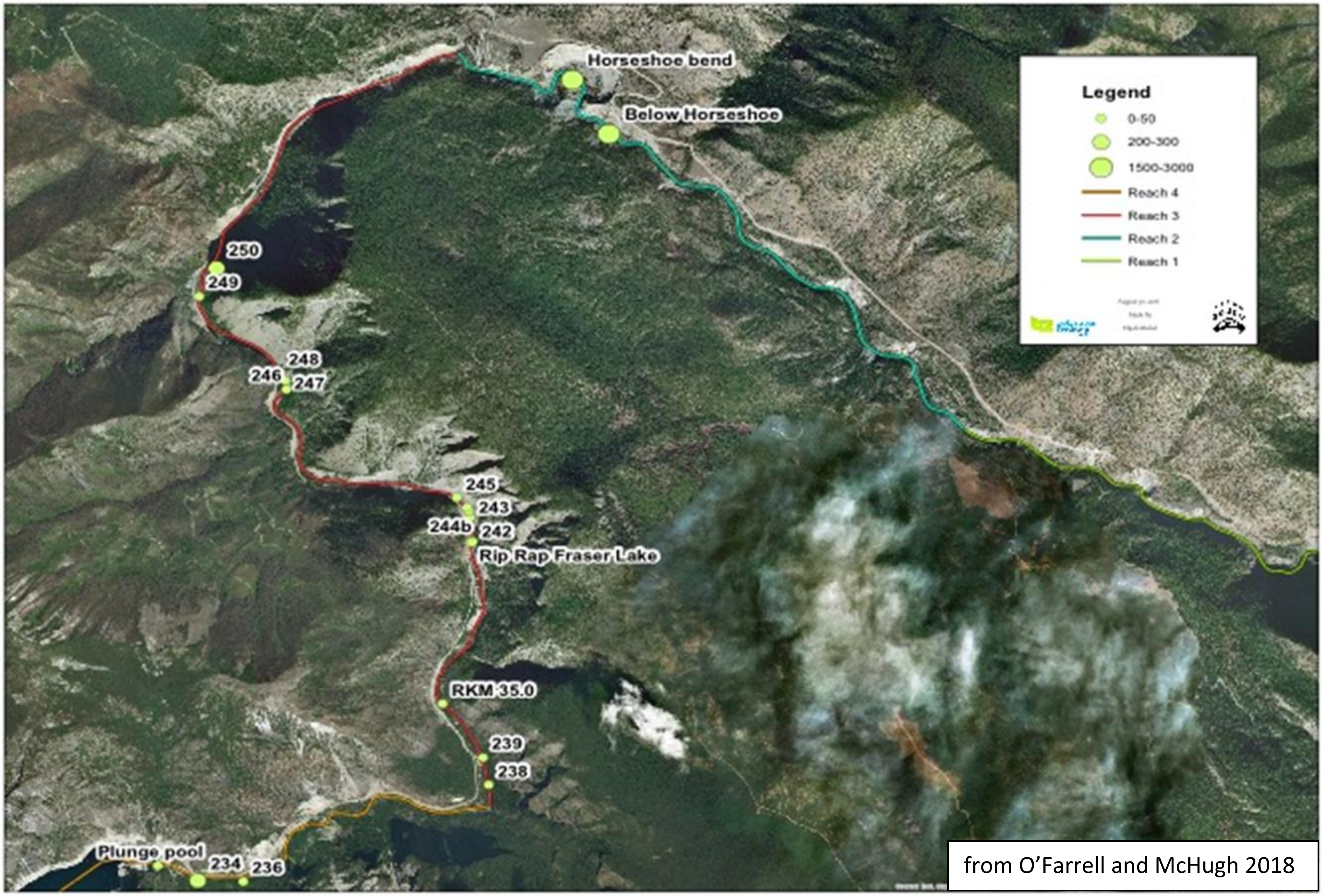
1997 mid-summer free-crest spill



1997 mid-summer free-crest spill



Appendix E – LBR Bank Erosion and Sediment Recruitment Sites assessed during High Flows in 2018.



from O'Farrell and McHugh 2018

Appendix F – Detailed Summary of Flow Rampdown Events and Fish Salvage Tallies

Table F1 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2021.

Year	Date	Event #	Ramp Duration (hours)	Start Flow ($m^3 \cdot s^{-1}$)	End Flow ($m^3 \cdot s^{-1}$)	Flow Change ($m^3 \cdot s^{-1}$)	Start Stage (cm)	End Stage (cm)	Stage Change (cm)	Mean Daily Rate (cm/hr)
2021	29 Jul	1	4	25.4	20.0	-5.4	163	154	-9	-2.1
	30 Jul	2	5	20.0	15.0	-5.0	154	144	-10	-2.1
High Flow Rampdown Summary		**2**	**5**	**25.4**	**15.0**	**-10.4**	**163**	**144**	**-19**	**-2.1 (Max.)**
2021	3 Aug	3	3	15.1	12.9	-2.2	144	138	-6	-2.4
	4 Aug	4	3	13.0	11.1	-1.9	138	133	-5	-2.0
	9 Aug	5	3	11.2	9.3	-1.9	133	128	-5	-2.0
	10 Aug	6	3	9.3	7.8	-1.5	128	123	-5	-2.0
	18 Aug	7	3	7.9	6.4	-1.5	123	118	-5	-1.7
	19 Aug	8	3	6.4	5.1	-1.3	118	111	-7	-2.3
	23 Aug	9	3	5.1	4.2	-0.9	111	106	-5	-2.0
	24 Aug	10	3	4.2	3.1	-1.1	106	99	-7	-2.7
	5 Oct	11	3	3.2	2.3	-0.9	98	91	-7	-2.1
	6 Oct	12	1	2.3	2.1	-0.2	90	89	-1	-1.3
	19 Oct	13	3	2.1	1.3	-0.8	89	84	-5	-2.0
WUP Rampdown Summary		**11**	**3**	**15.1**	**1.3**	**-13.8**	**144**	**84**	**-60**	**-2.7 (Max.)**

Table F2 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2019 and 2020.

Year	Date	Event #	Ramp Duration (hours)	Start Flow ($m^3 \cdot s^{-1}$)	End Flow ($m^3 \cdot s^{-1}$)	Flow Change ($m^3 \cdot s^{-1}$)	Start Stage (cm)	End Stage (cm)	Stage Change (cm)	Mean Daily Rate (cm/hr)
2020	21 Jul	1	4	15.1	12.9	-2.1	141	135	-6	-1.6
	22 Jul	2	4	13.0	11.0	-2.0	135	130	-6	-1.5
	28 Jul	3	4	11.1	9.3	-1.8	129	125	-3	-1.0
	29 Jul	4	5	9.3	7.7	-1.6	125	121	-4	-0.8
	4 Aug	5	4	7.7	6.4	-1.4	121	118	-4	-1.0
	5 Aug	6	4	6.4	5.1	-1.3	117	111	-6	-1.5
	6 Aug	7	4	5.1	4.1	-1.0	111	106	-5	-1.4
	7 Aug	8	5	4.1	3.0	-1.1	106	99	-6	-1.4
	5 Oct	9	7	3.0	1.7	-1.3	98	87	-11	-1.6
	6 Oct	10	3	1.7	1.5	-0.2	87	84	-3	-1.1
2020 Rampdown Summary			**4**	**15.1**	**1.5**	**-13.6**	**141**	**84**	**-57**	**-1.6 (Max.)**
2019	1 Aug	1	5	15.2	12.9	-2.3	144	138	-6	-1.1
	6 Aug	2	7	13.0	11.1	-1.9	138	133	-5	-0.7
	8 Aug	3	4	11.1	9.3	-1.8	133	128	-5	-1.3
	9 Aug	4	4	9.3	7.7	-1.6	128	123	-5	-1.3
	14 Aug	5	4	7.7	6.4	-1.3	122	118	-5	-1.2
	15 Aug	6	5	6.4	5.1	-1.3	117	111	-6	-1.3
	18 Aug	7	4	5.1	4.1	-1.1	111	106	-5	-1.3
	19 Aug	8	5	4.1	3.0	-1.1	106	99	-6	-1.3
	1 Oct	9	6	3.2	2.1	-1.0	98	91	-8	-1.3
	2 Oct	10	5	2.1	1.5	-0.6	90	84	-6	-1.2
2019 Rampdown Summary			**5**	**15.2**	**1.5**	**-13.7**	**144**	**84**	**-60**	**-1.3 (Max.)**

Table F3 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2018.

Year	Date	Event #	Ramp Duration (hours)	Start Flow $(m^3 \cdot s^{-1})$	End Flow $(m^3 \cdot s^{-1})$	Flow Change $(m^3 \cdot s^{-1})$	Start Stage (cm)	End Stage (cm)	Stage Change (cm)	Mean Daily Rate (cm/hr)
2018	4 Jul	1	4	102.0	82.6	-19.4	265	248	-16	-4.0
	10 Jul	2	5	82.9	66.9	-15.9	247	229	-18	-3.6
	17 Jul	3	5	67.2	55.2	-12.0	229	215	-15	-2.9
	18 Jul	4	5	55.3	44.2	-11.0	215	197	-18	-3.5
	24 Jul	5	8	44.4	35.1	-9.3	197	182	-14	-1.8
	25 Jul	6	8	35.1	27.1	-8.0	182	168	-14	-1.8
	31 Jul	7	6	27.2	20.0	-7.2	168	153	-15	-2.4
	1 Aug	8	7	20.0	15.1	-4.9	153	142	-11	-1.6
High Flow Rampdown Summary		**8**	**6**	**102.0**	**15.1**	**-86.9**	**265**	**142**	**-123**	**-4.0 (Max.)**
2018	2 Aug	9	7	15.1	11.0	-4.1	142	132	-10	-1.5
	8 Aug	10	4	11.1	9.3	-1.8	132	126	-5	-1.3
	9 Aug	11	4	9.3	7.7	-1.6	126	121	-5	-1.2
	15 Aug	12	4	7.8	6.4	-1.3	122 [a]	116 [a]	-6	-1.4
	16 Aug	13	5	6.4	5.2	-1.3	116 [a]	110 [a]	-6	-1.2
	21 Aug	14	5	5.2	4.1	-1.1	110 [a]	103 [a]	-7	-1.3
	22 Aug	15	6	4.1	3.0	-1.1	103 [a]	96 [a]	-8	-1.3
	2 Oct	16	6	3.1	2.1	-1.0	96	87	-9	-1.6
	3 Oct	17	4	2.1	1.5	-0.6	87	80	-6	-1.6
WUP Rampdown Summary		**9**	**5**	**15.1**	**1.5**	**-13.6**	**142**	**80**	**-62**	**-1.6 (Max.)**

[a] These values are based on the discharge-stage relationship (see Figure 3.39 in Section **Error! Reference source not found.**) since stage values for the Rkm 36.8 logger were not available on these dates in 2018.

Table F4 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2017.

Year	Date	Event #	Ramp Duration (hours)	Start Flow ($m^3 \cdot s^{-1}$)	End Flow ($m^3 \cdot s^{-1}$)	Flow Change ($m^3 \cdot s^{-1}$)	Start Stage (cm)	End Stage (cm)	Stage Change (cm)	Mean Daily Rate (cm/hr)
2017	28 Jun	1	7	126.9	109.2	-17.7	290	272	-17	-2.5
	4 Jul	2	7	111.3	96.6	-14.7	278	263	-15	-2.1
	7 Jul	3	7	97.2	79.6	-17.5	263	247	-15	-2.2
	11 Jul	4	4	80.4	67.1	-13.3	247	231	-16	-4.0
	12 Jul	5	4	67.2	55.1	-12.2	232	218	-13	-3.4
	13 Jul	6	4	55.2	44.7	-10.5	218	202	-16	-4.1
	18 Jul	7	8	45.1	35.1	-10.1	203	186	-17	-2.2
	19 Jul	8	8	35.1	26.6	-8.5	186	171	-15	-1.9
	20 Jul	9	7	26.6	19.8	-6.8	171	157	-13	-1.9
	21 Jul	10	6	19.8	14.9	-4.9	157	147	-10	-1.7
High Flow Rampdown Summary		**10**	**6**	**126.9**	**14.9**	**-112.0**	**290**	**147**	**-143**	**-4.1 (Max.)**
2017	1 Aug	11	7	15.3	11.0	-4.3	147	136	-12	-1.6
	9 Aug	12	4	11.1	9.2	-1.8	136	131	-5	-1.2
	10 Aug	13	4	9.3	7.7	-1.6	130	125	-5	-1.3
	15 Aug	14	3	7.7	6.4	-1.4	125	120	-5	-1.7
	16 Aug	15	4	6.4	5.1	-1.3	120	110	-10	-2.5
	22 Aug	16	4	5.1	4.1	-1.0	110	103	-7	-1.7
	23 Aug	17	4	4.1	3.0	-1.1	103	96	-8	-1.9
	26 Sep	18	5	3.1	2.3	-0.8	95	88	-7	-1.4
	27 Sep	19	3	2.3	1.5	-0.7	88	80	-8	-2.6
WUP Rampdown Summary		**9**	**4**	**15.3**	**1.5**	**-13.7**	**147**	**80**	**-67**	**-2.6 (Max.)**

Table F5 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2016.

Year	Date	Event #	Ramp Duration (hours)	Start Flow ($m^3 \cdot s^{-1}$)	End Flow ($m^3 \cdot s^{-1}$)	Flow Change ($m^3 \cdot s^{-1}$)	Start Stage (cm)	End Stage (cm)	Stage Change (cm)	Mean Daily Rate (cm/hr)
2016	20 Jun	1	8	96.5	80.6	-15.9	245	233	-12	-1.5
	22 Jun	2	7	80.7	67.1	-13.6	234	223	-10	-1.5
	29 Jun	3	7	67.9	55.3	-12.6	224	209	-15	-2.1
	5 Jul	4	8	56.0	45.2	-10.9	210	195	-16	-2.0
	12 Jul	5	7	45.5	35.7	-9.8	196	180	-16	-2.2
	19 Jul	6	7	36.0	27.6	-8.4	180	165	-15	-2.1
	20 Jul	7	6	27.6	20.6	-7.0	165	151	-14	-2.3
	25 Jul	8	7	20.8	15.1	-5.7	151	137	-14	-2.0
High Flow Rampdown Summary		**8**	**7**	**96.5**	**15.1**	**-81.4**	**245**	**137**	**-108**	**-2.3 (Max.)**
2016	5 Aug	9	6	15.3	13.2	-2.2	137	131	-6	-1.0
	8 Aug	10	4	13.2	11.1	-2.1	131	124	-7	-1.8
	9 Aug	11	4	11.1	9.4	-1.7	124	118	-6	-1.5
	10 Aug	12	4	9.4	7.7	-1.6	118	111	-7	-1.8
	17 Aug	13	4	7.8	6.4	-1.3	111	105	-6	-1.5
	18 Aug	14	4	6.4	5.1	-1.3	105	99	-6	-1.4
	23 Aug	15	4	5.1	4.1	-1.0	99	93	-6	-1.5
	24 Aug	16	5	4.1	3.0	-1.2	93	83	-10	-2.0
	27 Sep	17	4	3.1	2.2	-0.8	95	87	-8	-2.0
	28 Sep	18	3	2.3	1.5	-0.7	87	78	-9	-3.0
WUP Rampdown Summary		**10**	**4**	**15.3**	**1.5**	**-13.8**	**137**	**78**	**-59**	**-3.0 (Max.)**

Table F6 **Fish salvage tallies by species, ramp date and flow range – 2021 results (reaches 3 and 4 only). The high flow (>15 $m^3 \cdot s^{-1}$) and WUP flow results are separated in the table by a double line.**

Ramp Date and Flow Range ($m^3 \cdot s^{-1}$)	Species							Daily Totals
	Chinook	Coho	Mykiss	Bull Trout	Mountain Whitefish	Redside Shiner	Sculpin spp.	
29 Jul (25.4 - 20.0)	--- No fish salvaged ---							**0**
30 Jul (20.0 - 15.0)		18	21					**39**
3 Aug (15.1 - 12.9)	--- No fish salvaged ---							**0**
4 Aug (13.0 - 11.1)	--- No fish salvaged ---							**0**
9 Aug (11.2 - 9.3)		25	32				3	**60**
10 Aug (9.3 - 7.8)	4	96	72		3			**175**
18 Aug (7.9 - 6.4)	--- No fish salvaged --- [a]							**0**
19 Aug (6.4 - 5.1)	--- No fish salvaged --- [a]							**0**
23 Aug (5.1 - 4.2)	7	88	342	1				**438**
24 Aug (4.2 - 3.1)		64	121	1				**186**
5 Oct (3.2 - 2.3)		6	17					**23**
6 Oct (2.3 - 2.1)	1	3	14					**18**
19 Oct (2.1 - 1.3)	1	17	18					**36**
Species Totals	**13**	**317**	**637**	**2**	**3**	**0**	**3**	**975**

[a] The study area was inaccessible on 18 and 19 August 2021 due to the closure of Road 40 caused by wildfires in the area.