Bridge River Project Water Use Plan

Lower Bridge River Aquatic Monitoring

Implementation Year 8

Reference: BRGMON-1

BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 8 (2019) Results

Study Period: April 2019 to March 2020

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Executive Summary

Monitoring of juvenile salmonid production, periphyton and benthic communities and their habitat continued in 2019 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 2019 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. Two reference sites that were established in the Yalakom River in 2018 were again sampled in 2019.

The 2019 flows in the Lower Bridge River were the same as during Trial 2 (2011 – 2015). Given this similarity of flow and earlier findings that flow influenced patterns of benthic assemblages and juvenile salmonid production, we tested the hypothesis that the 2019 flows would reverse the declines in benthos abundance and diversity, and juvenile salmonid abundance found in Trial 3 compared to Trial 2. Invertebrate metrics used in this test were density of all taxa, density of all 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). 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. Analysis of fish salvage data were also included to provide guidance for shaping flow ramp down strategies from Terzaghi Dam.

For the benthic invertebrate results, this hypothesis was supported in data from Reach 3 but not from Reaches 4 and 2 where all metric values were at the low end or less than the range of values found during Trial 2. These reach-specific differences among metric values were found to be related to variation in bio-available N (measured as dissolved inorganic N and called DIN) and P (measured as soluble reactive P and called SRP) concentrations. These nutrient concentrations consistently showed a positive relationship with values of the invertebrate metrics. The addition of Reach 1 data showed that low temperature interacted with nutrient supply in limiting benthos abundance. Biomass of periphyton mediated the effects of nutrient supply and temperature on benthic invertebrates, including fish food organisms. Determining the relative importance of the various parameters influencing benthic communities was conducted using analysis of variance (ANOVA) blocked by flow trial, reach and the presence of pinks. This analysis highlighted a clear pink salmon effect on nutrients, periphyton biomass and invertebrate density and diversity.

At the 2019 flows, Reach 3 was enriched with nutrients from tributary inflows or groundwater or both. This nutrient supply supported high densities of fish food organisms that were similar to those found during Trial 2. Extreme phosphorus deficiency for algal growth contributed to low densities of invertebrates in Reaches 4 and 2, which did not occur during the Trial 2 years. Low SRP concentrations in Reach 4 were due to little phosphorus load from Carpenter Reservoir in 2019. Low SRP concentration in Reach 2 was due to dilution of nutrients in the Lower Bridge River by the Yalakom River that was extremely nutrient deficient. Negative effects of low temperature

in the fall (near 5°C) exceeded positive effects of high nutrient levels on benthos density in Reach 1. Enrichment from tributaries contributed to relatively high nutrient concentrations in Reach 1.

An overall conclusion from the 2019 monitoring of benthic communities and their habitat is that Peak flows exceeding 100 $m^3 \cdot s^{-1}$ that occurred in 2016 – 2018 yielded low densities of fish food organisms from physical forcing but at moderate Peak flows <20 $m^3 \cdot s^{-1}$ as in 2019 and during Trial 2, an interplay between nutrient concentration and temperature were the main drivers of fish food abundance.

For juvenile salmonids, the results were similar: the hypothesis was at least partially supported in data from Reach 3, where some recovery of fish production occurred, but not from Reaches 4 and 2 where juvenile size, condition and abundance were more on par with Trial 3 (high flow) results than the previous Trial 2 years. Total juvenile abundance for reaches 2, 3 and 4 was approx. 94,000 fish in 2019, which was an increase of approx. 15,000 – 31,000 fish relative to the three preceding high flow years (or 26,000 more than the Trial 3 average). However, the 2019 abundance was 82,000 – 240,000 fewer fish than the previous Trial 2 years (2011 – 2015), or 160,000 fewer fish than the Trial 2 average. Trends in biomass among flow treatments for all species and age classes closely matched those based on abundance because changes in average weight across flow treatments have been less significant than the changes in abundance.

The increased juvenile abundance from the Trial 3 (high flow) years to 2019 was entirely due to improved recruitment of mykiss fry and coho fry (i.e., by approx. 10,000–18,000 and 13,000–16,000 more fish relative to 2016 – 2018, respectively). Abundances of mykiss parr (8,000 fish) and chinook fry (12,000 fish) in 2019 were each similar to the estimates for these species-age classes from the Trial 3 years. The mykiss parr had recruited as Age-0+ fish under the last year of the Trial 3 flows in 2018, so they were not expected to have recovered this year. Chinook fry production has remained relatively low and stable for 10+ years (i.e., since the end of the Trial 1 period). By reach, highest abundances for mykiss (fry and parr) and coho (fry) were in Reach 3.

Mean weights of each species and age class were almost always highest (or among the highest) in each reach during the Trial 3 (high flow) years from 2016-2018 and in 2019 compared to the previous Trial 2 years (2011 – 2015). However, condition factor (Fulton's K) values were generally highest during trials 0, 1, or 2 (according to species/age class) and the lowest were generally during the Mod. Ops. years (i.e., Trial 3 and 2019). These results suggested that larger mean size was likely not due to improved growth, but that size selection based on flow magnitude may be the more likely explanation.

Interestingly, the reach-based recovery results for juvenile salmonids in 2019 also aligned with the benthic invertebrate and nutrient source information. This seems to suggest that potential food source dynamics and reach-based productivity metrics may be factors that explain why fish abundance improved in some reaches in 2019 and not others. It suggests that food source improvement contributed to some fish abundance recovery in Reach 3, whereas food source constraints may have been a limiting factor to fish recovery in reaches 2 and 4. However, mean

Fulton’s K score was lower in each of the study reaches in 2019 than during the other Trial 2 years, and values in Reach 3 were generally the same or lower than the other study reaches for each species. Since benthic invertebrate abundance reflects food source availability, then we would expect fish in Reach 3 to be the largest or in the best condition, which was not the case in 2019. Conversely, it is possible that higher food availability was contributing to better survival in Reach 3 in 2019, but it was density-dependent factors that were limiting mean weight and condition factor in that reach. Mean condition factors were higher during the previous Trial 2 years when fish abundances were 2- to 5-fold higher by reach; however, benthic invertebrate abundances were also substantially higher overall in those years. The results point to potential food source limitation hindering recovery in 2019; however, with only one year of post-high flow data to consult, this conclusion is still preliminary.

For stock-recruitment results, the 2019 data point was a stand-alone point for both coho and chinook which was not incorporated into the existing curves for Trial 2 or 3 since it represented the first post-high flow value (i.e., start of a new treatment) for each species. As such the existing curves (reported in Sneep et al. 2019) did not change. The 2019 datapoint for chinook reflected a very low spawner estimate in 2018 (n= 25; 95% confidence intervals: 14–44); though the escapement estimate may have been biased low due to the effect of fish fence operations on the spawner surveys that year (White et al. 2019). The spawner estimate was close to a cluster of other low escapement values near the origin of the x-axis, and the juvenile recruitment estimate was similar to all of the other values for trials 2 and 3 on the y-axis. The 2019 stock-recruitment data point for coho was approx. 37,000 fry below the asymptote of the Trial 2 curve despite a sizeable spawner return in 2018 (*n*= 1,245; 95% confidence intervals: 882–1,627), indicating that recruitment was poor for that spawner stock size compared to the Trial 2 years. The existing set of curves (based on 50% egg-to-fry survival) do not suggest that spawner escapements have been limiting recruitment; however, uncertainty remains about the steepness of the initial slope for these curves.

Salmonid abundance data were collected in Reach 1 for the first time in 2019. Results for this year highlighted that all of the target species and age classes were present in the reach, of which mykiss fry were the most abundant (~17,000 fish), followed by coho fry (~10,000 fish), chinook fry (4,000 fish), and then mykiss parr (~1,000 fish). The patterns of abundance among the species in Reach 1 were most similar to their relative contributions in Reach 2 during 2019. The total for all species and age classes (~32,000) was 1/3 of all juvenile salmonids in reaches 2, 3 and 4 combined (~94,000). However, due to the extensive length of Reach 1 (half of the total study area length), the lineal densities for all species/age class groups were lowest in Reach 1.

Summary of BRGMON-1 Management Questions and Interim (Year 8 – 2019) Status

Primary Objectives	**Management Questions**	**Year 8 (2019) Results To-Date**
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. 2019 was characterized by a return 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. • Estimated site-specific discharge estimates in 2019 highlighted that flow conditions among sites in reaches 3 and 4 were fairly similar, but that they differed greatly 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 2019), 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. • At the 2019 flows, Reach 3 was enriched with nutrients from tributary inflows or groundwater or both. Extreme phosphorus deficiency for algal growth was evident in Reaches 4 and 2, which did not occur during the Trial 2 years. Low SRP concentrations in Reach 4 were due to little phosphorus load from Carpenter Reservoir in 2019. Low SRP concentration in Reach 2 was due to dilution of nutrients in the Lower Bridge River by the Yalakom River that was extremely nutrient deficient. Enrichment from tributaries contributed to relatively high nutrient concentrations in Reach 1. • 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 Trial 3 (high flow) years and in 2019, 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). Water temperatures in February 2019 were colder than average which may have extended incubation timing for chinook relative to previous years with Trial 2 flows (2011-2015). There is a gradient of temperature associated with distance from the dam (e.g., Reach 1 temperatures were ~5°C cooler than Reach 4 in fall 2019).
	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?	• Flows during Trials 1 and 2 produced what might be called optimum conditions for the benthic communities. The average 73% decline in invertebrate density and low diversity associated with Trial 3 flows showed that physical conditions associated with high peak flow, including scour and bed movement, did not favour the benthic communities. Given that benthos found in the Lower Bridge River includes common fish food organisms, the Trial 3 flows caused a decline in the food available to fish at the time of measurement in the fall months. • Despite similar flows in 2019 to those during Trial 2, the 2019 values of metrics of benthic production did not recover from low values in 2016-2018 in Reaches 4 and 2. They did recover in Reach 3. • Updated analysis of relationships between physical and chemical habitat conditions among Reaches 4 to 1 showed that benthic productivity is driven by interactions between nutrient concentration and temperature at flows in 2019 and during Trial 2 years.

Primary Objectives	Management Questions	Year 8 (2019) Results To-Date
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.		• An overall conclusion to date is that scouring effects of high flow during Trial 3 reduced food for fish but at lower controlled flows typical during Trial 2, nutrient concentrations and temperature exerted control over food for fish.
	How do changes in physical conditions and trophic productivity resulting from flow changes together influence the recruitment of fish populations in Lower Bridge River?	• Similar to the results for benthic invertebrates, 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 the previous flow treatment, the high flows in 2016, 2017 and 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 the Trial 2 hydrograph in 2019 provided for modest recovery of mykiss and coho fry in Reach 3; however the abundance of mykiss parr (which recruited under high flows) and chinook fry (which are likely affected by temperatures during incubation for all flow trials) were unchanged and recovery of all species in reaches 2 and 4 was negligible. Abundances in Reach 1 were lowest of all the reaches for each species and age class on a lineal (# of fish per km) basis. Reach 1 is the length of reaches 2, 3 and 4 combined and produced 1/3 of the fish from those reaches. • Juvenile salmonid biomass trends mirrored those for abundance. • Interestingly, the alignment of reach-based recovery results for juvenile salmonids in 2019 with the benthic invertebrate and nutrient source information seemed to suggest that potential food source dynamics and reach-based productivity metrics may have been driving influences on juvenile salmonid abundance patterns in 2019. However, the abundance and condition factor data also suggest that food supply may have been limiting in each reach in 2019, contributing to the negligible recovery in reaches 2 and 4 and density-dependent factors limiting fish survival in Reach 3. • 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 is likely that production was not stock-limited in most study years; however, more data at low escapements (and correspondingly low recruitment) are required to reduce uncertainty in the initial slope. The 2019 data point for coho and chinook were compared to the curves and points for Trials 2 and 3 but were not incorporated into the existing curves since 2019 represents the start of a new flow treatment (post-high flows). As such the existing curves remained unchanged. • Hatchery-reared chinook fry (n= 3,189) were released into the LBR 3 weeks prior to stock assessment sampling in 2019, and 14 were subsequently recaptured. The most likely survival estimate for these fish was 40%, resulting in an addition of 1,280 fry (or 11%) to the naturally produced chinook population.
	What is the appropriate 'shape' of the descending limb of the 6 cms hydrograph, particularly from 15 cms to 3 cms?	• No new insights from 2019 for ramping strategy between 15 and 3 $m^3{\cdot}s^{-1}$ beyond what has already been documented in past reports and the fish stranding protocol. • Modified Operations (2016 – 2019) results did affirm 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.

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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).

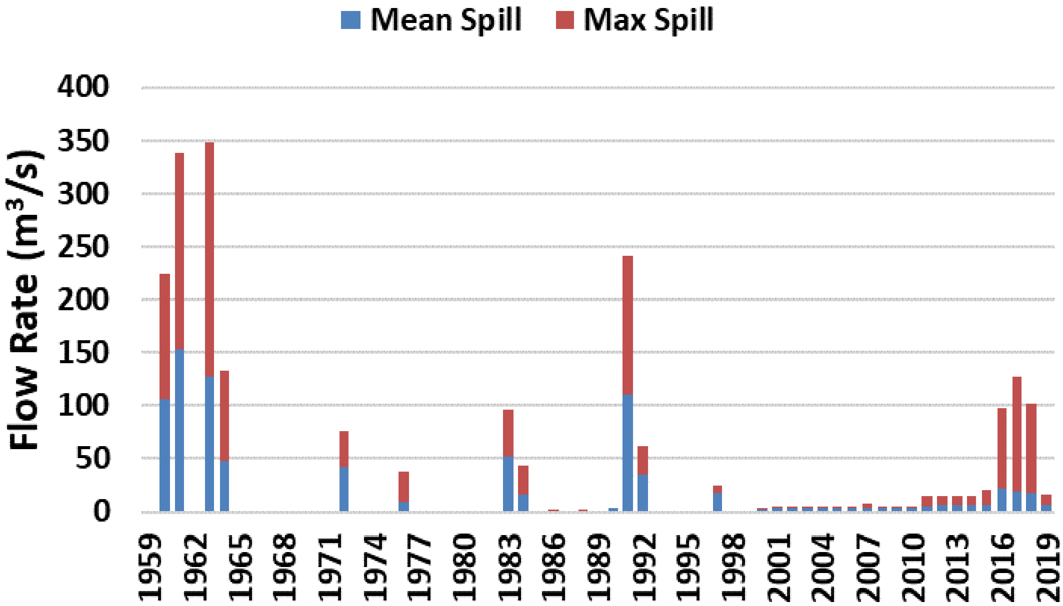


Figure 1.1 Frequency of spill and flow release events from Terzaghi Dam into the Lower Bridge River following impoundment in 1960.

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 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 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 that were designed by an interagency technical working group (Figure 1.2). 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 2012; 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.

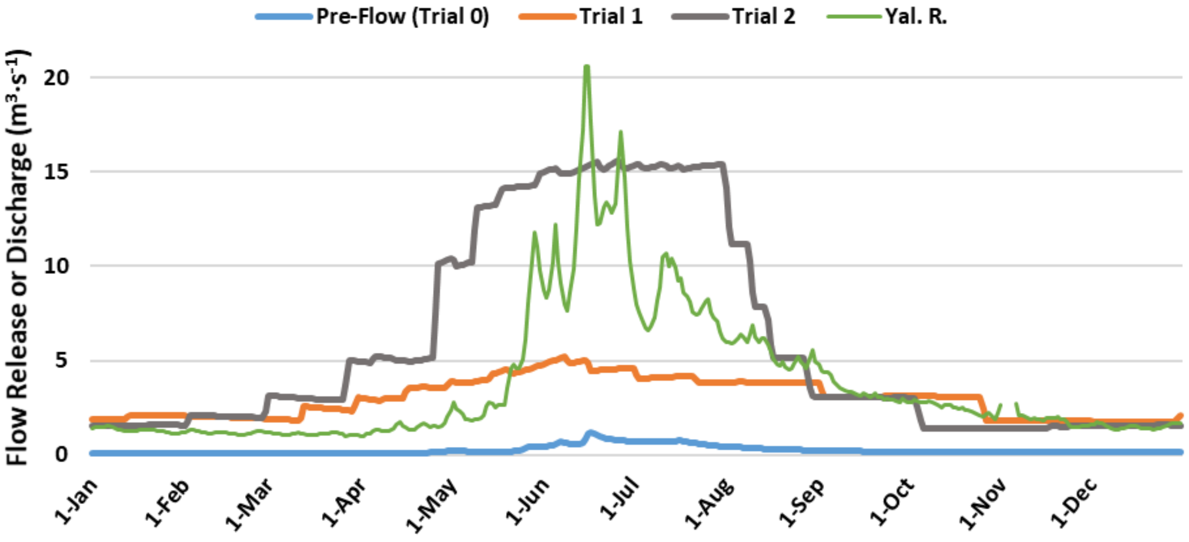


Figure 1.2 **Mean daily releases from Terzaghi Dam for Trial 1 and Trial 2 during the flow experiment. Typical hydrograph shapes during the Pre-flow period and for the unregulated Yalakom River discharges are included for reference.**

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 the narrow valley 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 system. 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 ideally sought to 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 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 may need 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 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. These high flow years are collectively referred to as “Trial 3” in this report.

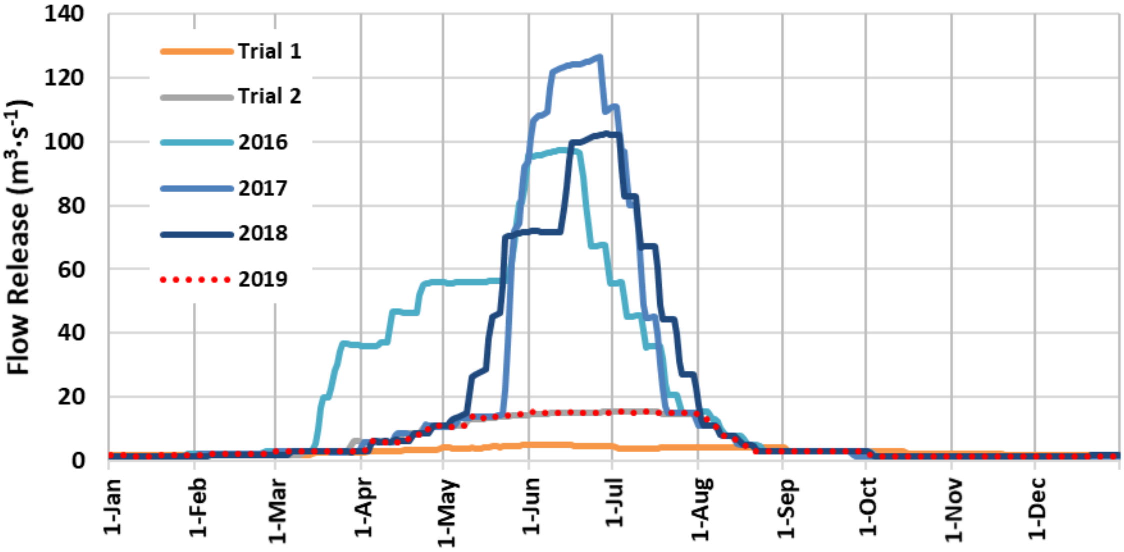


Figure 1.3 Terzaghi Dam flow release hydrograph shapes for Modified Operations years (2016–2019). Mean daily releases for the Trial 1 and Trial 2 hydrographs are shown for reference. Following three consecutive years of high flows, 2019 was a return to the Trial 2 hydrograph.

In 2019, reduced snowpack and inflow volumes allowed for a return to the Trial 2 hydrograph in the Lower Bridge River (mean annual flow = 6 $m^3 \cdot s^{-1}$; peak flow = 15.7 $m^3 \cdot s^{-1}$). As in the other Trial 2 years (2011 to 2015), flows ramped up from the winter lows during April and May (~3 to 15 $m^3 \cdot s^{-1}$), peaked across the months of June and July (15 $m^3 \cdot s^{-1}$), and then were ramped back down to 3 $m^3 \cdot s^{-1}$ between 1 and 19 August, and then to 1.5 $m^3 \cdot s^{-1}$ on 1-2 October. Flows above the Trial 2 peak were not necessary (2019 high flow duration = 0 days). Despite identical flow delivery characteristics to the Trial 2 releases, 2019 was considered a stand-alone year in the analyses and results provided in this report since it occurred within the context of the Modified Operations period and followed 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. Trial 3 produced a pronounced bell-shaped hydrograph with steep ascending and descending limbs and highest peak release among trials. 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, 2019 flow releases were characterized by the line representing the Trial 2 mean.

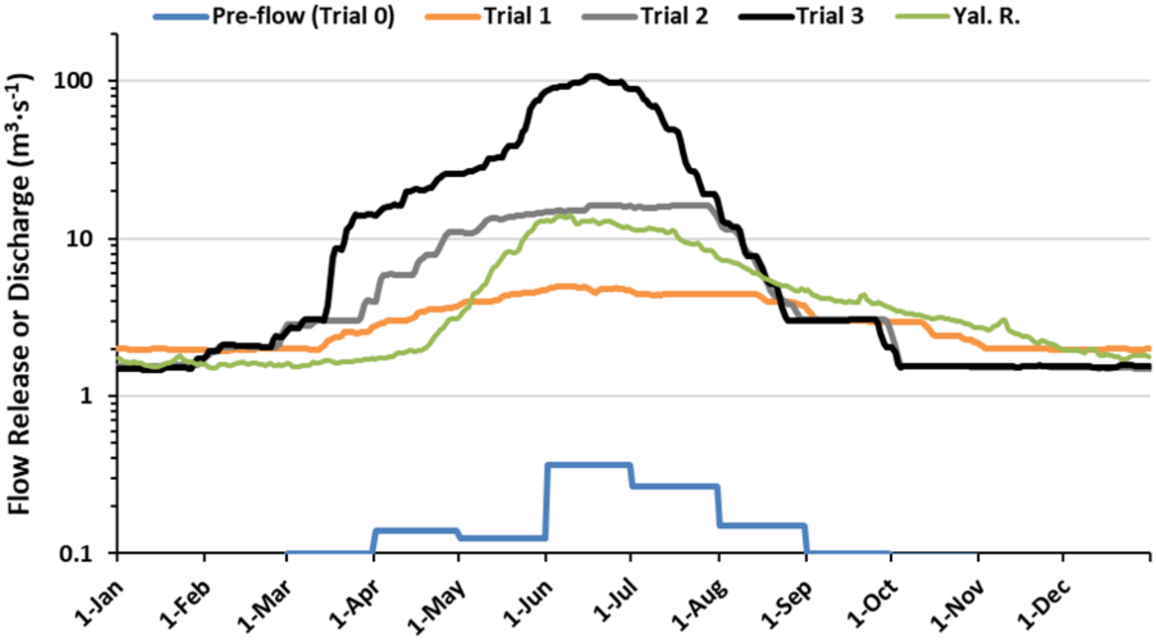


Figure 1.4 **Mean daily flow release from the Terzaghi Dam among all years in each flow Trial. Mean daily flow among all years (1996 – 2019) in the Yalakom River is shown for reference. Note the log scale on the Y axis.**

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 4.2–5.2 $m^3{\cdot}s^{-1}$ 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 1–2 $m^3{\cdot}s^{-1}$ in both the water release to the Bridge River and in the Yalakom River. Average peak flow in the Yalakom River was 22–25 $m^3{\cdot}s^{-1}$ among all blocks of Trial years, which was about 50% greater than the peak flow release during Trial 2 in the Bridge River.

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
Lower Bridge	1 (2001–2010) (n=10)	3.1 ± 0.2	1.8 ± 0.1	5.4 ± 1.1
Lower Bridge	2 (2012–2015, 2019) (n=5)	6.1 ± 0.3	1.1 ± 0.6	16.6 ± 2.3
Lower Bridge	3 (2016–2018) (n=3)	19.5 ± 2.1	1.4 ± 0.1	108.7 ± 15.7
Yalakom	0 (1996–1999) (n=4)	5.1 ± 1.2	1.2 ± 0.1	25.1 ± 12.7
Yalakom	1 (2001–2010) (n=10)	4.2 ± 0.9	1.2 ± 0.2	22.5 ± 10.7
Yalakom	2 (2012–2015, 2019) (n=5)	4.6 ± 0.5	0.7 ± 0.4	23.3 ± 5.4
Yalakom	3 (2016–2018) (n=3)	5.2 ± 0.4	0.9 ± 0.3	25.1 ± 1.8

[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 are expected to cause increased entrainment at Terzaghi Dam, limit juvenile salmonid rearing habitat area, cause erosion and sediment deposition throughout the river, and increase the 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, Management Questions and Study Hypotheses

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.

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.

While these management questions were originally intended to improve understanding of LBR aquatic productivity under the Trial 1 and Trial 2 hydrographs, the management questions are still considered relevant for understanding the effects of the high discharges from Terzaghi Dam (i.e., as an additional flow "trial"; a.k.a. Trial 3) in the context of the flow experiment.

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 and Trial 3 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).

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. However, due to manageable inflow volumes in 2019, releases above the Trial 2 peak of ~15 $m^3 \cdot s^{-1}$ from Terzaghi Dam were not required during that year. Therefore, the additional monitoring activities prescribed for Modified Operations years with high flows (i.e., High Flow Monitoring; Water Quality, Erosion and Entrainment Monitoring; High Flow Fish Salvage and Stranding Risk Assessment; Juvenile Salmonid Habitat Availability and Displacement; and Substrate Mobilization, Deposition and Composition Monitoring) were not implemented in 2019. As such, the management questions pertaining to Modified Operations (high flows) are not included in this report which focusses on the core (WUP) monitoring activities only.

We expected the change from high peak summertime flows in 2016 – 2018 to the lower peak flows in 2019, that were the same as those during Trial 2, would reverse the decline in benthos abundance and diversity found in Trial 3 compared to the earlier trials. This hypothesis is supported by findings from the 2018 analysis showing that flow, particularly peak flow, was the main factor driving variation among benthic invertebrate assemblages in the LBR (Sneep et al. 2019). We similarly hypothesize that the return to lower flows in 2019 would also increase juvenile salmonid abundance to the range of values documented for the previous Trial 2 years (2011 to 2015). In this report we test the hypothesis that the similarity of 2019 flows to those in Trial 2 will cause benthic assemblages and juvenile salmonid abundance in 2019 to be similar to those found during Trial 2.

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 for the first time (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

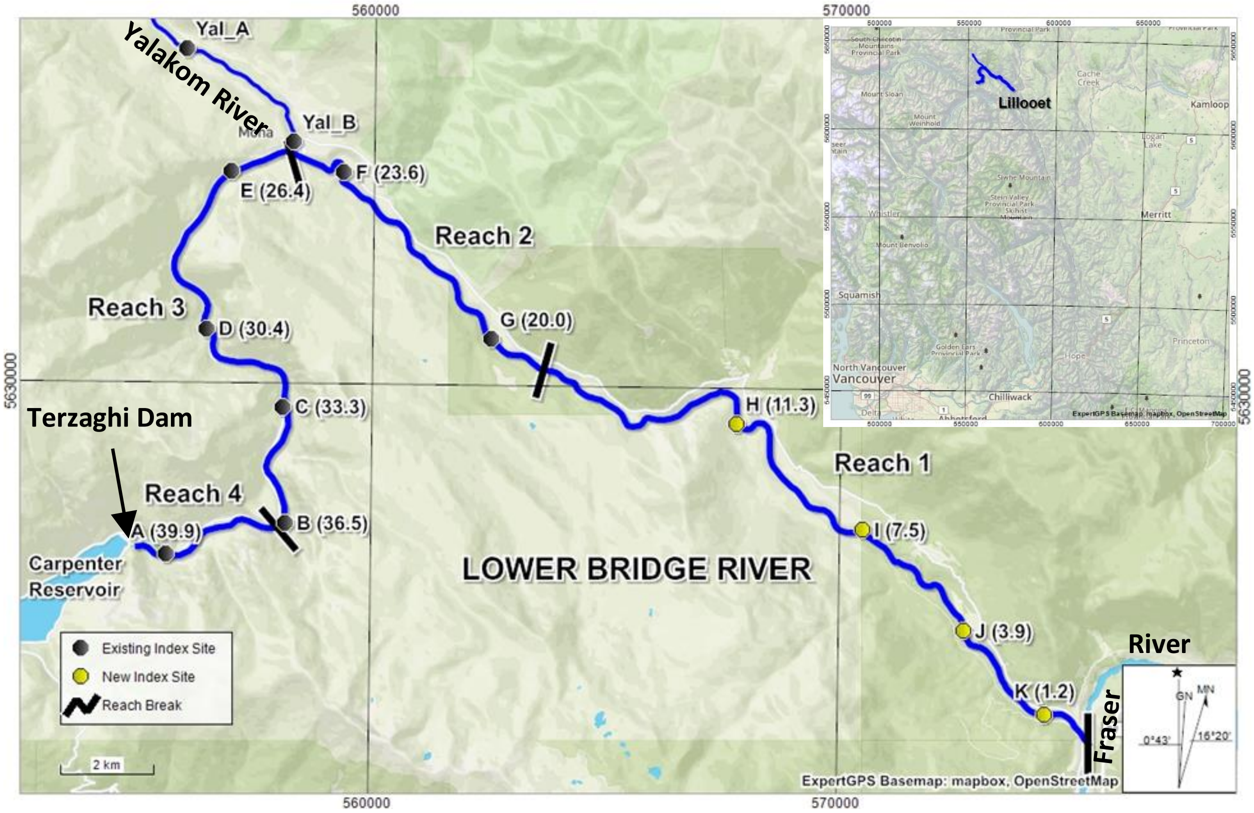


Figure 1.5 **The Lower Bridge River downstream of Terzaghi Dam near Lillooet, British Columbia. Reaches are labelled 4 (upstream) through 1 (downstream). Index sampling sites are labelled as distances upstream of the Fraser River and correspond to the following letters in some of the figures below: 39.9 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, and the new Reach 1 sites (established in 2019) are shown as yellow dots. 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). 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 $m^3 \cdot s^{-1}$). Reach 1 (length = ~19.0 km) receives inflow from some small tributaries (e.g., Applesprings, Moon and Ama Creeks) but, relative to the differences among trials in the other reaches, discharge rates in

Reach 1 were expected to be fairly similar to Reach 2. Other smaller tributaries to the Lower Bridge River include: Mission Creek, Yankee Creek, Russell Springs, Hell Creek, and Michelmoon Creek in Reach 3; and Antoine Creek, and Camoo Creek in Reach 2.

1.6. Study Period

Data collection in 2019 was conducted between April and December 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 2019; however, comparisons with previous years and flow trials are included where relevant and available.

Table 1.3 Summary of data to be included in BRGMON-1 analysis and reporting for monitoring year 2019. Components that have prior years of data are noted.

Task	Components	2019 Period	Prior Years of Data[1]
Physical Parameter Monitoring	Water temperature; river stage; discharge	Year-round	1996 to 2018
	Turbidity	1 Apr to 8 Oct 14 Nov to 4 Dec [2]	NA
Water Chemistry	Nutrients; alkalinity; pH	30 Sep & 9 Dec	1996 to 2018
Primary & Secondary Productivity	Periphyton accrual; benthic invertebrate diversity & abundance	24 Sep to 28 Nov (LBR)	1996 to 2018
		25 Sep to 27 Nov (Yalakom)	2018
Juvenile Salmonid Abundance	Annual standing stock assessment	29 Aug to 18 Sep	1996 to 2018
	Juvenile fish density in off-channel habitats	18 & 22 Oct	2018
WUP Ramp Down Monitoring	Stage monitoring; fish salvage	1-19 Aug & 1-2 Oct	2011 to 2018
High Flow Monitoring	Kokanee entrainment; water quality sampling; sediment erosion & deposition; fish stranding site reconnaissance	NA	2016 to 2018
Juvenile Salmonid Habitat Availability & Displacement	Single-pass, open site electrofishing at pre-selected low- and high-quality rearing sites	NA	2018
High Flow Ramp Down & Stranding Risk Assessment	Stage monitoring; fish salvage at flows >15 m^3/s	NA	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 2019 results and where this could be supported by the project budget.

[2] Note: While turbidity loggers were deployed for these periods 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.

Sampling methods were the same during all trials and in 2019. Tasks included: 1) continuous recording of flow release, river stage, temperature and turbidity; 2) assessment of water chemistry parameters, periphyton accrual, and aquatic invertebrate abundance and diversity during fall; 3) a fall standing stock assessment to estimate the relative abundance and distribution of juvenile salmonids in the study area; and 4) flow ramp down monitoring and fish salvage. As in 2018, activity 3) also included assessment of fish densities in two off-channel habitats during the stock assessment period.

Tasks 1) and 2) were conducted at eleven index sites in the LBR and two sites in the Yalakom River in 2019. They included the usual seven 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 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. 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. 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 3; and at 61 sites across reaches 1 to 4 in 2019.

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 High flow years (Trial 3), 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
Trial 3 (Mod. Ops.)	2016 – 2018	2, 3, 4	Fall	No target [a]	19.5 ± 2.1
	2018	Yalakom		n/a	n/a
	2019	1,2,3,4	Fall	No target [a]	6.0
		Yalakom		n/a	n/a

[a] Trial 3 flows 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 Trial 3 year.

Field data collection in 2019 was conducted by members of Coldstream Ecology Ltd., Xwísten and St'at'imc Eco-Resources. The field studies project manager Alyson McHugh and members of her team 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 (Trial 3) year in 2016 (McHugh and Soverel 2013, 2014, 2015, 2017; McHugh et al. 2017).

There was an important issue to note with the turbidity data from the loggers that were newly installed in the river in 2019 that precluded inclusion or analysis of these data for this report. After turbidity loggers were downloaded at the end of 2019, the turbidity sensors were found to have "blanked-out" for several hours during the daytime throughout the monitoring period. Correspondence with the manufacturer showed the likely cause was exposure to sunlight that can cause saturation of the photodiode amplifier, leading to loss of signal. Given that the instruments were installed in a river near the surface, there can be influence of high light. A solution from Seapoint, the manufacturer, was to remove exposure of the sensor to high light, which can be done by using a different protective housing. Furthermore, voltage output was found to clip below a lower threshold, resulting in a flat horizontal line at low values. A fix was developed by Seapoint to correct the clipping but this action did not allow 2019 data below the low threshold to be recovered. Given the data blanks during high daylight periods and data clipping at low values, the 2019 turbidity data were deemed unusable for analysis. Corrections were made in 2020 including a firmware update and construction of sensor shades for use of the turbidity sensors in 2020. Details of this explanation are found in emails between Chris Perrin (Limnotek) and Tomi Banjo (Seapoint) during May 12-13, 2020.

2.2. Physical Habitat Parameters

2.2.1. Discharge

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 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 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);

$Q_{i\text{-}1,\,d}$ = 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.

Three flow metrics were calculated from these site-specific flow data for analysis of benthic community response to flow Trials (Section 2.8.1). ***Incubation flow*** was the average value of mean daily site-specific flow during the period of periphyton and benthic invertebrate sampler incubations (Sections 2.3 and 2.4). ***Freshet flow*** was the mean site-specific flow during March 1 - August 31 preceding the fall periphyton and benthic invertebrate sampling. This metric captured the full spring and summer flow releases between the start of ramping up of flow releases and the end of ramping down each year. ***Peak flow*** was the peak site-specific flow during March 1 – August 31 preceding the fall periphyton and benthic invertebrate sampling. This metric captured the peak flow at each site during the spring and summer flow release.

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 was 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. River Stage

The relative stage of the river was continuously monitored and recorded at four stations (km 39.9, 26.0, 23.6, and 20.0) using water level data loggers manufactured by Onset Computer Corporation (Model: U20-001-01). The stage data was logged hourly throughout the year, and the loggers were checked and maintained every few months when they were accessible (i.e., not under high flows or mid-winter conditions). BC Hydro maintained 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. Hourly river stage data for this site was provided by BC Hydro Generation Operations.

2.2.4. Water Temperature

Water temperature was recorded hourly throughout 2019 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 observations made during streamwalk surveys conducted as part of BRGMON-3 (White et al. 2018).

2.2.5. Benthic habitat

Several habitat attributes were measured to describe the physical and chemical conditions to which the benthic biota were exposed among Trials and Reaches.

Among available flow metrics (flow release from the Terzaghi Dam, site – specific flows during different durations), **peak flow** was selected as the flow indicator. It was the highest site-specific flow (Section 2.2.1) during the spring and summer (March 1 – August 31) preceding the periphyton and benthos sampling. Peak flow was a precursor of flow conditions leading up to the time of sampling and thus set the physical conditions that would have affected biotic assemblages in the fall from flow events in the spring and summer. Flows at the same time of sampling were not considered an indicator of Trial because they were always at low base flow and thus not representative of the flow releases from the dam that defined Trial (Figure 1.4). Those releases were during the spring and summer. Water **depth and velocity** at each of the periphyton and benthic invertebrate samplers was measured weekly during the periods of sampler incubation (Sections 2.3 and 2.4) using a top-set wading rod and velocity meter (Swoffer Instruments, Inc.). **Water temperature** at each site was continuously logged during the periods of periphyton and benthic invertebrate sampling as described in Section 2.2.4. Water samples for analysis of **nutrient concentrations** were collected using standard grab methods at the start and end of each periphyton and benthic invertebrate sampling time series (Sections 2.3 and 2.4) at each station. All samples were kept cold 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, 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 was used to show 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 may be 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.

Particle size distribution was measured in 2019 by conducting pebble counts to measure the composition of the stone matrix in ambient habitat near the benthos samplers (Section 2.4). A 100 particle Wolman Pebble Count (Wolman 1954) was completed at each site, where the intermediate diameter of 100 randomly selected particles within the stream reach was measured using calipers. A crew member entered the stream in the vicinity of the basket samplers. Without looking, he or she picked up a stone and measured its intermediate diameter using calipers, moved two or three steps in any direction and repeated the measurement on another stone. This process was repeated to measure 100 stones. Measurements were done at each site except at sites H and I where safety risk prevented access on the day of sampling. From the sizing of the 100 stones, the 50th percentile called **D50** was calculated as the median particle size among the 100 measurements (Eaton et al. 2019).

2.3. Periphyton Biomass and Composition

Periphyton was sampled during the fall (October – December) in 2019 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 addition of four stations in Reach 1 was new for 2019. 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 2019 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 2.1). 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 2.1 Image of an installed periphyton plate.

Periphyton biomass was sampled weekly from each of three replicate plates at each location for 63 days between 25 September and 27 November 2019. This incubation period was the same as 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.

On the final periphyton sampling day (November 21, 2019), one additional core was removed from each plate and preserved in Lugol's solution, for later cell counts and biovolume by taxon. 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.

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 2.2), with 2 cm openings that was installed in the river for 63 days. 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. To maintain sampling consistency, the same substrates were used in each basket from year-to-year throughout this monitoring program, unless they needed to be supplemented due to spillage or loss during the sampling period. The sampling methods and equipment have remained consistent among all monitoring years.

At the start of each colonization period, the baskets with contained stones (which had been cleaned and dried since the previous sampling event) were placed among the natural river substrates. 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.

At the end of the 63-day incubation period, 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. All of the material scrubbed from the rocks 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.

Photo 2.2 Basket sampler before installation in the Lower Bridge River.

In the laboratory, formalin was removed from the samples before processing by washing with water through a 250µm 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 the remaining samples were washed through a 2 mm sieve to separate the large (Macro) substrate and specimens from the small (micro) specimens and substrate. All specimens were removed from the macro portions and stored in 70% ethanol for identification. The micro portions were subsampled using the following procedure:

a) 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.

b) Suspended micro portions were each homogenized with 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.

c) A random selection of three samples (10%) were sorted twice to ensure picking efficiency was consistently maintained at 95%.

d) 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 picked specimens from both macro portions and the subsampled micro portions were physically sorted into separate vials, including: 1) order level taxonomy for aquatic insects, 2) 'Other taxa' group (including terrestrial insects, non-insect aquatic invertebrates, and vertebrates). Specimens remain preserved with 70% ethanol and stored in labelled vials.

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 and Biomass

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. In 2019, 12 new sites were established in Reach 1 extending the spatial scope of monitoring to the confluence with the Fraser River (Figure 2.1 and Appendix A).

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 was 97 m^2 (range: 20 to 273 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.

Table 2.2 Years 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]		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	Trial 3 – Mod. Ops. ($>18\ m^3 \cdot s^{-1}$)	2-4	48	1 – 21 Sep
2017			49	5 – 20 Sep
2018			49	4 – 15 Sep
2019	Mod. Ops. ($6\ m^3 \cdot s^{-1}$)	1-4	61	29 Aug – 18 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.

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 5.4 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 2019; see September period on Figure 1.3).

Figure 2.1 **Distribution of Juvenile Stock Assessment Sites in the Lower Bridge River study area. Mainstem sites in Reach 1 were new in 2019.**

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).

Enhanced Off-channel Sites – Juvenile Salmonid Habitat Availability

Following from pilot efforts during high flow releases in 2018, fish sampling was undertaken in a couple of enhanced off-channel sites in June and August to characterize fish use of these habitats during the peak period and descending limb of the LBR hydrograph in 2019. 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. However, results from this sampling component were not included in the report for Year 8 (2019). Data collection in non-high flow years was intended to provide context and background information against which the high flow results can be compared in future reports. 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).

2019 surveys were conducted on 13 and 14 June, and 2 and 7 August. The first survey targeted peak flows for the Trial 2 hydrograph (i.e., 15 $m^3 \cdot s^{-1}$ on both June dates), and the second survey targeted the descending limb when flows were being ramped down from the peak (i.e., 12.9 $m^3 \cdot s^{-1}$ on 2 August and 11.0 $m^3 \cdot s^{-1}$ on 7 August). 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. Electrofishing effort (seconds) and the number of crew members carrying out sampling was recorded. The locations of the upper and lower extent of each site were recorded with a GPS, the length of shoreline, and the general characteristics of the site (habitat type, dominant/sub-dominant substrate, water visibility) were also recorded.

Enhanced Off-channel Sites – Juvenile Salmonid Stock Assessment

Enhanced side channel sites were also sampled for juvenile salmonids in fall of 2018 and 2019. During each year, a single site in riffle habitat, and a single site in pool habitat was sampled at both the Applesprings enhanced side channel located in Reach 1, and the enhanced Bluenose side channel in Reach 4 (refer to Appendix A for maps and UTM coordinates). 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 above); however, sample timing was similar to the mainstem depletion sampling in 2018 (i.e., 15 and 28

Sep), but was approx. a month later in 2019 (i.e., 18 and 22 Oct). A multi-pass electrofishing depletion approach was used to estimate abundance, density, and biomass. Estimates of density and biomass were compared to averages from mainstem sites during 2018 and 2019 and the average across trials 1 and 2. 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.

Hatchery Chinook Releases

Hatchery-reared chinook fry were released into the Lower Bridge River at the plunge pool immediately below Terzaghi Dam on 20 August, 2019. This date was approximately 2 weeks prior to the start of the annual fall stock assessment sampling (29 Aug to 18 Sep, 2019). The fish were from broodstock that had been collected at the fish fence installed at Rkm 26.1 (just upstream of the BRGMON-3 fish counter), and then reared at the DFO Tenderfoot Creek Hatchery near Merritt, BC. Approximately 3,189 chinook fry were released to the Lower Bridge River in 2019, and all were adipose-clipped for visual identification and also coded wire tagged (Dorian Turner, BC Hydro, pers. comm.). Crews that completed the juvenile stock assessment sampling were informed of the hatchery fish release and kept track of hatchery-marked fry separate from wild chinook fry on the datasheets. Otherwise, hatchery fry were processed and released in the same manner as for all the other captured fish.

The 2019 marked hatchery fish capture data were used to estimate their survival rate from the release date to the 2019 stock assessment period, and to check against the estimated capture probability for naturally-produced (wild) chinook juveniles estimated from the stock assessment data.

2.6. Adult Escapement

Adult spawner count data for the Lower Bridge River (up to 2018) 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. 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 stream side 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 Salmon from the Yalakom confluence to Terzaghi Dam (Reach 3 and 4) since the start of monitoring (White et al. 2019).

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 lateral water visibility were also recorded (Ramos-Espinoza et al. 2018). 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 Reach 2 and spot counts in accessible sections of Reach 1 (White et al. 2019). 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: no Chinook and one coho salmon were observed during streamwalks in Reach 2 and spot checks in Reach 1 during 2018).

Streamwalk surveys in 2018 started on August 9 and continued until December 13 when fish activity ceased based on streamwalk and telemetry data. 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. 2019). 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. 2019).

2.7. WUP Ramp Down Monitoring and Fish Salvage

Flow ramping and fish salvage data were collected across the range of WUP Trial 2 flows (15 to 1.5 $m^3 \cdot s^{-1}$) in 2019. The methods described in this section are based on documentation provided by Coldstream Ecology Ltd. 2019 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. The data were available as hourly values.

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. 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) was the sole method 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) were measured for a subset of salvaged fish. 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 2019), 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 2019 were compared with the results from previous study years to better inform the risk of fish stranding across the flow ramp down range.

2.8. Data Analysis

2.8.1. Benthic Communities

The design for analysis of periphyton and benthic invertebrates included several response variables blocked among three categorical variables. The response variables were algal PB and the count per sample 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 were selected as indicators of benthic invertebrates for three reasons. They represented most individuals among the invertebrate assemblages found in previous analyses (Sneep et al. 2019). The EPT are known to be 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). The categorical variables were Trial, Reach, and Pink (binary coding for on and off years). Trial corresponded to each of the flow trials. Given clear physical separations between reaches (see Section 1.5), "Reach" blocked the response variables according to space. "Pink" blocked the response variables as a third factor based on evidence that nutrient addition from salmon spawning and carcass decomposition can enhance production of benthic biota (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. 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.

Stacked bar graphs were first drawn to descriptively show the taxonomic composition of periphyton and benthic invertebrates by Trial and Reach plus observations from 2019 and from the Yalakom River. The stacked segments of the bars were algal divisions or invertebrate orders to give a general description of composition over time and space.

We tested Trial, Reach, and Pink effects on each of several biotic metrics using three-way analysis of variance (ANOVA). Each observation for the ANOVA's was the mean value of a given metric (PB or invertebrate metrics) among all samplers and stations, by reach, in a given year. Years were replicates. This definition overcame pseudo-replication of samplers at a station and stations within a reach. These nested observations were statistically not independent of each other, which violated the assumption of independence in ANOVA. 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 Trial, Reach, and Pink was found, the results were interpreted graphically and the Tukey test was applied to examine significance of interacting pairs of each level of Trial (1 to 3) and Reach (2 to 4) and Pink (0 and 1). If no significant interaction of Trial and Reach was found, then each factor (Trial or Reach or Pink) was examined independently. If any one of the factors was found to have a significant effect on a biological metric ($p<0.05$), the Tukey test was run on the significant factor 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 the combination of Trial 0 and Reach 4 was missing from the layout and would not allow testing of Trial effects and Reach effects (an overall ANOVA could still be run but individual effects of Trial and Reach could not be examined). Either Trial 0 or Reach 4 had to be omitted to allow testing of Trial and Reach effects. 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 2019. Without observations from Trials 1 to 3, Reach 1 could not be included in the ANOVA's. Based on these criteria Trial had 3 levels (Trials 1 to 3), Reach had 3 levels (Reaches 2 to 4), and Pink had two levels (0 for off years and 1 for on years).

Given the lack of replicate years following the high flow years of Trial 3, we used a different approach to test the null hypothesis that a biological metric value in 2019 was the same as those during Trial 2 when flows were the same as in 2019. First, arithmetic means and 95% confidence intervals were calculated for each metric in Trial 2 (years were replicates). If the average metric value for a given Reach in 2019 was within the Trial 2 95% confidence intervals for that same reach, that value was not considered different from the Trial 2 values. If the average metric value for a given Reach in 2019 was outside of the Trial 2 95% confidence intervals for that same reach, that value was considered different from the Trial 2 values.

2.8.2. 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. In the analysis presented here we included the 2019 data as an additional Trial 2 year. The results as presented for the other flow treatments did not change.

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 3.1.1). 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 across two flow treatments for two reasons. First, this would involve a large number of comparisons. There are 6 potential flow treatment comparisons (Pre-flow to Trial 1, Pre-flow to Trial 2, Pre-flow to High flow period, Trial 1 to Trial 2, Trial 1 to High flow period, and Trial 2 to High flow period) for both reaches 2 and 3, and 3 flow comparisons for Reach 4. This results in 15 different flow treatment comparisons for each of four species-age classes for a total of 60 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 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 under the Trial 3 high flows 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.

2.8.3. 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 four 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 (Trial 3). 2019 is currently a lone datapoint representing the first year post-high flows under the Trial 2 hydrograph; as such, it was not included in any of the treatment averages.

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). Note: 2019 was not included in any averages since it was the first, and only, post-high flow data point.

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
Trial 3 – High Flow	>18 $m^3 \cdot s^{-1}$	2016-2018	2017-2018

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 (Trial 3) regime. As noted above, results from 2019 were not included under any of the previous trial flow averages since this was the first year of lower flows following 3 consecutive years of high flows (see Figure 1.1).

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 Trial 2 flows (i.e., 2019) were also not included in the treatment averages for mykiss parr.

2.8.4. 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 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. 2019). 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 2018 produced fry that were sampled in the fall of 2019). 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, and 4 represent how much the stock-recruitment curve shifts under the 3 and 6 $m^3 \cdot s^{-1}$ treatments, and under high flow conditions (2016 to 2018), 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; and that; 2) 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 Habitat Parameters

3.1.1. Discharge

In 2019, the flow release from Terzaghi Dam conformed to the Trial 2 hydrograph for the entire year. Minimum flows were 1.5 – 2.0 $m^3 \cdot s^{-1}$ from 1 January to 28 February, and 2 October to 31 December. Flows were ramped up from the winter lows in a series of steps between 1 March and 13 May and reached a stable peak of ~15 $m^3 \cdot s^{-1}$ from 14 May to 31 July. Flows were ramped down from 15 to 3 $m^3 \cdot s^{-1}$ in a series of eight steps between 1 and 19 August, and then down to the minimum release (i.e., 1.5 $m^3 \cdot s^{-1}$) in two steps on 1 and 2 October. Mean annual discharge from the dam in 2019 was 6.0 $m^3 \cdot s^{-1}$. 2019 was the first year of lower flows following three consecutive years of high flows (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 3.1). The maximum difference in flow volume between Site A (top of Reach 4) and Site E (bottom of Reach 3) was 4.7 $m^3 \cdot s^{-1}$ 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 early August. 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 29.0 $m^3 \cdot s^{-1}$ in 2019. The difference in flows across sites in reaches 2 and 1 was moderate (i.e., 2019 max. = 11.3 $m^3 \cdot s^{-1}$).

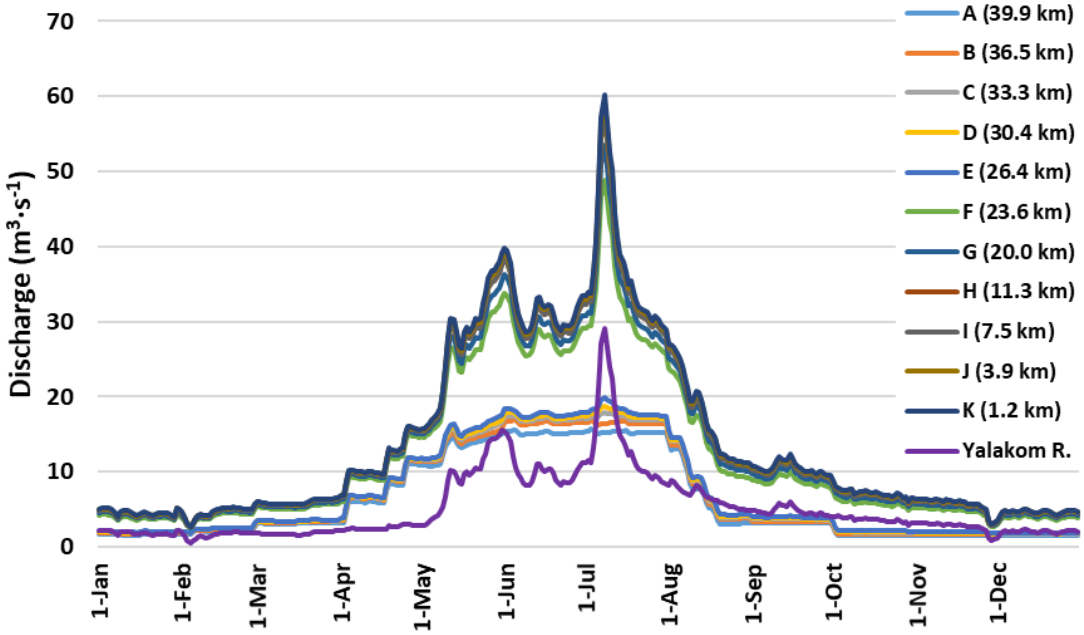


Figure 3.1 **Site-specific discharge estimates (shown as mean daily values) in the Lower Bridge River during 2019. Site A is in Reach 4, sites B to E are in Reach 3, sites F and G are in Reach 2, and sites H to K are in Reach 1. 2019 Yalakom River discharge is also shown.**

The Yalakom hydrograph in 2019 had two main peaks: an initial peak in late-May (up to 15.5 $m^3 \cdot s^{-1}$), followed by a higher peak in early July (up to 29.0 $m^3 \cdot s^{-1}$) (Figure 3.2). The second, higher peak represented a substantial volume of flow (i.e., 99th percentile) for that early July period based on the record from 1996 to 2019. The only other higher peak in the month of July occurred in 1999 (i.e., 42.1 $m^3 \cdot s^{-1}$ on 15 July). MAD for the Yalakom River in 2019 was 5.1 $m^3 \cdot s^{-1}$, which was in the 70th percentile for the 24-year monitoring period. The peak release flows from

Terzaghi Dam (i.e., from 14 May to 31 July 2019) overlapped the timing of the peak period on the Yalakom River (i.e., from approx. 3 May to July 2019), although the LBR flows started ramping up about a month earlier and were ramped down about three weeks later (Figure 3.1).

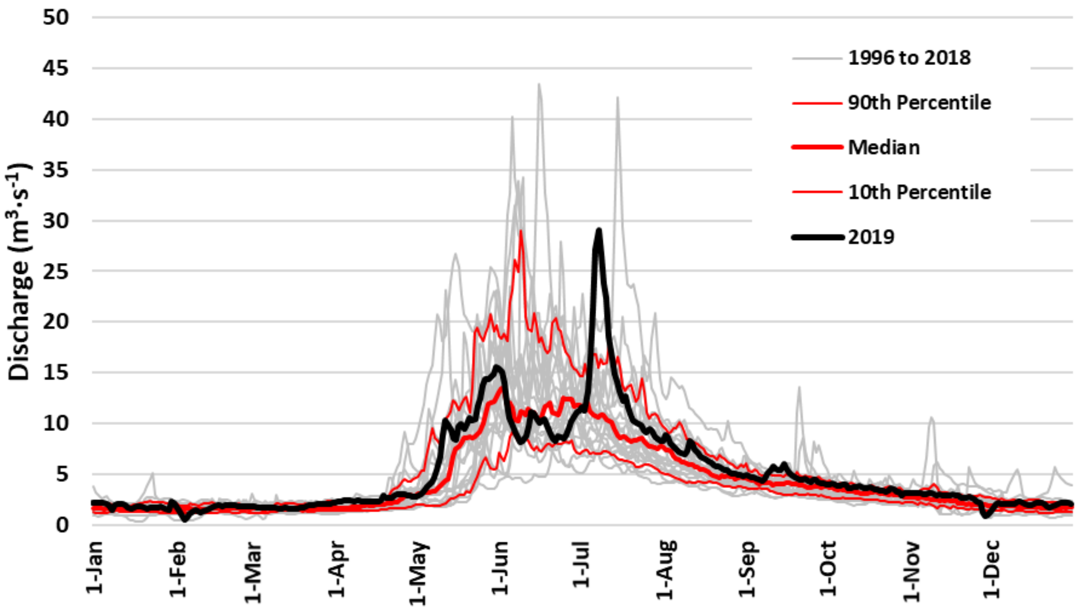


Figure 3.2 **Mean daily discharge in the Yalakom River for the period 1996 to 2019. The 10th and 90th percentiles are represented by the thin red lines (lower and upper, respectively) and the median is the bolded red line. The 2019 hydrograph is shown as the bolded black line.**

3.1.2. Wetted habitat area

Wetted habitat area was a logarithmic function of site-specific flow in all reaches (Figure 3.3). Without correction for reach length, the curves show wetted areas in Reach 3 > Reach 2 > Reach 4 (Figure 3.3 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 3.3 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 between flows was only comparable between reaches where flows were the same: 5 – 15 $m^3 \cdot s^{-1}$, (Figure 3.3). Application of the models in Figure 3.3 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.

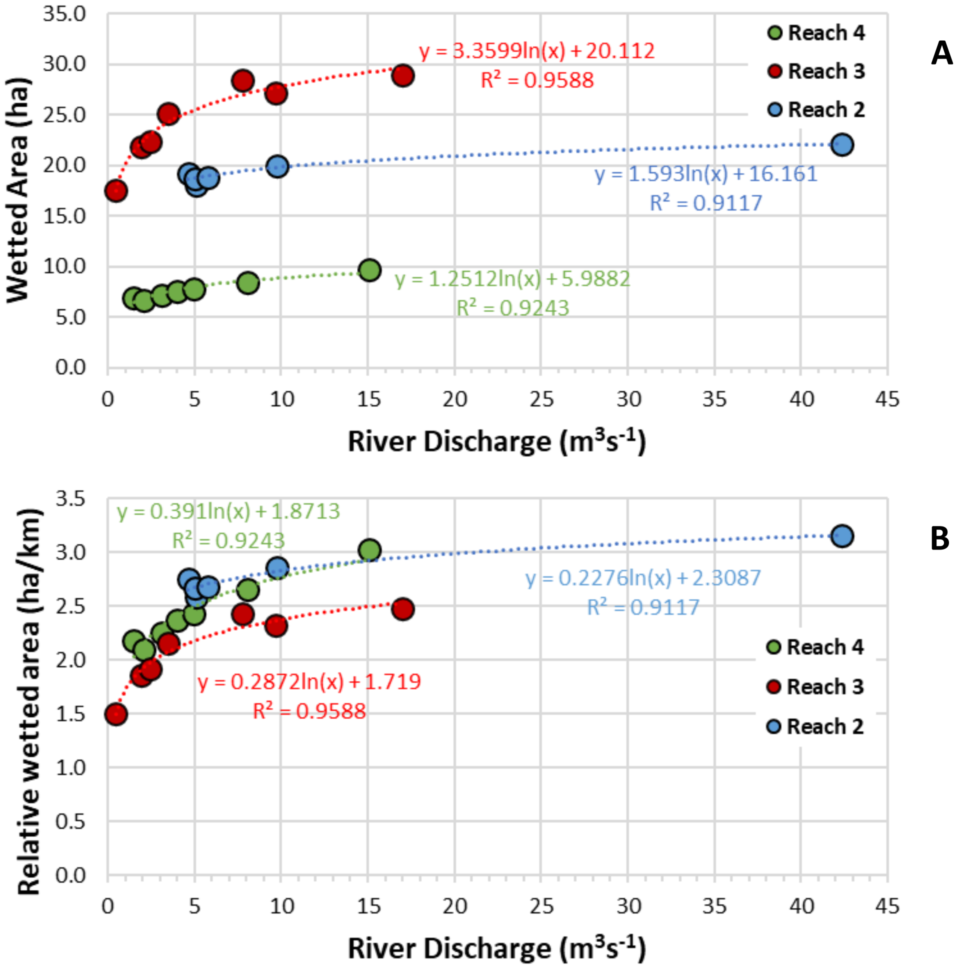


Figure 3.3 **Relationships between river discharge and wetted habitat area, by reach, (A) and between discharge and relative wetted area (wetted area divided by reach length) (B) in the lower Bridge River.**

3.1.3. 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 (Figure 3.4). 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, Trial 3 flows from 2016 to 2018 were also characterized by 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 release in 2019 followed the Trial 2 hydrograph and the temperature profiles by reach generally followed the patterns observed in 2011-2015. However, there were a few exceptions: temperatures in 2019 were generally 1–3°C cooler than the Trial 2 years in February; temperatures spiked higher in May; and temperatures were warmer than most Trial 2 years in spring and summer, which was most evident in reaches 3 and 4.

The cause of the cooler water temperatures in February and the spike in May are likely due to ambient temperature influence since these effects were apparent in all reaches and the Yalakom River (Appendix C). Mean monthly air temperatures were colder in February 2019 and warmer in May 2019 than they were in those months during the previous Trial 2 period (Table 3.1; Data provided by Environment Canada). The warmer temperatures in spring and summer may be due to upstream operations as the effect was also observed during the other modified operations years. Ambient temperatures in June to September 2019 were not notably warmer than this period during the previous flow trials. Water temperatures during the spawning and early incubation period for Chinook and pink salmon (i.e., September to December) were elevated relative to pre-flow temperature conditions, but on par with other years since the flow release was initiated (i.e., since 2000).

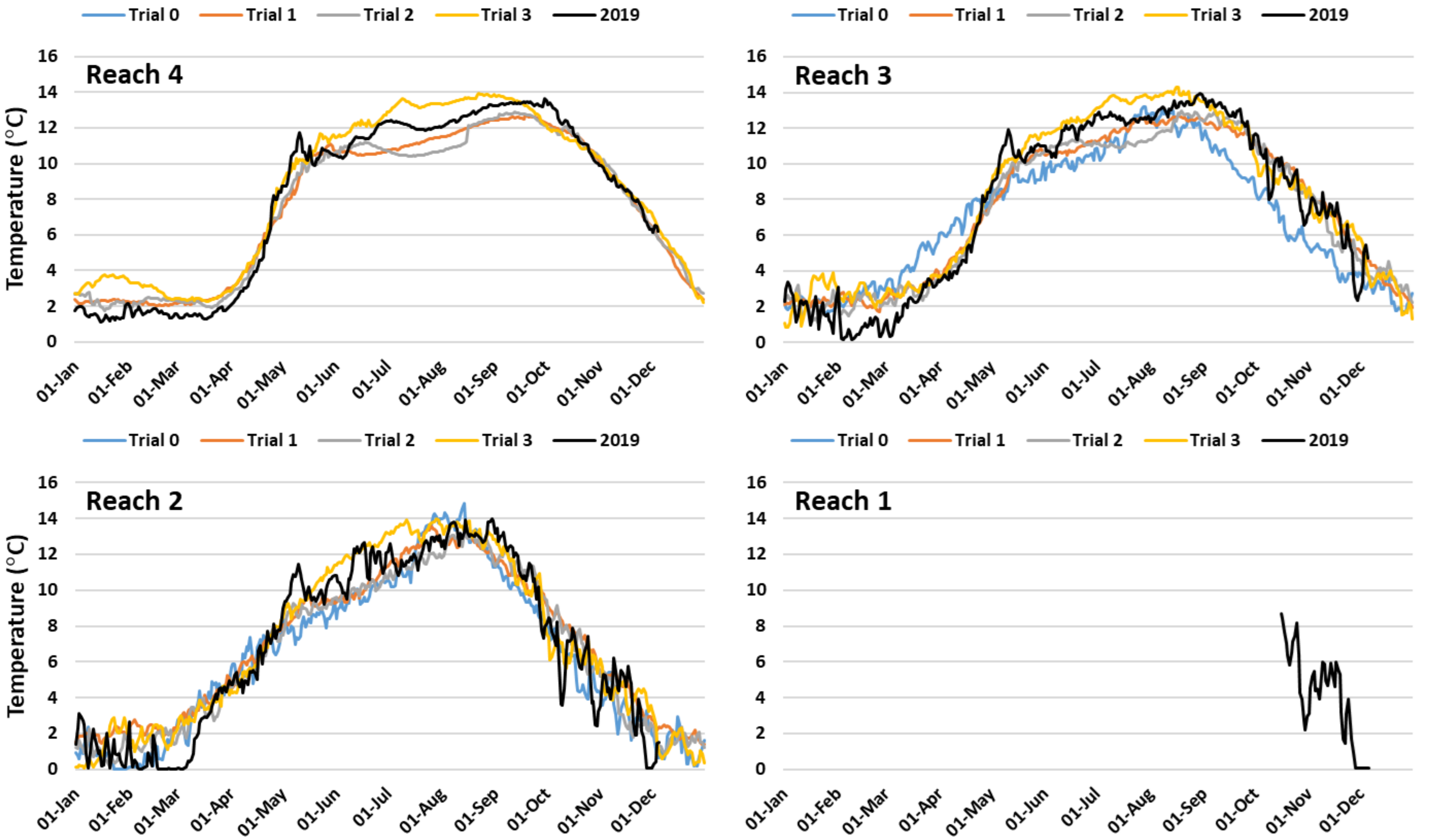


Figure 3.4 Mean daily water temperatures during Trial 0 (pre-flow), Trial 1 (3 $m^3 \cdot s^{-1}$), Trial 2 (6 $m^3 \cdot s^{-1}$), Trial 3 (high flows), and 2019 periods for Reach 4 (top left), Reach 3 (top right), Reach 2 (bottom left) and Reach 1 (bottom right). Note the temperature loggers in Reach 1 were installed on 16 October 2019.

Table 3.1 Mean monthly air temperatures for Lillooet, BC summarized by LBR flow trial/period (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)
Trial 3	-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)
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

The warmer spring/summer temperatures in 2019 were most evident in reaches 3 and 4 from June to September, and there were warmer temperatures in Reach 2 during May and June. The temperature profiles in reaches 3 and 4 are most influenced by the flow release from Terzaghi Dam (due to minimal tributary inflows); whereas, under Trial 1 and 2 flow conditions in the mainstem LBR, the temperatures in Reach 2 are strongly influenced by the Yalakom River (which also had warmer temperatures in May and June, but not in July and August 2019; Appendix C).

Higher spring and summer temperatures from the flow release were also observed under Trial 3 (to a greater extent than in 2019), and are likely caused by: 1) higher conveyance of water through Terzaghi Dam, which may affect draw from the various thermal layers in Carpenter Reservoir; 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 2019); 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.

Water temperatures during the early part of the salmon incubation period in fall (i.e., Sep to Dec) have been elevated during all flow trial years (2000 to 2019) by up to 4°C at the top of Reach 3 (relative to the pre-flow period - Trial 0). Differences among the flow trials during that seasonal period were small, as were differences among years within trials (refer to Reach 4 figure in Appendix C). 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 qualitative 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 3.5). 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. 2018).

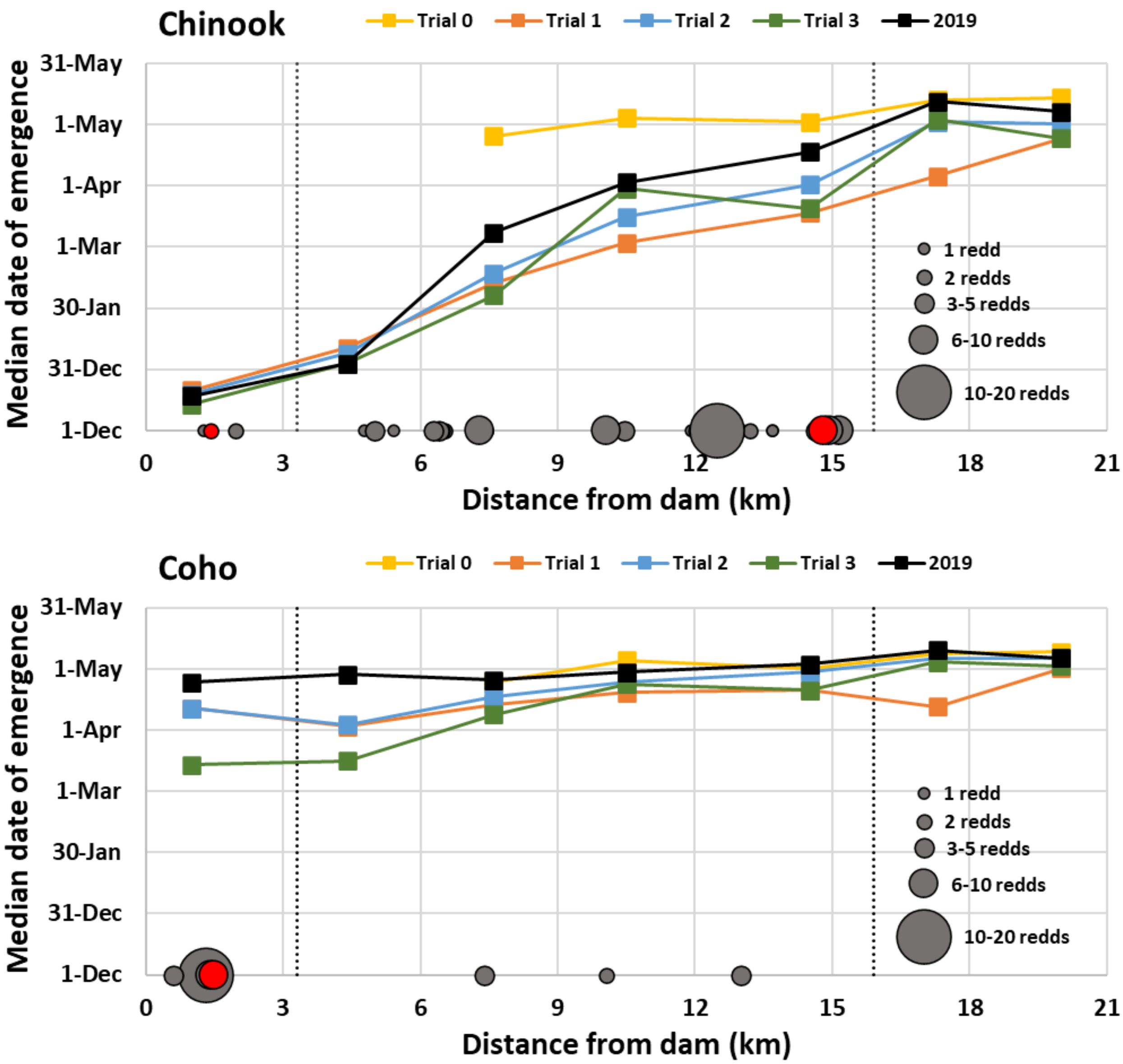


Figure 3.5 **Modelled median emergence dates for Chinook (top) and coho (bottom) salmon fry at varying distances below Terzaghi Dam based on observed daily mean temperatures for each flow treatment. The vertical dotted lines indicate the location of reach breaks. The locations of redds observed for each species during the available period (2014 - 2019) are represented by the dots along the x-axis on each plot (data provided by BRGMON-3). The colour of dot reflects the period (red = 2019; grey = 2014-2018) and the size of dot indicates the relative number of redds at each location.**

After the initiation of flow from the dam in Trial 1, predicted emergence time for chinook salmon advanced by 1-4 months with the greatest change occurring at sites nearest the dam. The 0.5 $m^3 \cdot s^{-1}$ reduction in October-January flows under Trial 2 and 3 compared to Trial 1 (Figure 1.2) resulted in no discernible change at the first three stations below the dam (i.e., sites A, B and C), but effected a slight delay in predicted emergence timing at the other stations (i.e., slightly closer to the Pre-flow timing). Predicted emergence timing in 2019 for sites C, D and E (i.e., 7.6, 10.5 and 14.5 km downstream of the dam, respectively) shifted 2-3 weeks later than the trials 1-3 averages because of the cold water temperatures in February. Incubating alevins at sites A and B (i.e., 1.0 and 4.4 km downstream of the dam) in 2019 were predicted to emerge in December and January so their emergence timing was not influenced by the cold spell in February.

Due to later spawn timing, the impact of the flow release on coho salmon emergence timing was much smaller than for Chinook, with emergence predicted to be advanced by less than 15 days at most locations (Figure 3.5). However, release temperatures during the Trial 3 high flow years tended to be warmer than the previous flow trials during the January and February period (Sneep et al. 2019). The reason for the warmer temperatures during this period were not clear, but in Reach 4 and the top of Reach 3 shifted the predicted median emergence dates for coho fry between 18 and 28 days earlier, from mid-April to mid-March. It was also unclear what effect, if any, this potential shift in emergence timing may have had for coho recruitment in Trial 3. In 2019, similar to the effect for chinook, the colder water temperatures in the LBR in February shifted coho emergence timing estimates slightly later than the previous trial averages and, in this case, very close to the Trial 0 (pre-flow release) estimates.

Chinook and coho spawners have utilized spawning areas in both reaches 3 and 4 during trials 2 and 3 (location-specific spawning information was not available for trials 0 or 1), but the distribution of redds among those reaches has been different for the two species (Table 3.2 and Figure 3.5). The total number of chinook redds observed from 2014 to 2019 was 131. Five percent of those (*n*= 7) were observed in Reach 4, and the remaining 95% (*n*= 124) were distributed across Reach 3. Based on these findings, and assuming equivalent survival among locations, approx. 5% 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 71% would have had a PME of mid- March to early April in the bottom portion of Reach 3.

In 2019, only eight redds were observed, although collection of these data was affected by the operation of a broodstock collection fence installed early in the spawning period at the bottom of Reach 3 (C. White, personal communication). Of the observed redds, 25% (*n*= 2) were observed in Reach 4 and 75% (*n*= 6) were observed at the bottom of Reach 3 near the fence. It is also important to note that, of the chinook spawners collected at the fence in 2019, none of them were determined to be of Bridge River origin (based on DNA analysis conducted by DFO at Pacific Biological Station lab). This was likely because of an increased incidence of straying in 2019 caused by a slide event on the Fraser River upstream of the Bridge River confluence that blocked salmon migration.

Observations for coho redd locations were available for 2018 and 2019 only, and the total number of coho redds observed in those years was 37 (2018 *n*= 31; 2019 *n*= 6). Unlike chinook, a much higher proportion of coho spawning was observed in Reach 4 (73%), with an associated PME timing of mid-March or late April for 2018 and 2019, respectively. The remaining 27% of redds were observed in Reach 3 with corresponding PME timing between 14 and 21 April, according to location (Table 3.2).

Table 3.2 Proportion of chinook and coho spawning, according to observed redd locations (2014 to 2019), by distance from dam and predicted median emergence timing in reaches 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	19-Dec	7	5% (5%)
	3	B (36.5)	4.4	8-Jan	17	13% (18%)
		C (33.3)	7.6	17-Feb	15	11% (29%)
		D (30.4)	10.5	17-Mar	54	41% (70%)
		E (26.4)	14.5	2-Apr	40	30% (100%)
Coho	4	A (39.9)	1.0	12-Apr	27	73% (73%)
	3	B (36.5)	4.4	3-Apr	0	0% (73%)
		C (33.3)	7.6	14-Apr	4	11% (84%)
		D (30.4)	10.5	20-Apr	2	5% (89%)
		E (26.4)	14.5	21-Apr	4	11% (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 and 2019 only, as these were the only years of redd count data available for this species.

3.1.4. Benthic habitat

Chemical attributes varied with trial and reach, described as follows and shown in Table 3.3.

Total alkalinity increased upstream to downstream in all trials. There was an accompanying increase in pH with rising alkalinity. At the pH's of 7.5 to 8.1 found among all 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. The increasing alkalinity upstream to downstream showed carbon was added to the Bridge River with distance from the Terzaghi Dam.

DIN concentrations (sum of NO_3-N and NH_4-N) followed the same pattern as total alkalinity by increasing upstream to downstream. An exception was during Trial 0 when Bridge River flow upstream of the Yalakom confluence was due to cumulative seepage. DIN was mainly comprised of NO_3-N in Trials 0 and 3 but during Trials 1 and 2, NH_4-N and NO_3-N concentrations were about the same, mainly in Reaches 3 and 2. These relatively high NH_4-N concentrations showed addition of reduced forms of nitrogen that would ultimately be oxidized to NO_3-N via nitrification given expected high oxygen saturation. Given the absence of disturbance or forest silvicultural activities in the drainage of Reaches 2 and 3, thus eliminating an inorganic source of NH_4-N, this occurrence of NH_4-N must have come from enrichment by organic matter containing nitrogen during Trials 1 and 2 that did not occur or was less apparent in Trials 0 and 3. In 2019, the NH_4-N concentrations were 19 – 27 µg·L^{-1}, which was about 4 times greater than during Trial 3 but were less than half of those during Trials 1 and 2. These 2019 NH_4-N concentrations infer smaller organic enrichment than during Trials 1 and 2.

SRP concentration in Reach 4 declined over time from a peak of 3.1 µg·L^{-1} in Trial 1 to <1 µg·L^{-1} in 2019 (Table 3.3 and Figure 3.6). In contrast, SRP concentration in Reach 3 increased between Trial 0 (3.5 µg·L^{-1}) and Trial 2 (10.2 µg·L^{-1}), with the Trial 2 values being several times greater than in Reach 4, indicating addition of phosphorus in Reach 3. The same pattern occurred in Reach 2 except SRP concentrations were about the same between Trials 1 and 2. During Trial 3, mean SRP concentrations in both of Reaches 3 and 2 declined to <2 µg·L^{-1}, showing absence of phosphorus enrichment that was apparent during Trials 1 and 2. In 2019, SRP concentration in Reach 3 increased by more than 4 times from values in Trial 3 while concentrations in Reach 4 continued their time course decline (Figure 3.6). Reach 2 SRP concentration either stayed the same as in Reach 3 or declined due to dilution from the Yalakom River (SRP concentrations <1 µg·L^{-1}). Higher SRP concentrations in Reach 1 compared to Reach 2 in 2019 implied some enrichment in Reach 1.

Table 3.3 Mean values ± standard error of habitat attributes by reach and trial.

Habitat variable and units	Reach	Mean value ± standard error of arithmetic mean				
		Trial 0	Trial 1	Trial 2	Trial 3	2019
Total alkalinity (mg·L^{-1} as $CaCO_3$)	Reach 4	No flow	56.1 ± 0.7	42.4 ± 4.4	28.2 ± 0.3	33.5**
	Reach 3	168 ± 3.5	72.4 ± 0.9	55.3 ± 3.0	41.2 ± 0.9	41.9 ± 1.1
	Reach 2	192 ± 2.6	140.7 ± 1.2	110.6 ± 8.5	85.6 ± 0.3	84.4 ± 0.3
	Reach 1	Not measured	Not measured	Not measured	Not measured	86.5 ± 2.1
	Yalakom River	Not measured	Not measured	Not measured	113*	112.5**
pH	Reach 4	No flow	7.5 ± 0.05	7.6 ± 0.06	7.7 ± 0.008	7.5**
	Reach 3	8.1 ± 0.03	7.5 ± 0.02	7.7 ± 0.02	7.8 ± 0.02	7.6 ± 0.02
	Reach 2	8.1 ± 0.03	7.8 ± 0.03	8.0 ± 0.02	8.1 ± 0.005	8.0 ± 0.003
	Reach 1	Not measured	Not measured	Not measured	Not measured	8.1 ± 0.006
	Yalakom River	Not measured	Not measured	Not measured	8.2 ±	8.0**
NH_4-N (µg·L^{-1})	Reach 4	No flow	5.4 ± 0.6	11.0 ± 2.3	5.5 ± 0.3	7.8**
	Reach 3	6.1 ± 0.5	28.4 ± 4.3	57.8 ± 10.0	5.8 ± 0.2	26.5 ± 3.1
	Reach 2	8.2 ± 0.6	50.8 ± 8.0	70.2 ± 16.1	5.4 ± 0.2	19.1 ± 0.1

*Only measured in 2018 during Trial 3.

**Only one station in 2019 so no replicates for calculation of standard error.

Table 3.3 Continued.

Habitat variable and units	Reach	Mean value ± standard error of arithmetic mean				
		Trial 0	Trial 1	Trial 2	Trial 3	2019
NH$_4$-N (μg·L^{-1})	Reach 1	Not measured	Not measured	Not measured	Not measured	26.1 ± 0.9
	Yalakom River	Not measured	Not measured	Not measured	<5.0	<5.0
NO$_3$-N (μg·L^{-1})	Reach 4	No flow	19.0 ± 1.3	17.6 ± 2.2	10.2 ± 0.7	12.0**
	Reach 3	39.2 ± 4.5	30.5 ± 1.8	68.9 ± 7.5	20.9 ± 1.3	46.0 ± 4.3
	Reach 2	23.6 ± 1.6	46.9 ± 3.5	81.0 ± 7.4	49.6 ± 0.8	72.3 ± 0.9
	Reach 1	Not measured	Not measured	Not measured	Not measured	62.9 ± 2.3
	Yalakom River	Not measured	Not measured	Not measured	76.5*	72.0**
DIN (μg·L^{-1})	Reach 4	No flow	24.4 ± 1.6	28.6 ± 3.8	15.7 ± 0.7	19.8**
	Reach 3	45.3 ± 4.6	58.9 ± 5.8	126.7 ± 17	26.7 ± 1.3	72.5 ± 7.4
	Reach 2	31.8 ± 1.8	97.6 ± 10.9	151.2 ± 22.6	55.0 ± 0.7	91.4 ± 1
	Reach 1	Not measured	Not measured	Not measured	Not measured	89.1 ± 2.1
	Yalakom River	Not measured	Not measured	Not measured	81.5*	77.0**
SRP (μg·L^{-1})	Reach 4	No flow	3.1 ± 0.2	2.7 ± 0.2	1.5 ± 0.2	<1.0
	Reach 3	3.5 ± 0.1	6.6 ± 0.9	10.2 ± 1.4	1.7 ±0.1	7.8 ± 1.1
	Reach 2	3.7 ± 0.4	9.5 ± 1.7	9.2 ± 2.2	1.1 ± 0	3.9 ± 0.3
	Reach 1	Not measured	Not measured	Not measured	Not measured	5.5 ± 0.6
	Yalakom River	Not measured	Not measured	Not measured	1.0*	1.0 ± 0
Temperature (°C)	Reach 4	No flow	10.9 ± 0.1	10.1 ± 0.1	10.2 ± 0.2	10.6**
	Reach 3	8.4 ± 0.3	9.6 ± 0.1	8.2 ± 0.1	8.3 ± 0.2	8.7 ± 0.2
	Reach 2	7.5 ± 0.5	7.2 ± 0.1	5.6 ± 0.1	5.5 ± 0.2	5.5 ± 0
	Reach 1	Not measured	Not measured	Not measured	Not measured	5.2 ± 0.1
	Yalakom River	Not measured	Not measured	Not measured	4.1*	4.2**
Peak flow (m^3·s^{-1})	Reach 4	No flow	5.0 ± 0.1	16.6 ± 0.6	108.7 ± 4.5	15.7**
	Reach 3	10.2 ± 1.9	6.3 ± 0.1	18.9 ± 0.3	110.9 ± 2.2	18.3 ± 0.3
	Reach 2	36.2 ± 3.4	26.1 ± 1.4	44.5 ± 1.4	130.0 ± 2.7	51.2 ± 1
	Reach 1	Not measured	Not measured	Not measured	Not measured	58.5 ± 0.4
	Yalakom River	Not measured	Not measured	Not measured	26.7*	29.0 ± 0
Water depth at samplers (cm)	Reach 4	No flow	0.27 ± 0.02	0.29 ± 0.04	0.20 ± 0.02	0.17 ± 0.02
	Reach 3	0.17 ± 0.01	0.30 ± 0.01	0.33 ± 0.02	0.31 ± 0.02	0.21 ± 0.03
	Reach 2	0.17 ± 0.02	0.26 ± 0.01	0.30 ± 0.03	0.27 ± 0.03	0.17 ± 0.03
	Reach 1	Not measured	Not measured	Not measured	Not measured	0.18 ± 0.03
	Yalakom River	Not measured	Not measured	Not measured	0.32*	0.24 ± 0.04
Water velocity at samplers (m^3·s^{-1})	Reach 4	No flow	0.26 ± 0.03	0.26 ± 0.06	0.05 ± 0.02	0.12 ± 0.02
	Reach 3	0.27 ± 0.02	0.34 ± 0.02	0.21 ± 0.04	0.17 ± 0.03	0.10 ± 0.03
	Reach 2	0.42 ± 0.05	0.35 ± 0.02	0.42 ± 0.05	0.15 ± 0.03	0.09 ± 0.03
	Reach 1	Not measured	Not measured	Not measured	Not measured	0.06 ± 0.02
	Yalakom River	Not measured	Not measured	Not measured	0.20*	0.10 ± 0.04

*Only measured in 2018 during Trial 3.

**Only one station in 2019 so no replicates for calculation of standard error.

Table 3.3 Continued.

Habitat variable and units	Reach	Mean value ± standard error of arithmetic mean				
		Trial 0	Trial 1	Trial 2	Trial 3	2019
D50 (cm)	Reach 4	Not measured	Not measured	Not measured	Not measured	11.7 ± 0
	Reach 3	Not measured	Not measured	Not measured	Not measured	7.6 ± 0.6
	Reach 2	Not measured	Not measured	Not measured	Not measured	9.1 ± 0.2
	Reach 1	Not measured	Not measured	Not measured	Not measured	7.0 ± 0.6
	Yalakom River	Not measured	Not measured	Not measured	Not measured	3.5 ± 0.4

*Only measured in 2018 during Trial 3.

**Only one station in 2019 so no replicates for calculation of standard error.

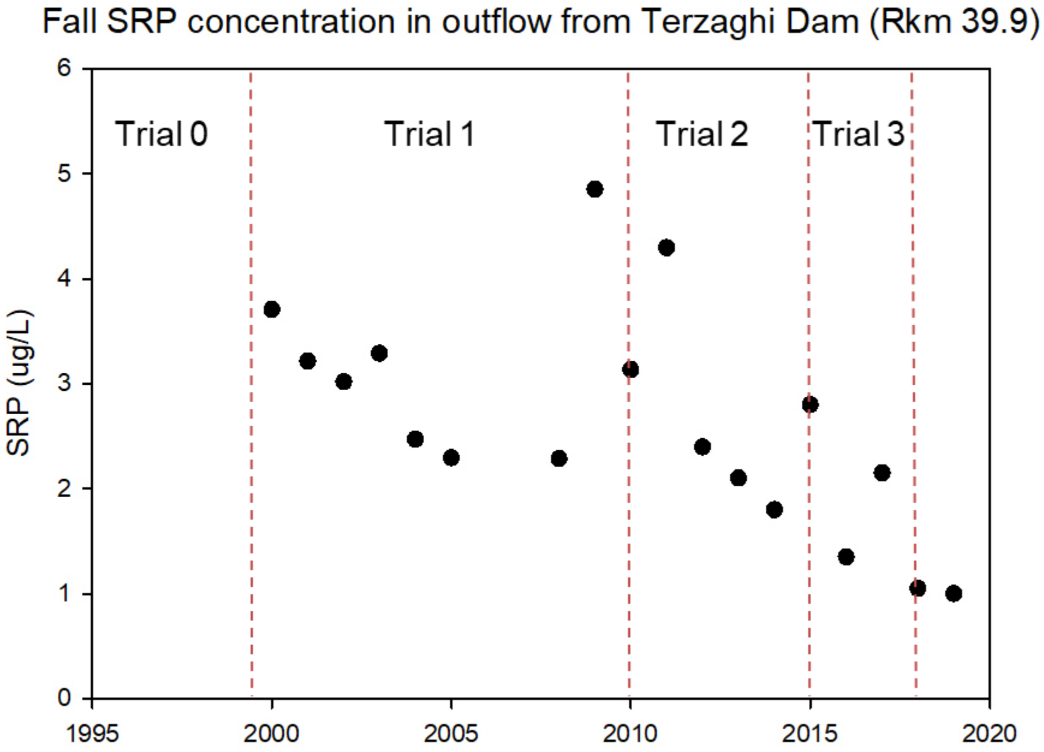


Figure 3.6 Fall SRP concentration in water released from Terzaghi Dam among years of all flow trials

The relatively high mean SRP and NH_4-N concentrations and to a smaller extent NO_3-N concentrations in Reaches 3 and 2 during Trials 1 and 2 were the combination of high concentrations in many of the odd years but consistently low concentrations in even years (Figure 3.7). These inter-annual 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).

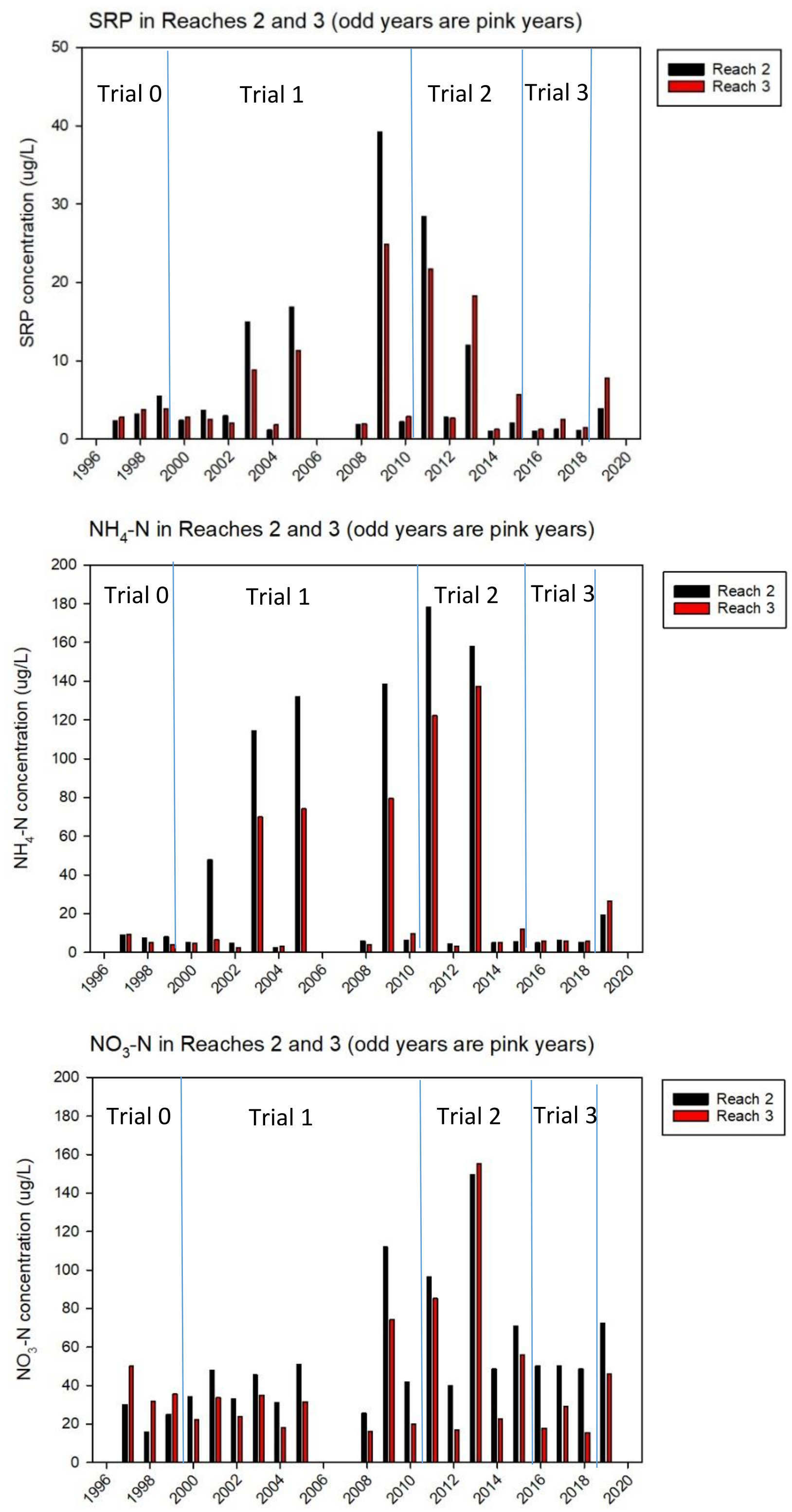


Figure 3.7 **Variation in concentration of SRP (top), NH_4-N (middle), and NO_3-N (bottom) among years in Reaches 2 and 3 with reference to pink salmon spawning in odd years.**

The variation in DIN and SRP concentration produced shifts in potential limitation of benthic algal growth at different times and places based on molar N:P ratios (Figure 3.8). 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 Trials 1 to 3, molar N:P remained about the same in Reach 3, again showing co-limitation by N and P among algal species (N:P of 20 - 50). During 2019, the Reach 3 N:P declined due to relatively high phosphorus enrichment. In contrast, N:P in Reach 2 shifted to greater potential phosphorus deficiency (>50) over time, particularly in Trials 2 and 3 due to high DIN concentration and undetectable SRP concentration in the inflowing Yalakom River. The Reach 2 N:P declined by about half in 2019 due to a four-times increase in SRP concentrations originating from Reach 3. In Reach 4, the declining SRP concentrations over time increased N:P from 17 in Trial 1 to 44 in 2019, showing increasing phosphorus deficiency with the largest shift occurring between Trial 3 and 2019.

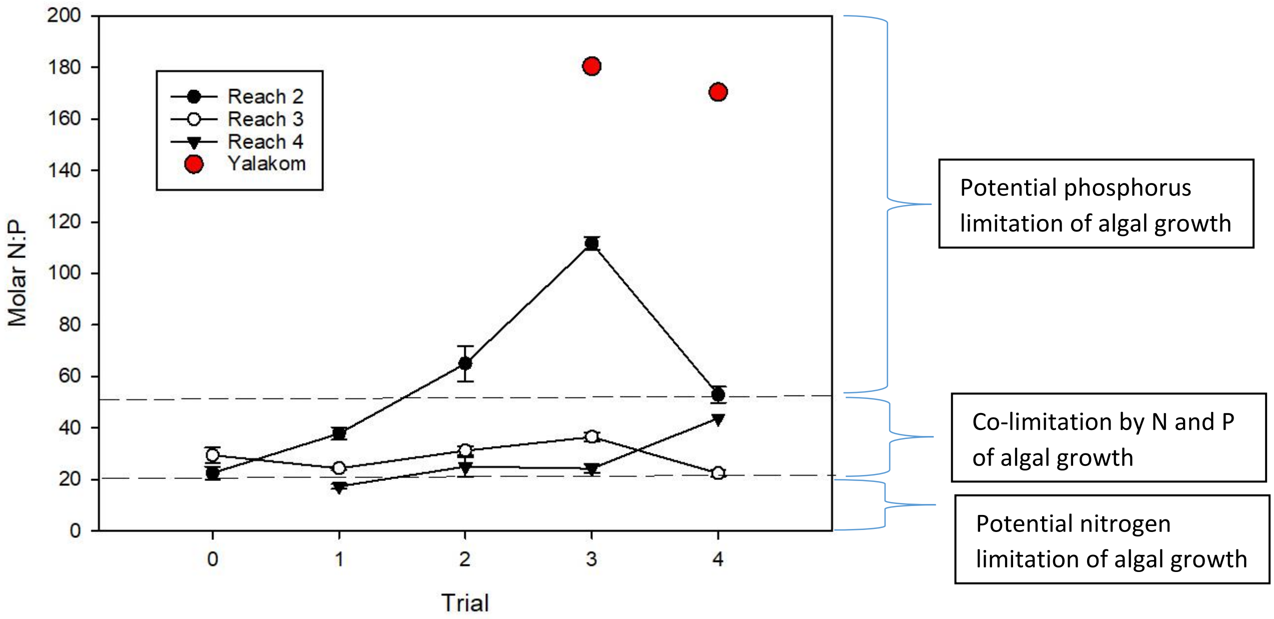
The presence of spawning Pink salmon (odd years) shifted N:P downwards in both of Reaches 3 and 2 compared to even years when pinks were absent (Figure 3.9). This shift only occurred with flow release from the dam (e.g. after Trial 0). Hence, the presence of pinks increased potential nitrogen deficiency among algae due to disproportionate phosphorus enrichment compared to nitrogen within Trials 1 to 3. The narrower gaps between N:P Pink – on lines and Pink – off lines in Reach 3 compared to Reach 3 showed the pink effect on N:P was greater in Reach 2 than in Reach 3.

Mean water temperature during the periphyton and benthic invertebrate sampling periods declined upstream to downstream (Table 3.3). Relatively high temperatures in Reach 4 were due to release of warm hypolimnetic water from Carpenter Reservoir caused by mixing of very warm (>20°C) epilimnetic water and cooler hypolimnetic water in the early fall (Limnotek 2019). The water cooled with downstream flow. Water temperature was relatively constant among trials in a given reach except in Reach 2 where temperatures have been declining among trials (7.5°C in Trial 0 down to 5.5°C in 2019) due to cooling from the Yalakom River.

Site – specific mean peak flow recorded during March 1 – August 31 of each year among flow trials matched the Trial hydrographs in Figure 1.4 (Table 3.3). Peak flow increased from Trial 0 to Trial 3, reaching a maximum of 109 $m^3 \cdot s^{-1}$ in Reach 4 and 130 $m^3 \cdot s^{-1}$ in Reach 2. In 2019, the summer peak flow preceding the fall biological sampling was similar to that during Trial 2.

Mean water depths and velocities at the periphyton and benthic invertebrate samplers were set according to placement of the samplers. Velocities were 0.05 – 0.42 $m \cdot s^{-1}$ and depths were 0.17 – 0.33 m. These ranges were representative of river margins where current was adequate to maintain optimal water exchange but not high enough to cause physical disturbance.

The median particle size of river substrata (D50) measured in 2019 was 11.7 cm in Reach 4, declining to 7 cm in Reach 1. These sizes were larger than the 3.5 cm size measured in the Yalakom River.



Potential phosphorus limitation of algal growth

Co-limitation by N and P of algal growth

Potential nitrogen limitation of algal growth

Figure 3.8 **Mean molar N:P (±standard error) by reach among trials. Trial 4 is 2019. Notation for nutrient limitation is based on criteria from Guildford and Hecky (2000).**

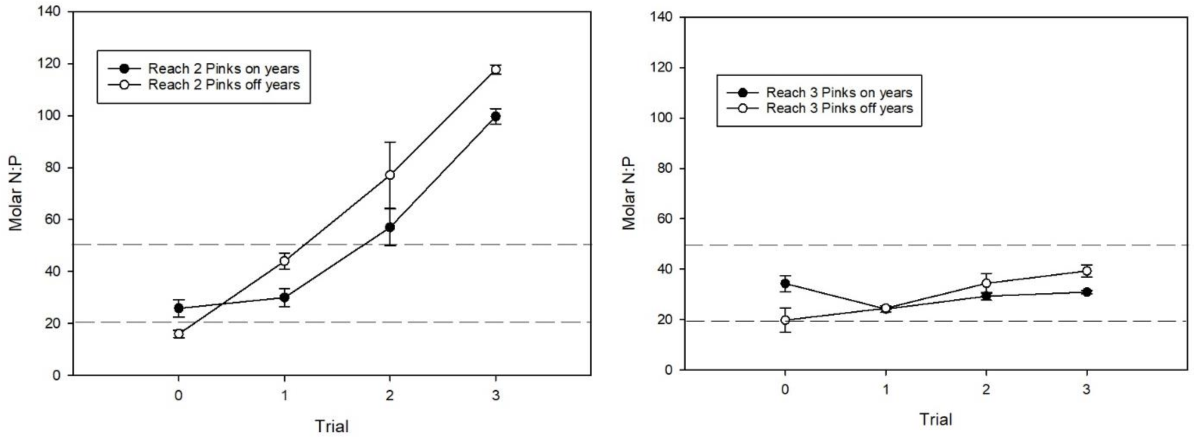


Figure 3.9 **Mean molar N:P (±standard error) in Reach 2 (left) and Reach 3 (right) stratified by Pink on years and Pink off years between Trials. N:P values above the top dashed line show potential P limitation of algal growth, values below the bottom dashed line show potential N limitation, and values in between the lines show co-limitation by N and P. The 2019 data are not shown because there was no comparable Pink – off year(s) following Trial 3 in data to date.**

3.2. Periphyton

Mean periphyton cell densities were 8,000 – 80,000 cells x $10^6 \cdot m^{-2}$ among trials and reaches (Figure 3.10). Taxa were mostly diatoms with low numbers of cyanobacteria (commonly known as blue green algae), Chlorophyta (green algae), chryso-cryptophytes (flagellates), and euglenoids (a phylum of flagellates). The flagellates likely originated as washout from Carpenter Reservoir. Small counts of Deuteromycota (fungi), Miozoa (dinoflaggelates), and Ochrophyta (golden brown algae) were combined into an "Other" category in Figure 3.10. Most abundant diatom genera included *Achnanthidium, Diatoma, Encyonema, Eunotia, Gomphonema, Nitzschia, and Rossithidium.* There were 24 less common diatoms. The cyanobacteria included *Pseudanabaena, Oscillatoria, Anabaena*, and *Lyngbya*. During Trial 2, which is being used as a reference for the 2019 periphyton, the most abundant diatom genera were the same as those in 2019 (*Achnanthidium, Encyonema, Eunotia, Gomphonema,* and *Nitzschia*). There were 42 less common diatom genera found during Trial 2; about double the count of less common genera in 2019. The cyanobacteria were more diverse in Trial 2 than in 2019 and included *Pseudanabaena, Oscillatoria, Anabaena, Aphanizomenon, Arthrospira, Gleocapsa, Lyngbya,* and *Merismopedia.*

Mean periphyton PB was 3.2 – 14.5 µg chl-a·cm^{-2} among Trials and Reaches (Figure 3.11). PB values were normally distributed so no transformations were applied prior to running the ANOVA's to test for Trial, Reach, and Pink effects. All interactions of those effects were not significant ($p \geq 0.1$), which meant that Trial, Reach, and Pink effects on PB could be examined independently. No Trial effect ($p=0.1$) and no Reach effect ($p=0.09$) on PB was found (Table 3.4). Presence of Pinks in the odd years resulted in significantly greater PB than did absence of Pinks in the even years ($p=0.02$, Table 3.4, Figure 3.11). Increasing overlap of PB standard errors from Reach 2 to Reach 4 among all Trials showed greater effect of Pinks on PB in Reach 2 than in Reach 4, which is consistent with the greater effect of Pinks on molar N:P in Reach 2 than upstream in Reach 3 (Figure 3.9).

The mean PB in 2019 contrasted with 95% confidence intervals around the PB mean calculated from Trial 2 showed no significant difference in PB at the similar flows between 2019 and the Trial 2 years among all reaches (Figure 3.12). This finding is consistent with finding no Trial effect on PB (Table 3.4).

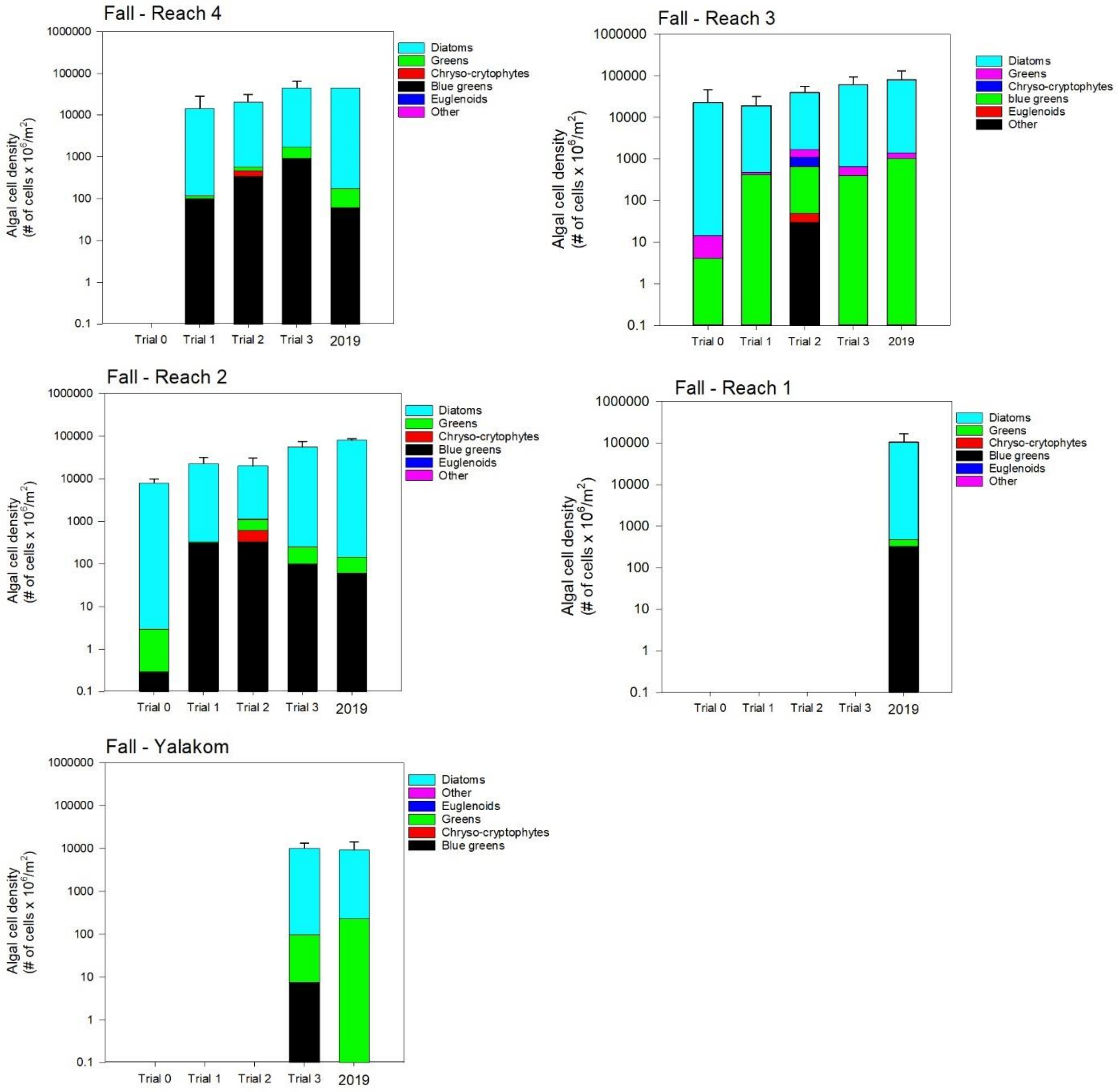


Figure 3.10 **Mean algal cell density (± standard deviation) of algal classes in the fall in each of Reach 4 (top left), Reach 3 (top right), Reach 2 (middle left), Reach 1 (middle right), and the Yalakom River.**

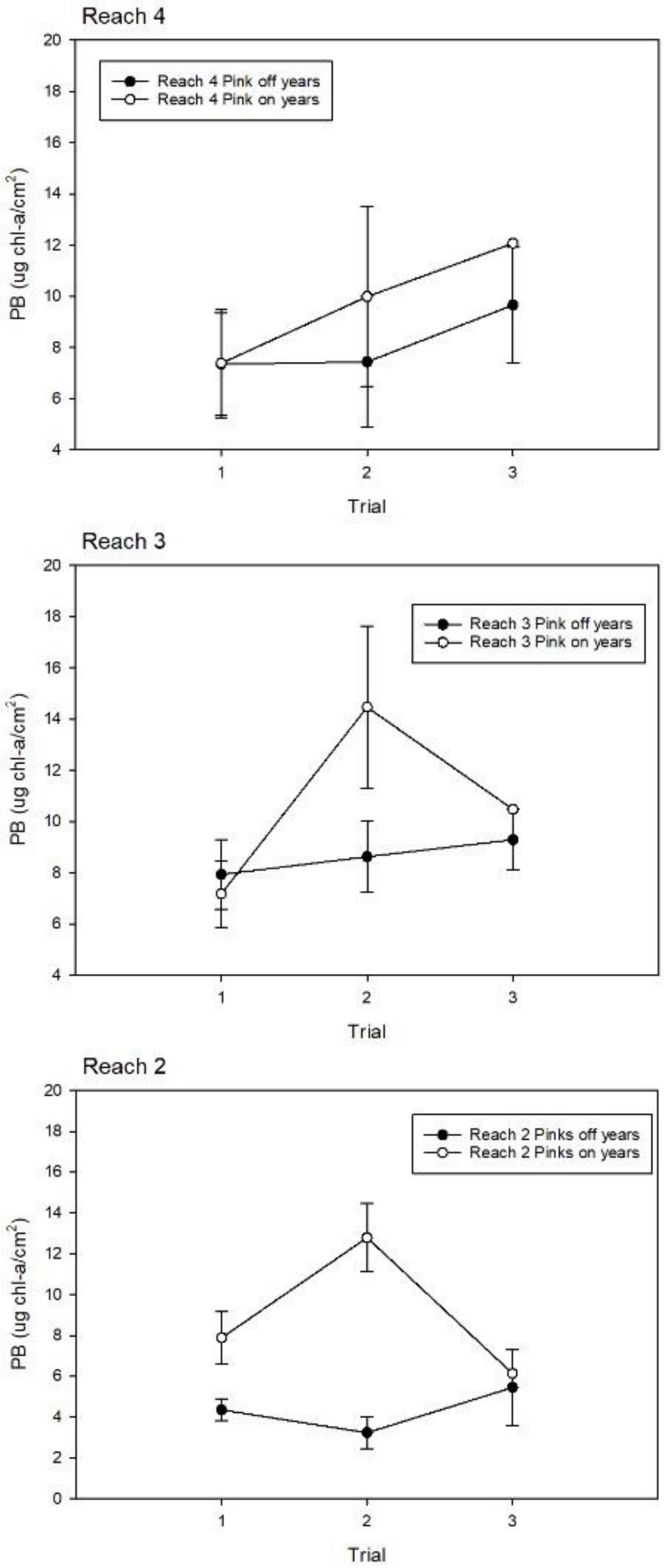


Figure 3.11 **Mean periphyton PB (± standard error) by Trial among Pink salmon spawning on and off years in Reach 4 (top), Reach 3 (middle), and Reach 2 (bottom).**

Table 3.4 Results of PB ANOVA showing partitioning of variance between main effects (trial, reach, pink) and residual variance (error). P values less than 0.05 indicated a significant effect.

Source	Sums of squares	Degrees of freedom	Mean Squares	F-Value	p-Value
Trial	61.0	2	30.5	2.5	0.1
Reach	65.0	2	32.5	2.6	0.09
Pink	75.2	1	75.2	6.1	0.02
Trial*Reach	30.3	4	7.6	0.6	0.66
Trial*Pink	62.4	2	31.2	2.5	0.1
Reach*Pink	16.2	2	8.1	0.7	0.53
Trial*Reach*Pink	20.0	4	5.0	0.4	0.80
Error	406.7	33	12.3		

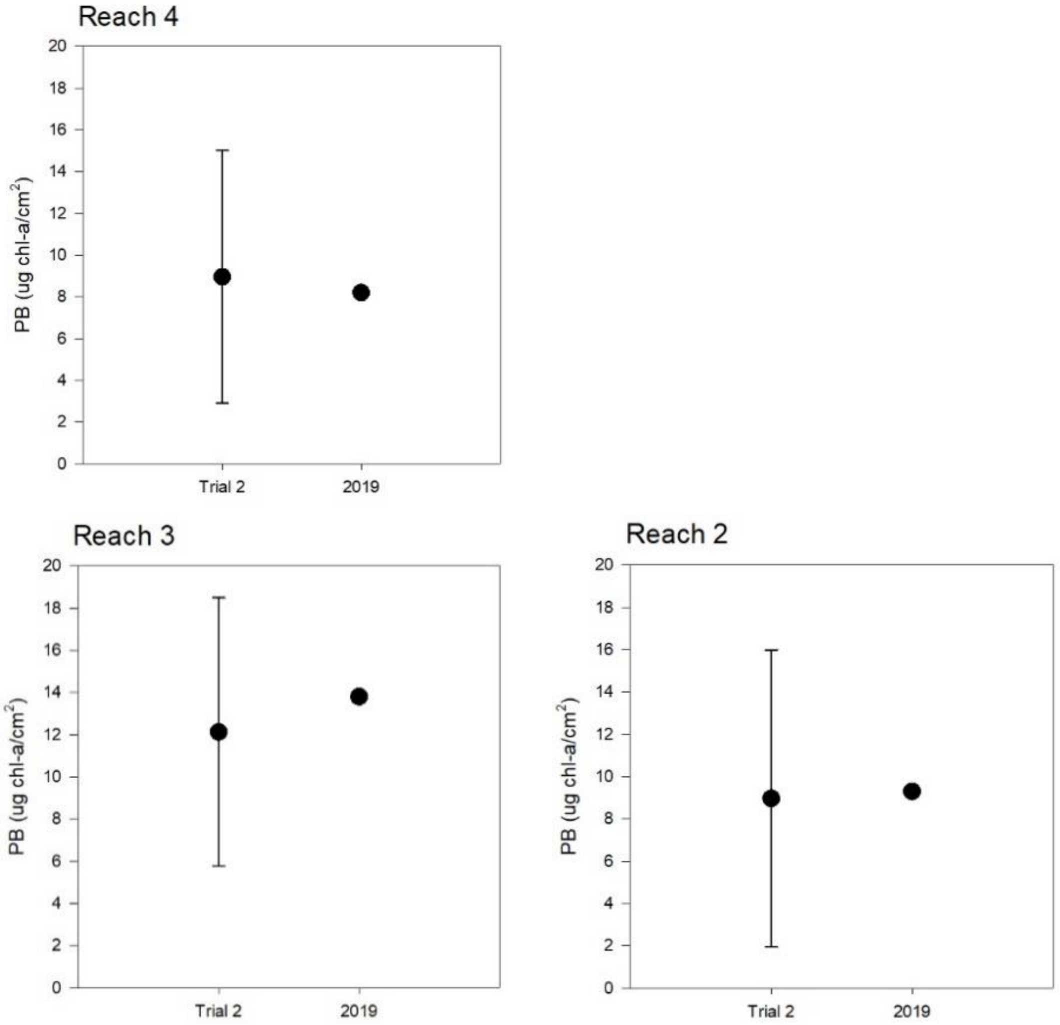


Figure 3.12 Mean PB in Trial 2 (±95% confidence interval) compared to PB in 2019. If the 2019 value is within the range of Trial 2 95% confidence interval, the 2019 value was considered not significantly different from the Trial 2 value.

3.3. Benthic Invertebrates

Benthic invertebrates in the Lower Bridge River included Ephemeroptera (mayflies), Plecoptera (stoneflies), Tricoptera (caddisflies), Diptera (true flies, including chironomids) and "Other" taxa including Oligochaeta, ostracods, Hemiptera and other true bugs (Figure 3.13). Mean density of all taxa was 67,000 animals·m^{-2} in Trial 0, 90,000 animals·m^{-2} in Trial 1, about the same in Trial 2, a decline by 69% to 28,000 animals·m^{-2} in Trial 3, and a subsequent increase to 35,000 animals·m^{-2} in 2019. Yalakom River samples included the same taxa found in the Lower Bridge River (Ephemeroptera, Plecoptera, Tricoptera, Diptera) (Figure 3.13) with a mean invertebrate density of 45,300 animals·m^{-2} in 2018 and 18,000 animals·m^{-2} in 2019. Mean invertebrate density in Reach 1 that was sampled for the first time in 2019 was 13,000 animals·m^{-2}. Taxa in Reach 1 were similar to those in the other Bridge River reaches.

There were no significant interactions of Reach, Trial, and Pinks on benthic invertebrate metrics, which meant the main factors could be examined independently (p=0.07 to 1.0 among all interactions, Table 3.5). Total benthos density was significantly lower during Trial 3 compared to the other Trials (Tukey pairwise contrast p<0.001, Figure 3.14). All common taxa contributed to this effect of Trial 3 (Figure 3.14). It was the same between Trials 1 and 2. Total benthos density was not different between reaches (p=0.54). This same pattern was found for the EPT. Chironomid density was also lower during Trial 3 compared to the other trials (p=0.001). Unlike the EPT, chironomid density was lower in Reach 3 than in the other reaches among all flow Trials (Tukey pairwise contrast, Reach 2 x Reach 3 p=0.046, Reach 3 x Reach 4 p=0.009).

Family richness and Simpsons Index showed different responses to Trial and Reach (Figure 3.14, Table 3.5). Richness declined during Trial 3 compared to Trials 1 and 2 (Tukey test p<0.001) while Simpsons Index and thus taxonomic heterogeneity was not affected by the flow trials (p=0.78). Richness was lowest in Reach 4, highest in Reach 2, and in between in Reach 3 among all flow trials (Tukeys test, p<0.009), which showed addition of taxa with distance from the dam. Simpsons Index that was more sensitive to common taxa than was richness was lower in Reach 4 compared to Reaches 3 and 2 (Tukeys test, p<0.01, Reach 3 x 2 contrast p=0.58), showing that community heterogeneity increased between Reach 4 and Reach 3 but it didn't change further downstream.

Density of all invertebrates in 2019 was at the low end of the lower 95% confidence interval in Trial 2 in Reaches 4 and 2 while it was closer to the Trial 2 mean in Reach 3 (Figure 3.15). EPT density followed a similar pattern, as expected because the EPT were a large part of total benthos. In contrast, chironomid density in 2019 was closer to that in Trial 2 in Reaches 4 and 3 but was at the low end of the lower confidence interval in Reach 2.

Diversity metric values in 2019 were lower than or close to the low end of the 95% confidence intervals during Trial 2 (Figure 3.16). Richness was lower in 2019 than in Trial 2 across all reaches. In both of Trial 2 and in 2019, richness increased upstream to downstream. Simpsons Index in

Reaches 2 and 3 was lower in 2019 than during Trial 2 and it was at the low end of the Trial 2 95% confidence interval in Reach 4.

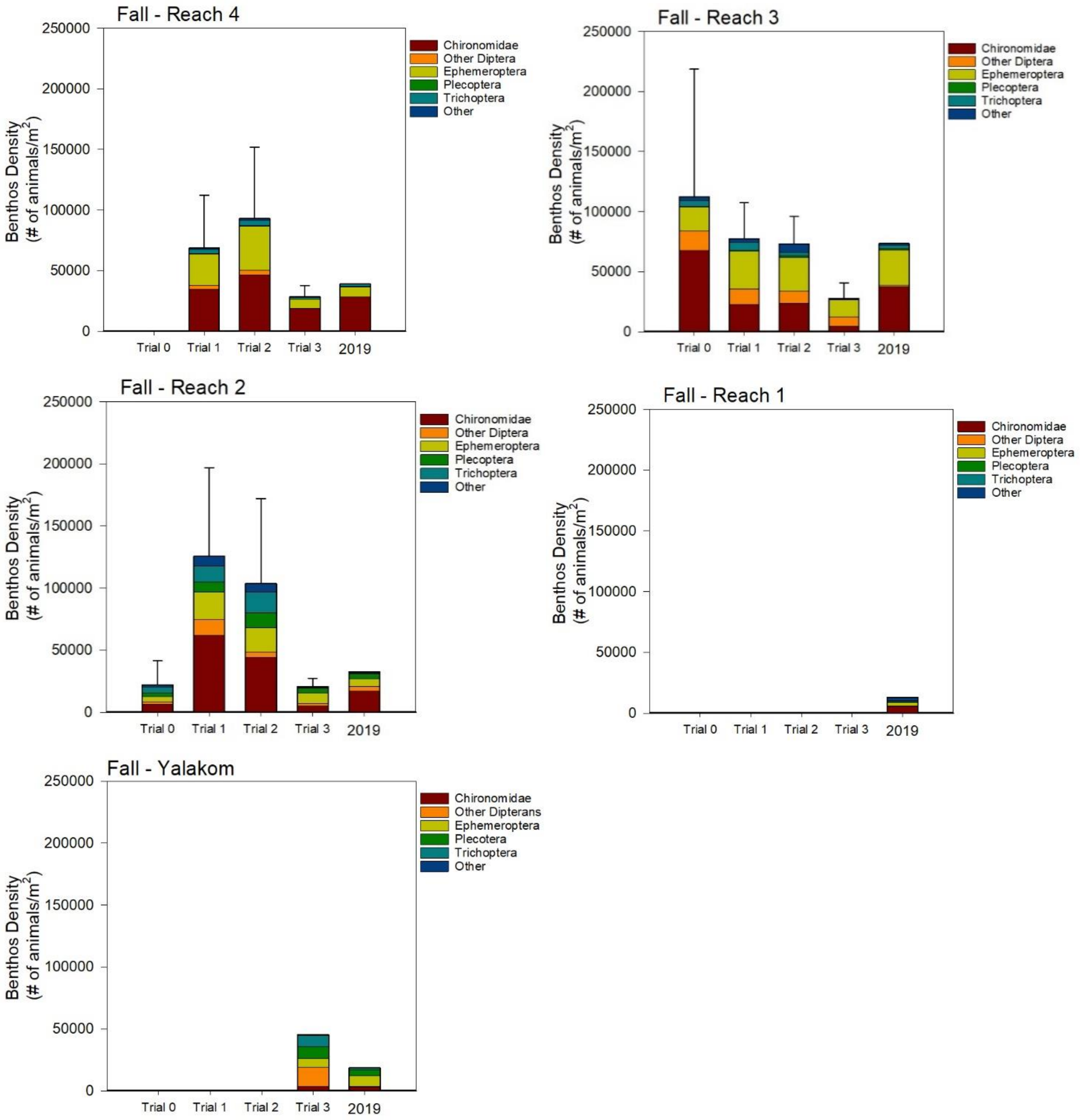


Figure 3.13 Mean abundance (±standard deviation) of invertebrate orders in the fall among trials plus 2019 in Reach 4 (top left), Reach 3 (top right), Reach 2 (middle left), Reach 1 (middle right) and the Yalakom River (bottom left).

Table 3.5 Probabilities of effect of main factors and interactions of factors on benthos metrics. Values <0.05 show a significant effect (green shading).

Source in ANOVAs	P values from ANOVAs			
	Total benthos	Chironomids	EPT	Richness
Reach	0.56	0.002	0.38	<0.001
Trial	<0.001	0.002	<0.001	<0.001
Pink	0.03	<0.001	0.29	0.18
Trial*Reach	0.57	0.11	0.43	0.56
Reach*Pink	0.48	0.26	0.60	1
Trial*Pink	0.99	0.89	0.9	0.07
Trial*Reach*Pink	0.86	0.87	0.43	0.85

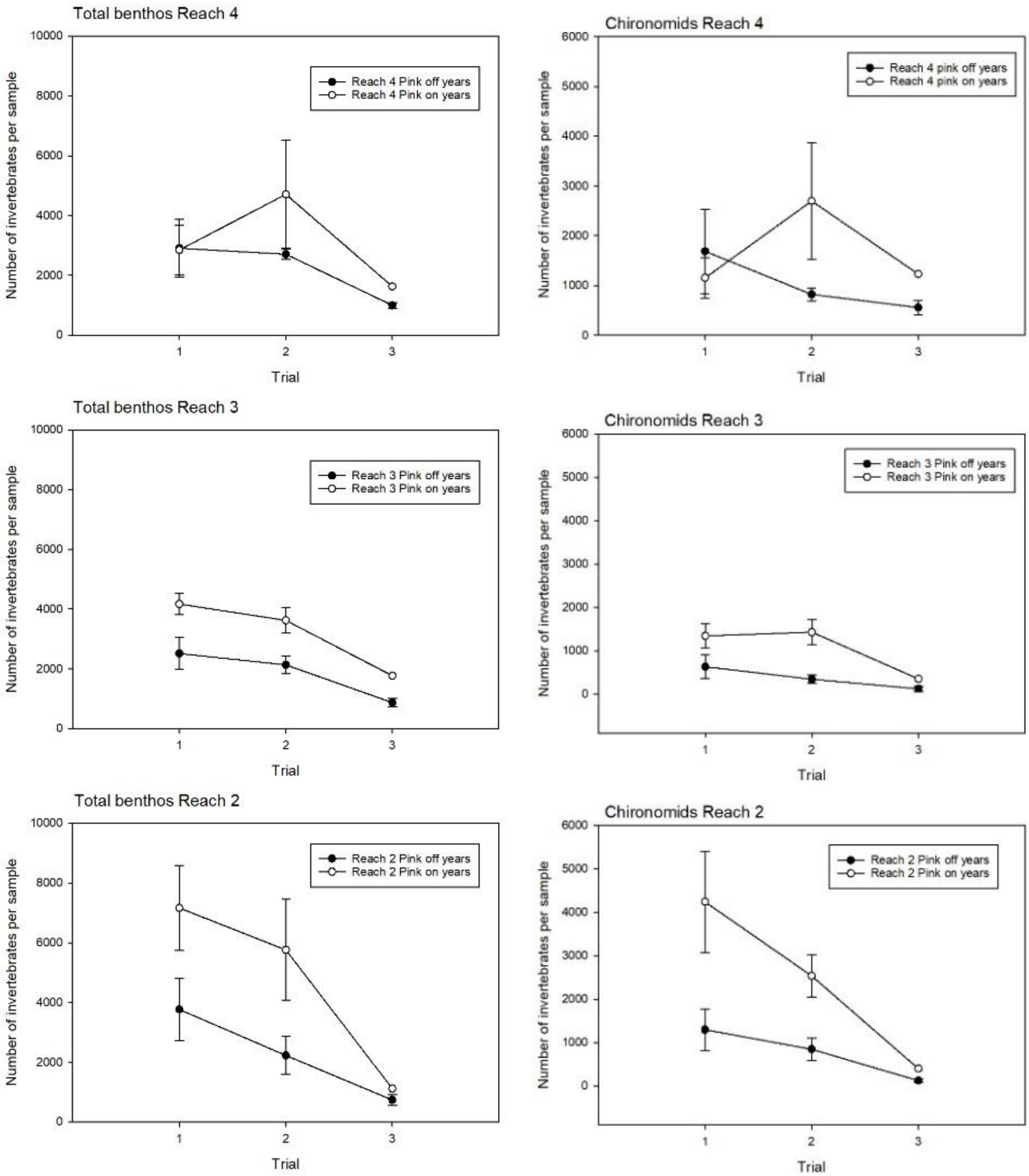


Figure 3.14 **Arithmetic mean ± standard error of total invertebrates (left column of plots) and chironomidae (right column). Continued on next page.**

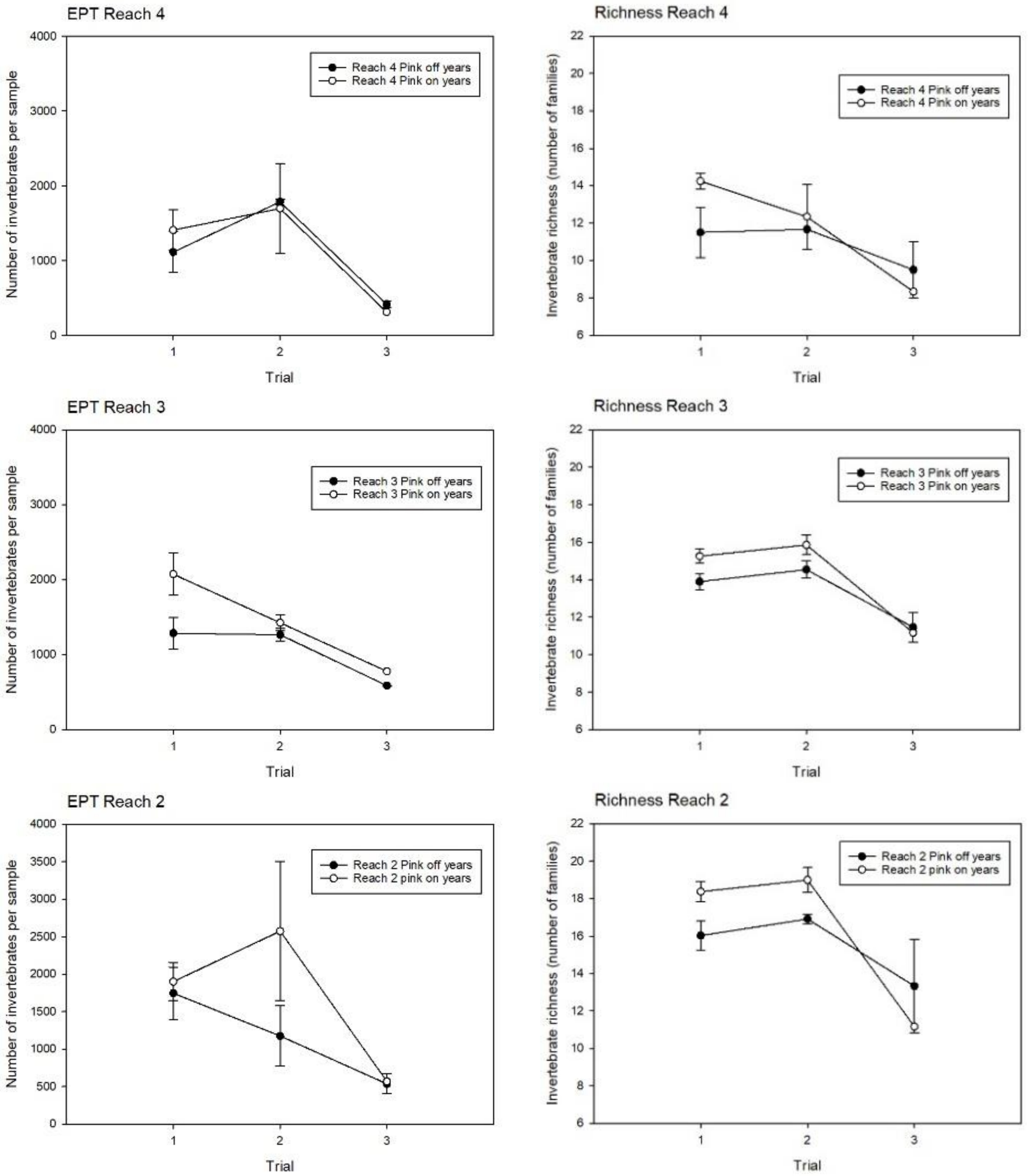


Figure 3.14 (Continued). Arithmetic mean ± standard error of EPT (left column of plots) and family richness (right column) by trial and Reach during pink salmon on and off years.

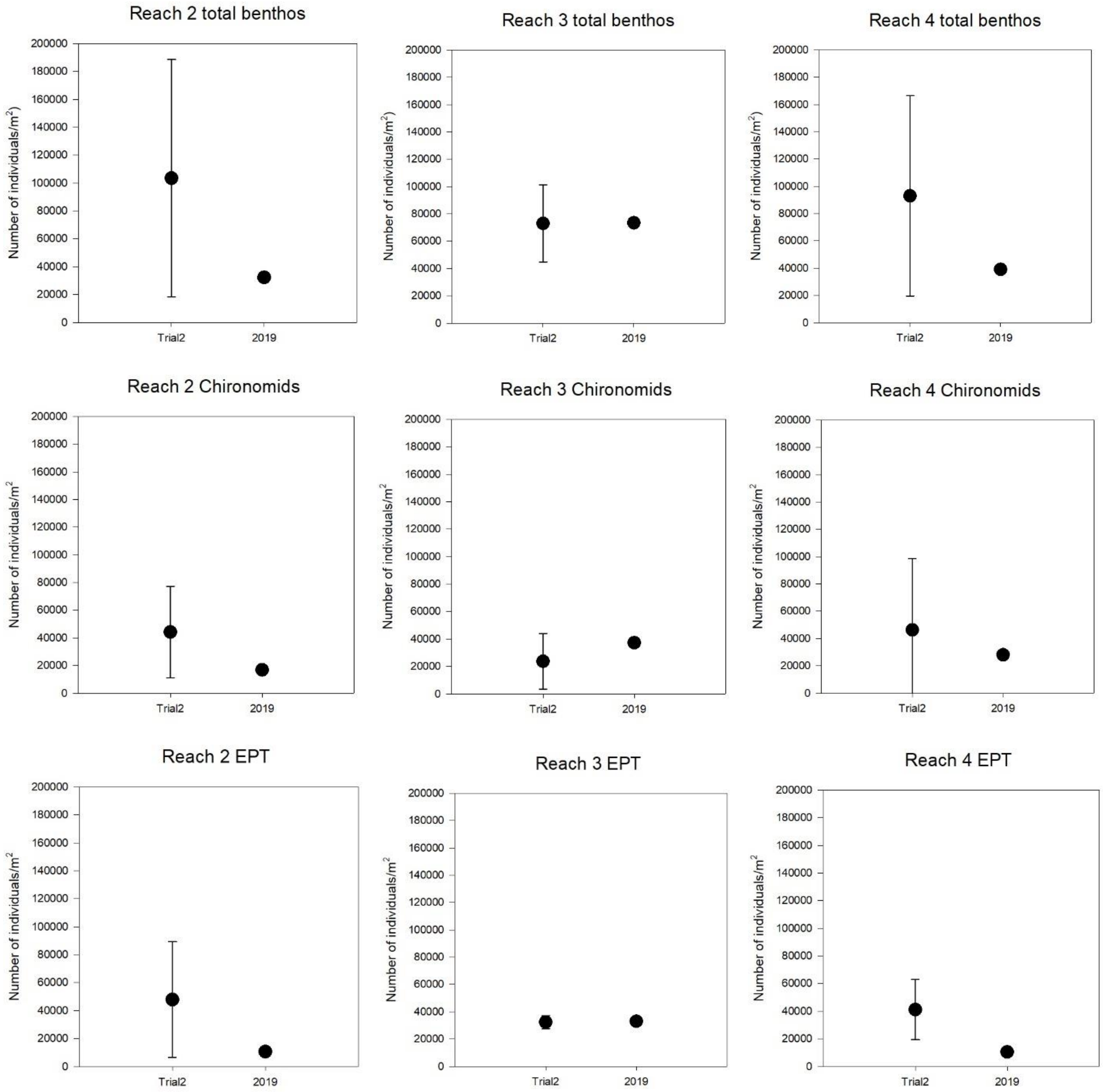


Figure 3.15 **Values of invertebrate density metrics compared between 2019 and Trial 2. Trial 2 data are means by reach ± 95% confidence intervals. The 2019 value is the mean among all samples in a reach.**

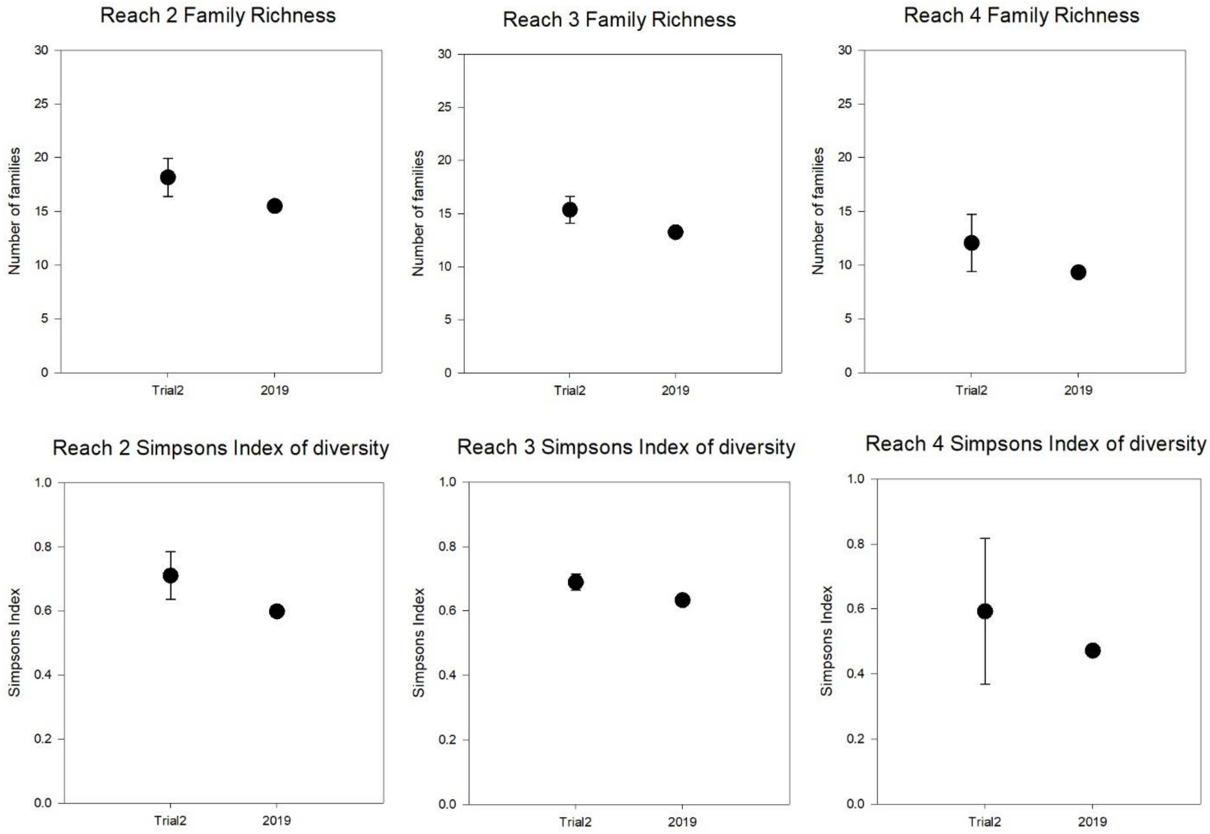


Figure 3.16 **Values of invertebrate diversity metrics compared between 2019 and Trial 2. Trial 2 data are means by reach ± 95% confidence intervals. The 2019 value is the mean among all samples in a reach.**

3.4. Juvenile Fish Production

3.4.1. Size and Condition

Mean weight of mykiss fry (Age-0+) in all reaches was almost always higher during the high flow period (Trial 3; 2016-2018) compared to other treatment periods (Figure 3.17). This likely occurred for a few possible reasons: 1) reduced density (see Figure 3.20 in Section 3.4.2) which reduced competition for available food; 2) warmer temperatures during the summer rearing period (see Figure 3.4) 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. Growth in reach 3 was also higher during the Trial 0 pre-flow period (0 $m^3 \cdot s^{-1}$) likely due to the higher benthic invertebrate abundance (see Figure 3.13 in Section 3.3) combined with the quality rearing conditions in this reach prior to the flow release. Mean size was also greater during the high flow period for mykiss parr (Age-1+), particularly in reaches 3 and 4; however, there was considerable overlap in standard deviation error bars.

Patterns in mean weight for coho fry across flow treatments in reaches 2, 3 and 4 closely matched the patterns seen for mykiss fry and were likely caused by higher growth in Reach 3 during Trial 0 due to better food availability (benthic invertebrate abundance), and increased size during the high flow period due to the same set of reasons provided for mykiss fry, above. Average weight of coho fry increased during Trial 2 (6 $m^3{\cdot}s^{-1}$) relative to Trial 1 (3 $m^3{\cdot}s^{-1}$), but there was considerable overlap in standard error bars. Abundance of coho fry decreased slightly overall from Trial 1 to Trial 2 (see Figure 3.20), so the potentially larger size under Trial 2 could be due to lower densities.

In Reach 2, mean weight of Chinook fry was higher under Trial 2 (6 $m^3{\cdot}s^{-1}$) and the Trial 3 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 lower density. In Reach 3, mean weight was higher under the Trial 1 and 2 treatments relative to Trial 0 but there was considerable overlap in error bars owing to large variance in mean weight during the pre-flow period. Mean weight was highest in reaches 3 and 4 during the Trial 3 high flow years, likely due to reduced competition from lower fish densities overall, as well as earlier emergence (relative to Trial 0; See Section 3.1.3).

Given uncertainty in the factors driving differences in mean weight among the flow trials (particularly for the Trial 3 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 3.18). Interestingly, the condition factor values showed a different pattern among flow trials than the mean weight data, but similarly, there was extensive overlap among the standard deviations. In most cases, the highest mean K values were during the pre-flow period (Trial 0) and lowest values were during the Trial 3 high flow period. This was true for mykiss fry, coho fry and chinook fry in reaches 2 and 3. Exceptions for coho and chinook fry were in Reach 4 where highest K values for that reach were during Trial 3; however, these values were also based on the smallest sample sizes in the analysis (Trial 3 coho fry *n*= 30; Trial 3 chinook fry *n*= 13). For mykiss parr, lowest K values were in Trial 3, but highest K values for this age class were in flow trials 1 and 2, rather than Trial 0. Condition factors in Trial 1 vs. Trial 2 were generally very similar for all species and age class combinations.

We also compared condition factor values from 2019 with the Trial 2 years (2011 to 2015) that occurred prior to the high flows (Figure 3.19). These comparisons showed that, despite the same flow release throughout the year, K values in 2019 were generally at or below the Trial 2 lower standard deviation line for each species and age class in each reach. Exceptions were 2019 mykiss parr and chinook fry in Reach 3 which were equivalent to the Trial 2 means, and chinook fry in Reach 4 which had a higher K value than the upper standard deviation line; however, the sample size for this data point was very small (*n*= 6).

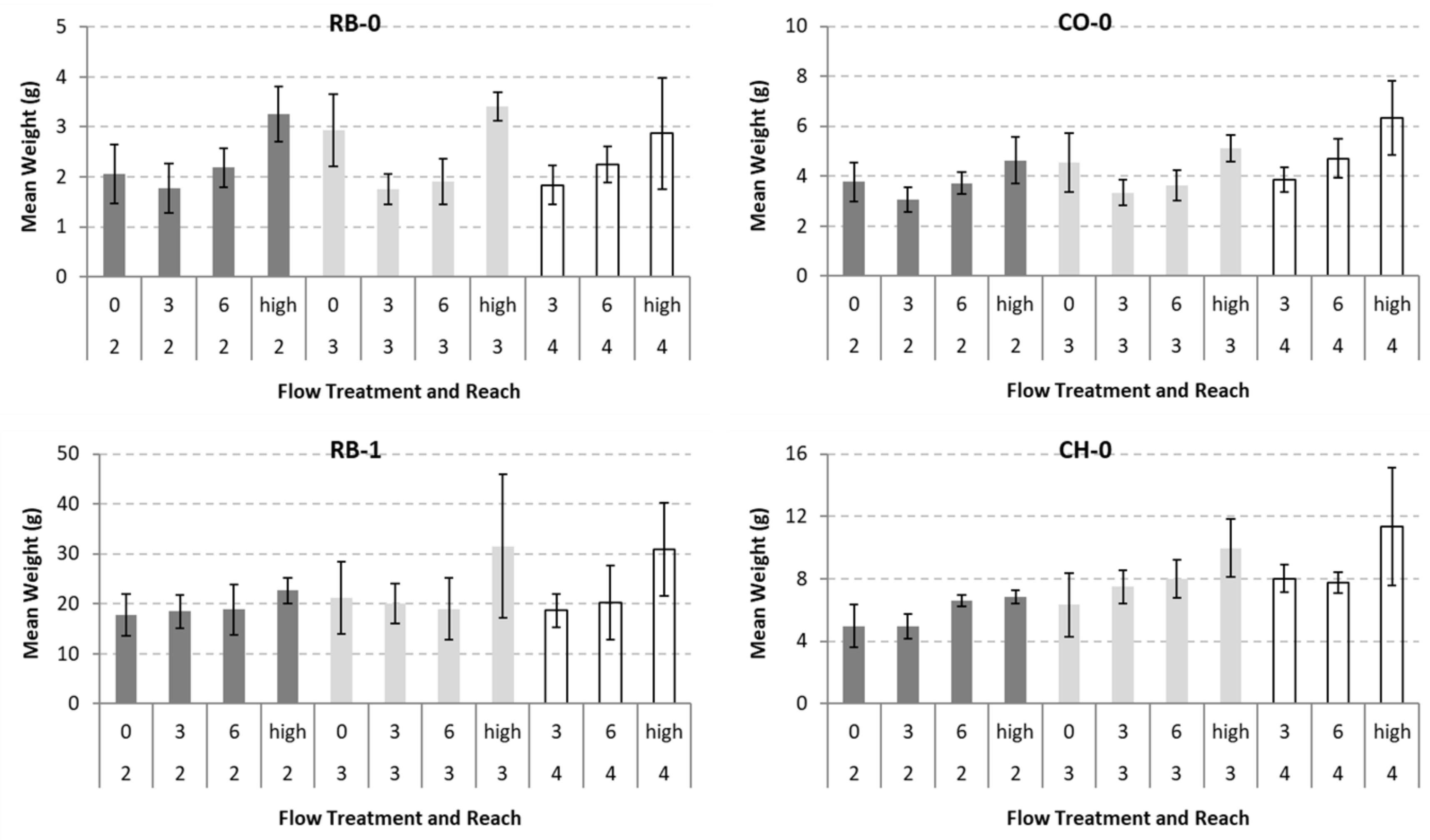


Figure 3.17 Mean juvenile salmonid weight during fall standing stock assessments across flow treatments (0, 3, and 6 $m^3 \cdot s^{-1}$ trials and the high flow period) and reaches (2, 3, and 4). RB-0, RB-1, CO-0, and CH-0 denote mykiss fry, mykiss parr, coho fry, and chinook fry, respectively. Height 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).

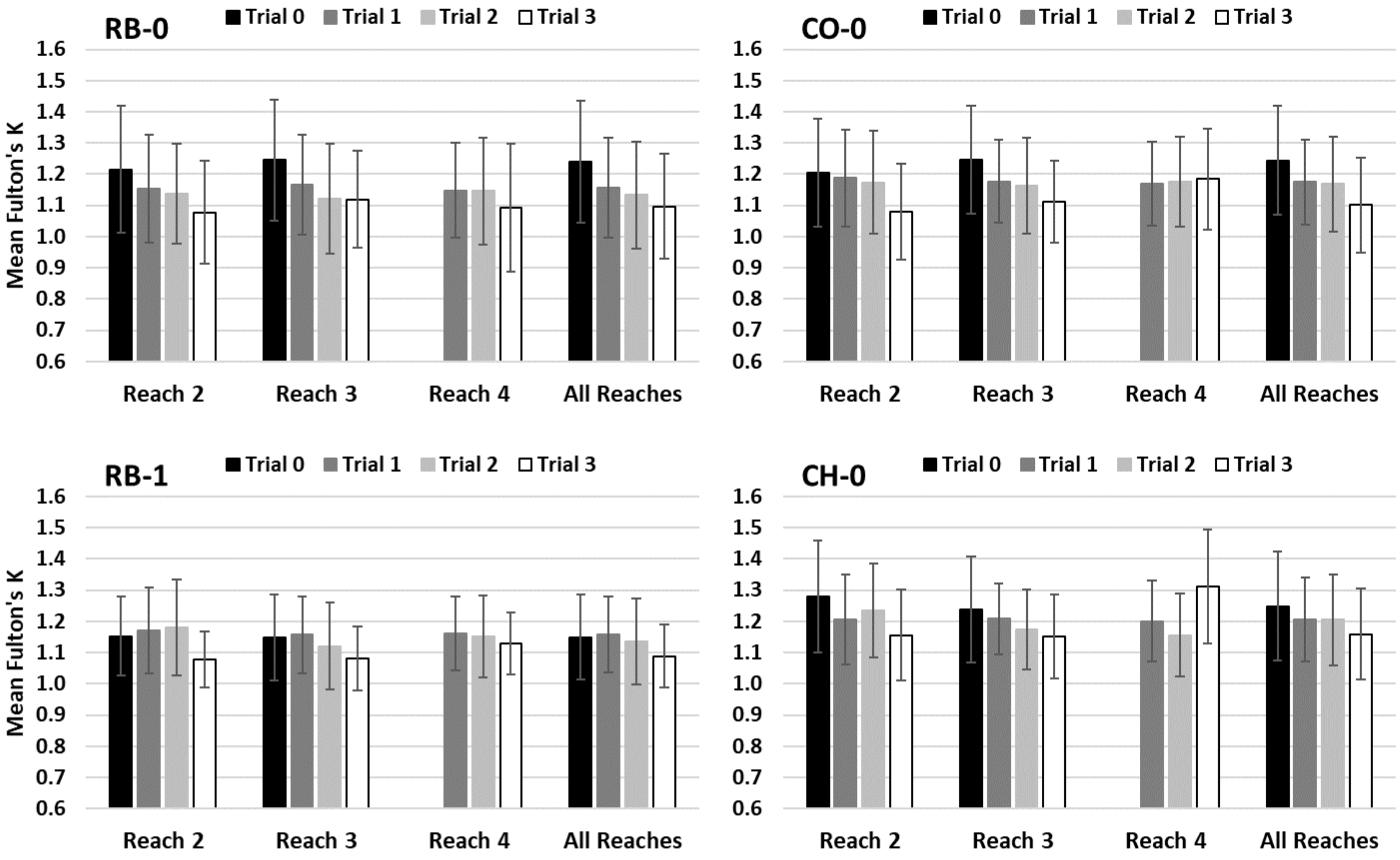


Figure 3.18 Mean condition factor for juvenile salmonids during fall standing stock assessments across flow trials (0, 1, 2 and 3) and reaches (2, 3 and 4). Species and age designations are the same as described for Figure 3.17. Height of bars represents the means of annual values for each reach-flow treatment combination and error bars denote 1 standard deviation (variation in annual values within treatments).

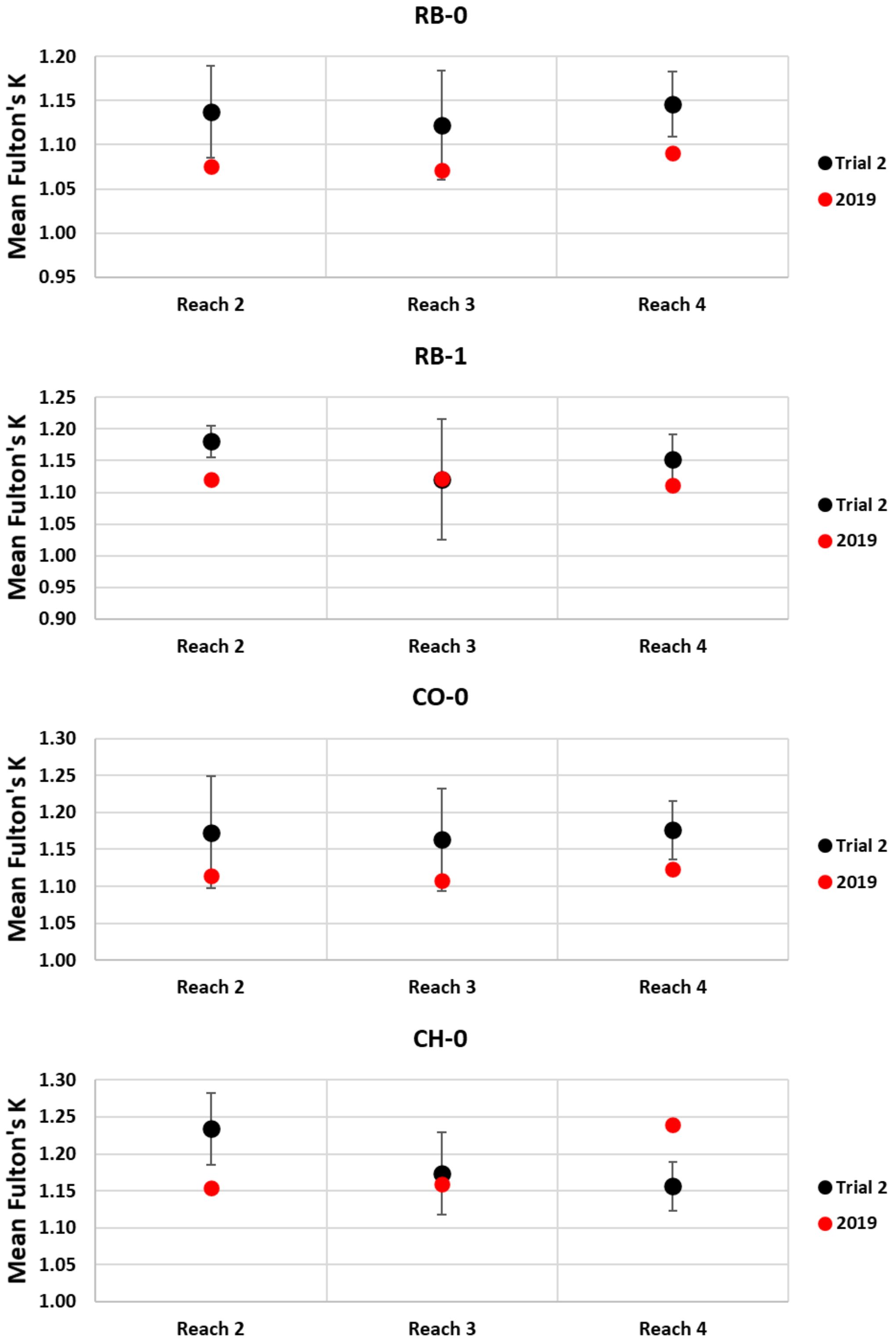


Figure 3.19 **Mean condition factor (Fulton's K value) between 2019 and all Trial 2 years for mykiss fry, mykiss parr, coho fry and Chinook fry in reaches 2, 3 and 4.**

3.4.2. Abundance and Biomass

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 and 3 and there was substantial new production in Reach 4 (Table 3.6a, Table 3.7, and Figure 3.20). Mykiss fry abundance increased by an average of 1.9- and 1.8-fold under the Trial 1 and Trial 2 (6 $m^3 \cdot s^{-1}$) treatments compared to Trial 0, respectively. In contrast, mykiss fry abundance during the high flow years (2016–2018) was only 0.42-fold of the abundance under Trial 0 (i.e., abundance was 58% lower). Relative to Trials 1 and 2, the high flow mykiss abundance was 0.22- and 0.23-fold, or a decline of 78% and 77%, respectively. In 2019, mykiss fry abundance increased 1.34-fold from mean high flow abundance but was 0.31-fold (or 69% lower than) the mean Trial 2 abundance.

Table 3.6 Average total abundance (a, '000s) and biomass (b, kg) of juvenile salmonids in the Lower Bridge River across all reaches by flow treatment (2019 included as a stand-alone column). 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	Trial 3	2019
RB-0	90	174	162	38	51
RB-1	36	35	33	10	8
CO-0	25	81	76	8	23
CH-0	38	22	13	13	12

b) Biomass

Species-Age	Trial 0	Trial 1	Trial 2	Trial 3	2019
RB-0	249	305	282	124	141
RB-1	690	653	554	326	243
CO-0	108	281	255	39	89
CH-0	228	134	91	114	72

Mykiss parr abundance increased a small amount in Reach 2 from Trial 0 to Trial 1 while there was a large decrease (i.e., ~13,000 less) in Reach 3. Trial 1 produced about 11,000 additional parr in Reach 4. Across reaches there were negligible changes in mykiss parr abundance across the Trial 0, Trial 1, and Trial 2 treatments. Mykiss parr abundance under high flows was 1/3rd of the Trial 2 average, representing a decrease of approx. 70%. Note that this average high flow abundance was based on only two years (2017 and 2018) and was within the range of low annual abundance estimates observed during the pre-treatment period. The abundance of mykiss parr in 2019 was very comparable to the Trial 3 (high flow) average. The 2019 cohort reflects mykiss that were recruited and reared under high flows in 2018 and were in their second year of rearing under lower flows (i.e., the Trial 2 hydrograph). Because of this mixed treatment effect, this cohort will not be included in any trial averages (i.e., Trial 3 or post-high flow years) going forward.

Table 3.7 Relative number of fish produced (by species and age class) under each flow treatment (2019 included as a stand-alone column). 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 values comparing 2019 values to the Trial 2 averages are highlighted yellow.

Species-Age Class	Flow Treatment (Mean Annual Release)				
		Trial 1	Trial 2	High Flows	2019
RB Age-0+	Pre-Flow	1.9	1.8	0.4	0.6
	Trial 1		0.9	0.2	0.3
	Trial 2			0.2	0.3
	High Flows				1.3
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				0.9
CH Age-0+	Pre-Flow	0.6	0.3	0.3	0.3
	Trial 1		0.6	0.6	0.5
	Trial 2			1.0	0.9
	High Flows				0.9
CO Age-0+	Pre-Flow	3.3	3.1	0.3	0.9
	Trial 1		0.9	0.1	0.3
	Trial 2			0.1	0.3
	High Flows				2.9
All Salmonids	Pre-Flow	1.7	1.5	0.4	0.5
	Trial 1		0.9	0.2	0.3
	Trial 2			0.2	0.3
	High Flows				1.4

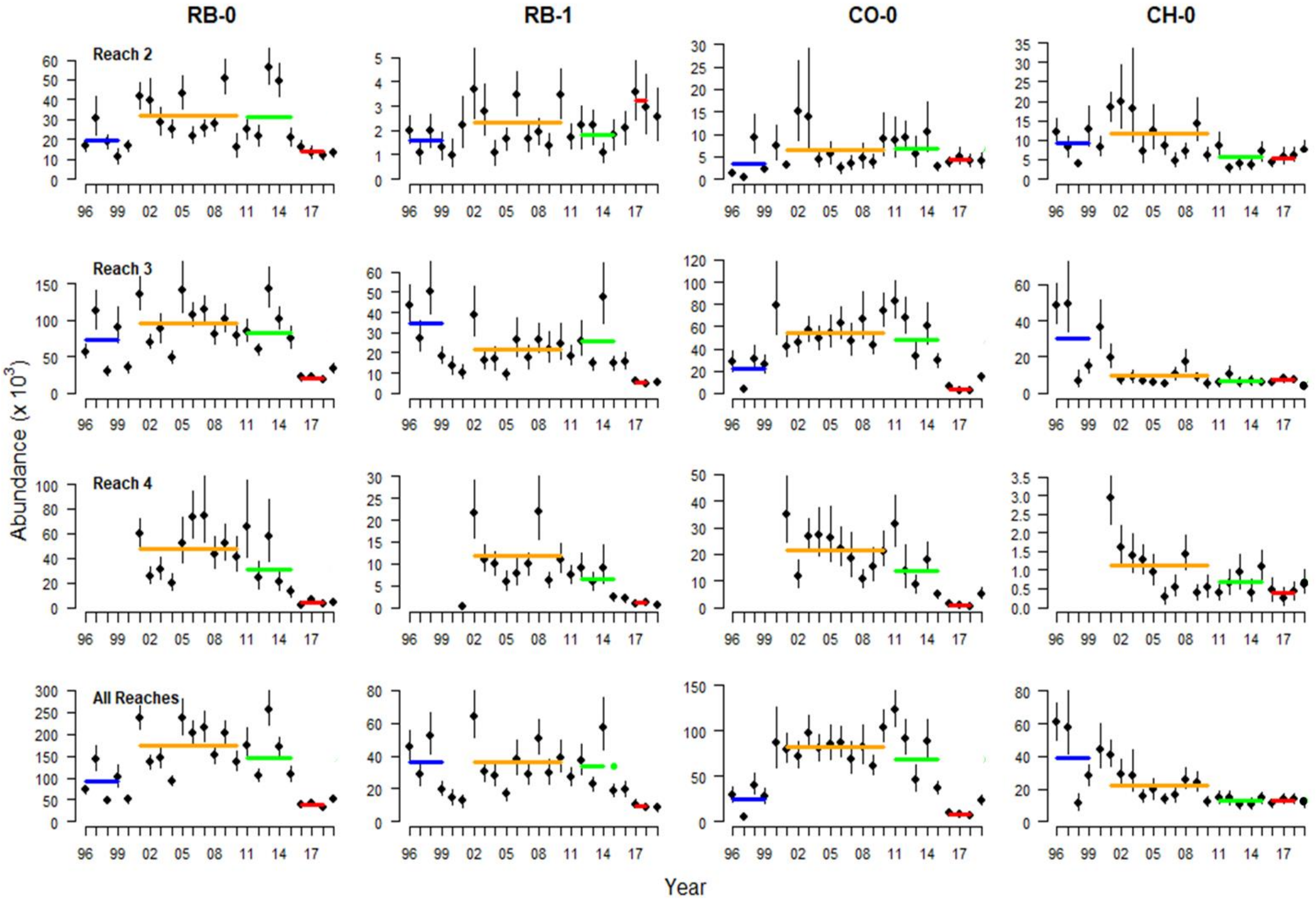


Figure 3.20 Abundance (in thousands) of juvenile salmonids in the lower Bridge River by reach (row) and species-age class (column). Points and vertical lines show mean values and 90% credible intervals from posterior distributions of abundance for each year from the hierarchical Bayesian model, respectively. Blue, orange, green and red lines show the mean values for trials 0, 1, 2, and 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.

Coho fry abundance trends followed those for mykiss fry with increases in reaches 2 and 3 between Trial 0 and Trial 1 and substantial gains in Reach 4, and little change in abundance under Trial 2 (Table 3.6a and Figure 3.20). On average, coho fry abundance increased by 3.27- and 3.08-fold under the Trial 1 and Trial 2 treatments compared to under the Trial 0 pre-flow condition, respectively (Table 3.7). Similar to mykiss fry, coho fry abundance during the high flow years (2016–2018) was only 1/3rd of the abundance under Trial 0, and 1/10th of the abundance estimates under trials 1 and 2 (i.e., a 90% reduction). In 2019, coho fry abundance increased 2.88-fold from mean high flow abundance, and was 0.94-fold of the Trial 0 mean for this species, but was 0.29- and 0.31-fold (or 71% and 69% lower than) the mean Trial 1 and Trial 2 abundances, respectively.

Chinook fry abundance increased slightly in Reach 2 under the Trial 1 treatment relative to Trial 0, but declined in Reach 3 likely owing to higher incubation temperatures which result in premature emergence in that reach (Table 3.6a and Figure 3.20). Chinook recruitment in Reach 4 has been low across all flow treatments. As a result of these factors, chinook fry abundance under the Trial 1 and 2 treatments and high flows (Trial 3) have been 0.57-, 0.33- and 0.34-fold of the abundance under Trial 0, respectively (Table 3.7). 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 depressed due to other factors (e.g., water temperatures during the incubation period). Chinook fry abundance in 2019 remained low (relative to Trial 0) and stable (relative to trials 2 and 3) further suggesting 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.

Despite some differences in mean weights among species and age classes under the different flow treatments described in Section 3.4.1 (see Figure 3.17), the trends in biomass among flow treatments generally followed those based on abundance (Table 3.6b, Figure 3.21 and Figure 3.22). 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 years and in 2019 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 Trial 3 and in 2019 (except for Chinook fry in 2019), and was most notable for mykiss fry and parr since the increases in mean weight were more substantial for this species. As such, the biomass estimates in 2019 were 50%, 44%, 79% and 35% of the Trial 2 estimates for mykiss fry, mykiss parr, chinook fry and coho fry, respectively (compared to 32%, 25%, 92% and 31%, respectively, for abundance).

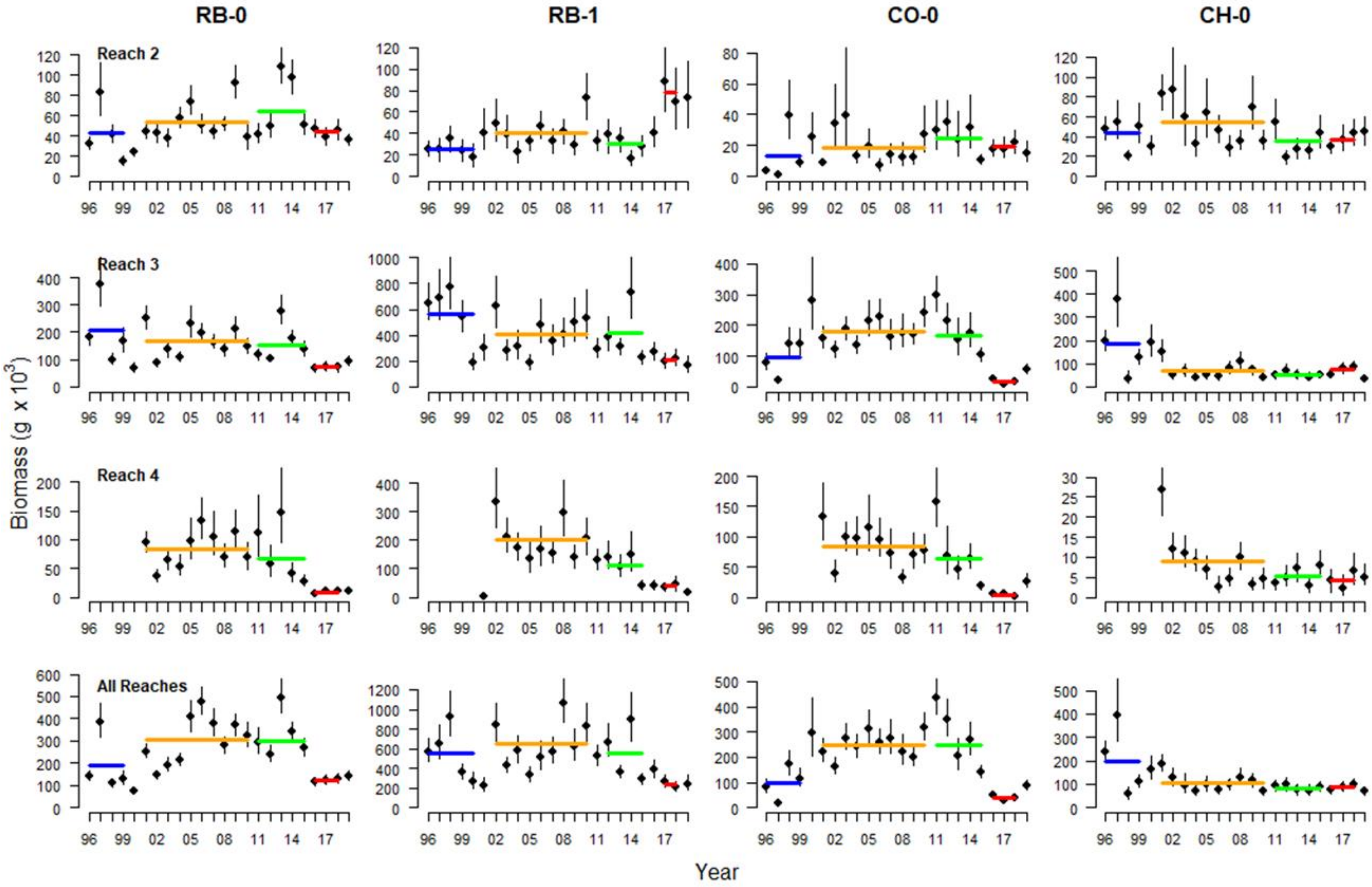


Figure 3.21 **Biomass (in thousands of grams or kilograms) of juvenile salmonids in the Lower Bridge River by reach (row) and species-age class (column). See caption for Figure 3.20 for details.**

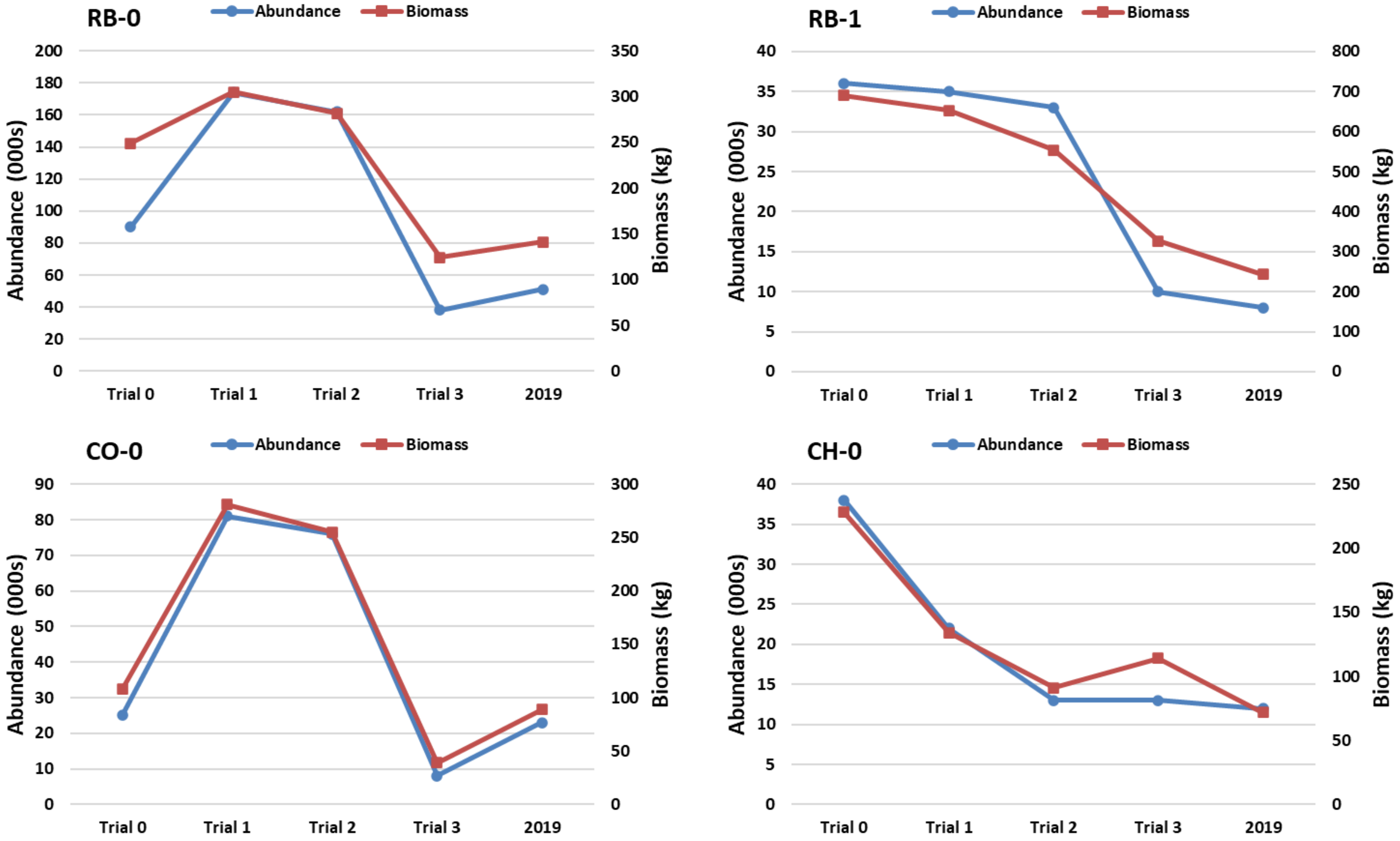


Figure 3.22 Total abundance (in thousands; 1° y-axis) and biomass (in kg; 2° y-axis) estimates for each species and age class to show relative trends in these parameters among the various flow trials and in 2019. Slight differences in biomass versus abundance trends are due to changes in mean weights among flow trials and years.

Under a return to lower flows (based on the Trial 2 hydrograph) in 2019, juvenile salmonid numbers increased overall; however, this change was reflected by some species and age classes but not all (Figure 3.20 and Figure 3.22). Mykiss fry abundance increased by 10,000–18,000 more fish than the individual high flow years up to a total estimate of ~51,000 in 2019. The mykiss parr estimate dropped slightly from 9,000–11,000 fish during the high flow years (2017,2018) to 8,000 fish in 2019. However, as explained above, this cohort had been recruited in a high flow year (2018) when mykiss fry production was reduced so it does not reflect a true post-high flow abundance in 2019. Coho fry abundance increased by 13,000–16,000 more fish than the high flow years up to a total estimate of ~23,000 in 2019. Chinook fry abundance was 12,000 in 2019 which was on par with the 11,000 to 14,000 estimates for this species since 2010 (i.e., across all Trial 2 and Trial 3 years).

For the species and age classes that increased, the change was largely attributable to Reach 3 (Figure 3.20). For mykiss fry, ~11,000–15,000 more fish (or ~97% of the 2019 increase) were estimated for Reach 3 and for coho fry, ~9,000–13,000 more fish (or ~72% of the 2019 increase) were estimated for Reach 3. Coho fry also increased by ~4,000 fish in Reach 4 (~28% of the 2019 increase). Contributions by reach for mykiss parr and chinook fry were generally equivalent to the distribution during the high flow years.

Reach 1 Results

Analysis of sampling data from Reach 1 yielded abundance results by species/age class of approx. 17,000, 1,000, 10,000, and 4,000 for mykiss fry, mykiss parr, coho fry and chinook fry for that reach, respectively (Figure 3.23 Top). Total for all species in Reach 1 was ~32,000 fish, which represented 1/3 of the total fish production (i.e., ~94,000) in reaches 2, 3 and 4 in 2019; 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 were the lowest in Reach 1 for each species and age class among all the reaches (Figure 3.23 Bottom).

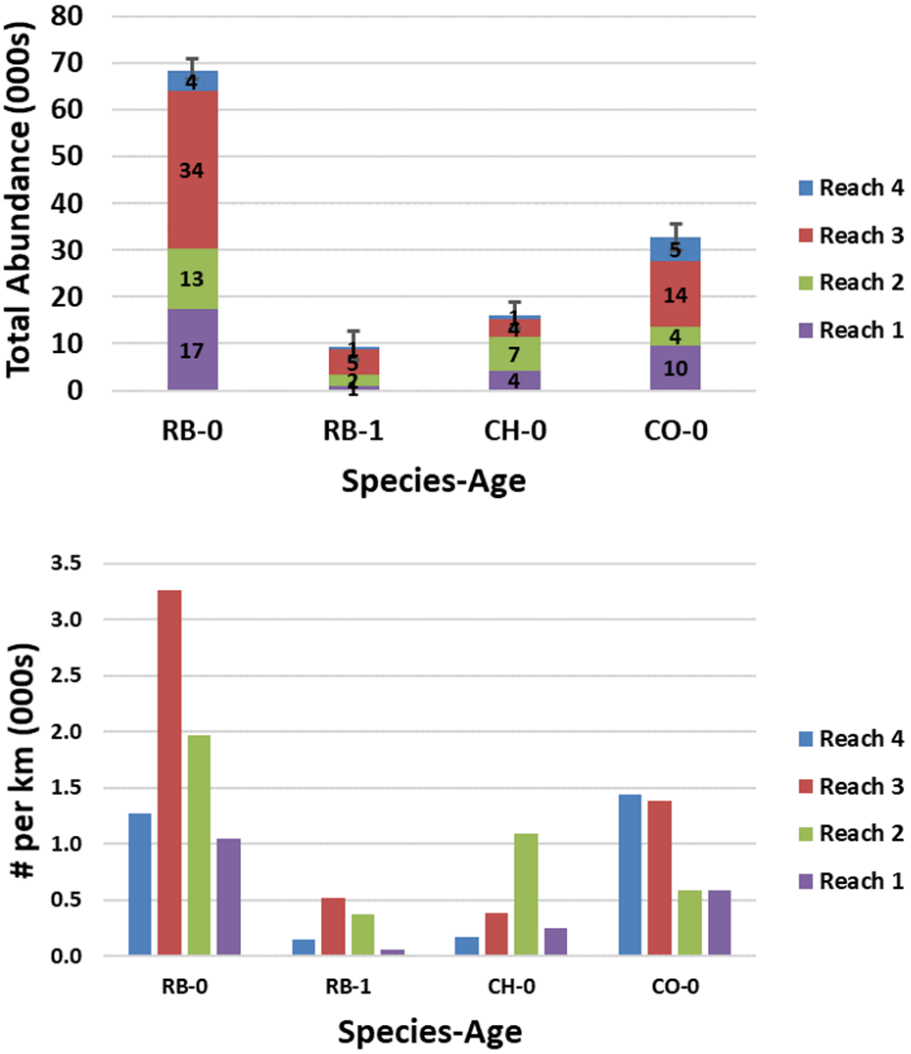


Figure 3.23 Abundance (in thousands) of mykiss fry (RB-0), mykiss parr (RB-1), chinook fry (CH-0) , and coho fry (CO-0) among reaches of the Lower Bridge River in 2019 (Top), and lineal densities (# of fish per km) by species and reach in 2019 (Bottom). Vertical lines in the top plot show 90% credible intervals from posterior distributions of abundance for each year from the hierarchical Bayesian model.

Enhanced Off-channel Habitats (2018 and 2019)

In both 2018 and 2019, the Bluenose off-channel habitat located in Reach 4 had very low catches of coho fry (CO-0) and no chinook fry (CH-0) were caught, but did have moderate densities of mykiss fry and parr (RB-0 and RB-1; Table 3.8). Mykiss fry were less abundant in this channel in 2019 than they were in 2018. The Applesprings site in Reach 1 had high catches of coho fry in 2018 but not in 2019, and only a few chinook fry and mykiss fry were sampled in this habitat in both years. The Bluenose site had higher densities of mykiss (fry and parr) compared to the mainstem in fall of 2018 and 2019, while the Applesprings site had higher densities of coho fry compared to densities in the mainstem of Reach 2 in 2018 and 2019 (Figure 3.24 top and middle panels). The densities in the off-channel sites in 2018 were similar to average densities in their associated reaches during trials 1 and 2 (Figure 3.24 bottom panel). The densities in enhanced sites in 2019 were less than the average densities in their associated reaches during trials 1 and 2.

Table 3.8 Catch, capture probability (pCap), abundance, density, and biomass of juvenile salmonids at enhanced side channel sites in fall of 2018 (a) and 2019 (b). Note: the suffix "RI" and "PO" in the site names denotes riffle and pool habitats, respectively.

a) 2018

Sp-Age	Site	Catch	pCap (per pass)	Abundance	Density (#/100 m^2)	Biomass (g/100 m^2)
CO-0	Bluenose_RI	0		0		
	Bluenose_PO	1	0.17	2	3	12
	Applesprings_RI	32	0.81	32	50	250
	Applesprings_PO	38	0.59	39	53	221
CH-0	Bluenose_RI	0				
	Bluenose_PO	0				
	Applesprings_RI	1	0.20	2	3	15
	Applesprings_PO	2	0.33	3	4	20
RB-0	Bluenose_RI	26	0.80	26	108	203
	Bluenose_PO	73	0.73	74	100	207
	Applesprings_RI	3	0.41	4	6	25
	Applesprings_PO	0				
RB-1	Bluenose_RI	10	0.68	11	45	534
	Bluenose_PO	7	0.64	8	11	86
	Applesprings_RI	2	0.33	3	5	78
	Applesprings_PO	0				

Table 3.8 Continued.

b) 2019

Sp-Age	Site	Catch	pCap (per pass)	Abundance	Density (#/100 m²)	Biomass (g/100 m²)
CO-0	Bluenose_RI	0				
	Bluenose_PO	10	0.68	11	14	72
	Applesprings_RI	7	0.54	8	12	92
	Applesprings_PO	1	0.20	2	3	15
CH-0	Bluenose_RI	0				
	Bluenose_PO	0				
	Applesprings_RI	2	0.33	3	5	47
	Applesprings_PO	0				
RB-0	Bluenose_RI	15	0.52	16	40	213
	Bluenose_PO	20	0.74	20	27	156
	Applesprings_RI	7	0.54	8	12	55
	Applesprings_PO	0				
RB-1	Bluenose_RI	6	0.50	7	17	301
	Bluenose_PO	10	0.52	11	15	194
	Applesprings_RI	0				
	Applesprings_PO	0				

These results demonstrated the use of these two off-channel sites by juvenile salmonids following the period of high flows in 2018, and that they likely function as refuge habitats when rearing conditions in the mainstem are poor. The sum of abundance estimates for juvenile salmonids in 2018 at Bluenose (~1,500) and Applesprings (~5,000) was ~6,500 fish, which was ~10% of the number of fish in the mainstem across reaches 2 – 4 in 2018 (63,000). This was a fair contribution considering the difference in wetted area (i.e., ~1 ha for the off-channel habitats vs. ~50 ha for reaches 2, 3 and 4). However, use of the off-channel habitats was reduced in 2019 when flows in the LBR were lower (i.e., Trial 2 hydrograph) and mainstem habitats were presumably more suitable than under high flows. The sum of abundance estimates for juvenile salmonids in 2019 was 1,100 fish at Bluenose (658) and Applesprings (452), compared to 94,000 fish across reaches 2 – 4. Thus, in 2019, enhanced side channel sites provided only ~1% of the total production from the LBR.

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. Fish densities at these sites in 2018 represented use when the mainstem LBR had been subjected to high flows and could be useful for estimating the area required to maintain fish populations under those conditions. For example, Coho abundance in 2018 was ~69,000 fewer fish than in 2011-2015 (76,000 vs. 7,000). Based on the maximum density of coho observed in the Applesprings and Bluenose sites in 2018 (53 fish/100 m^2 in Applesprings pond habitat), a total area of 130,000 m^2 would be required (equivalent to 15 Applesprings sites). The same calculation for mykiss fry (maximum density 108 fish/100 m^2 in Bluenose riffle habitat) yielded an area of 119,000 m^2 (or approximately 103 Bluenose-sized sites) to offset mainstem losses. These rough estimates could also be considered conservative because the maximum site-specific densities applied were not representative of fish densities in these offchannel habitats as a whole.

Note: the stock assessment sampling in both the mainstem and offchannel habitats is conducted in the fall (i.e., September) to document densities during the post-freshet rearing period. However, the relative use of these areas during other seasonal periods (e.g., freshet) is less understood because it has not been quantified by comparable methods.

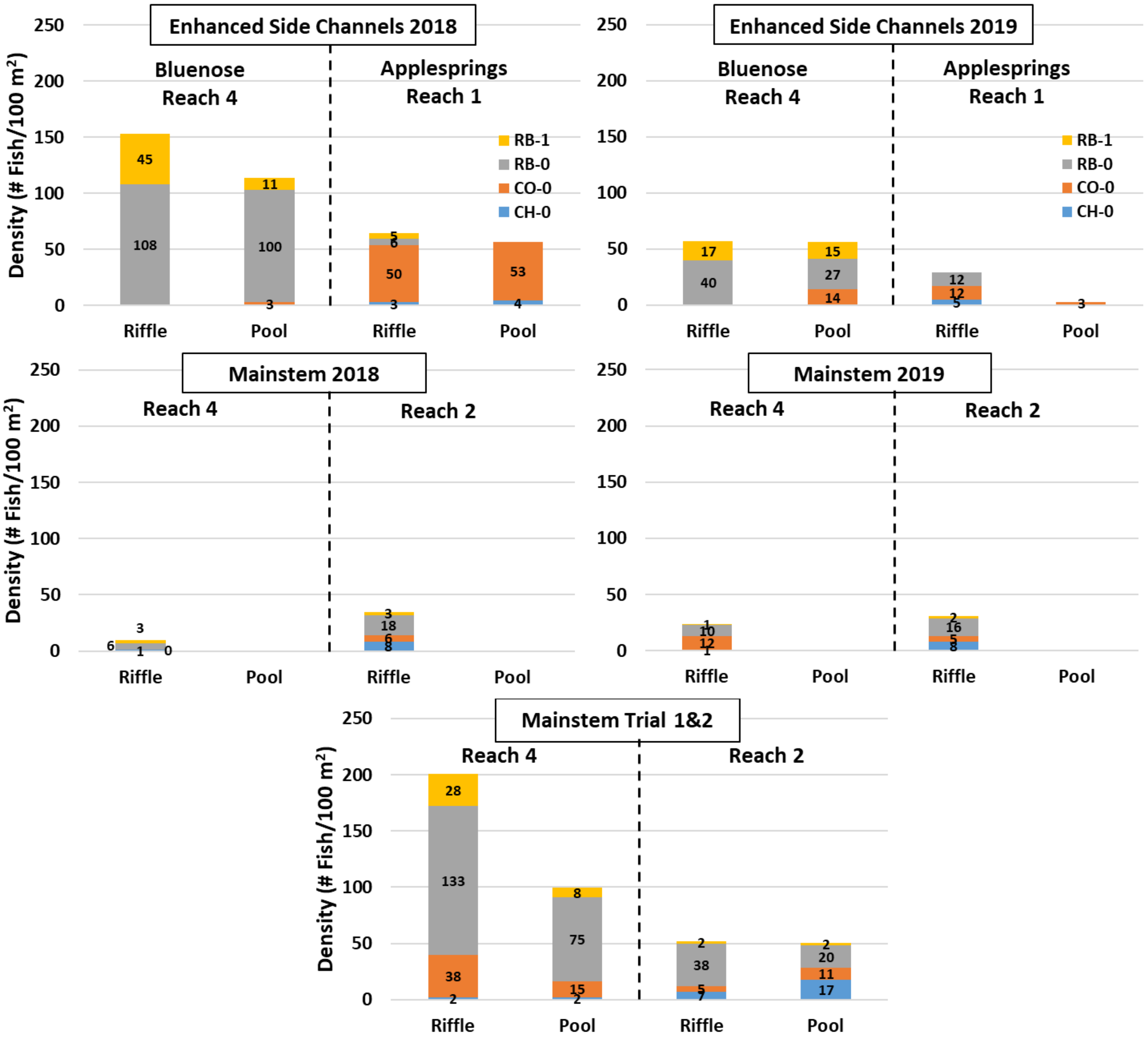


Figure 3.24 Density of juvenile salmonids in enhanced side channels in fall of 2018 (top left) and 2019 (top right) by habitat type compared to mainstem densities during the same time period (middle left and right) and average values in the mainstem prior to 2016 (bottom, trials 1 & 2).

3.4.3. Stock-Recruitment

As noted in previous annual reports, 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. The newest data point reflecting the 2018 spawner escapement and 2019 fry abundance estimates for coho and chinook was not applied to any of the existing curves since it reflects the first year of post-high flow conditions (considered a new treatment). As such it is shown as a stand-alone point on the plots and we simply compare it to the other existing curves for this analysis.

The maximum likelihood estimate for coho spawner escapement in 2018 was 1,245 (95% confidence intervals: 882–1,627) for reaches 3 and 4 (White et al. 2019). This was the highest AUC-based estimate for coho since the Trial 2 years (range: 410–3,563). It was paired with the 2019 coho fry abundance estimate of 23,190 (90% credible intervals: 18,680–29,230). The maximum likelihood estimate for chinook spawners in 2018 was 25 (95% confidence intervals: 14–44) for reaches 3 and 4 (White et al. 2019). This was the lowest estimate since 2009 (*n*= 21), but was only based on chinook spawners observed prior to September 1st, 2018 due to installation of a broodstock collection fence at the bottom of Reach 3 on that date. As such, this must be viewed as a minimum (and highly tenuous) run size estimate. The 2018 escapement estimate was paired with the 2019 chinook fry estimate of 11,670 (90% credible intervals: 8,880–14,600).

It must be noted that the reliability of the stock-recruit analysis for estimating production capacity within the study area and determining when stock size may be limiting juvenile recruitment for each flow treatment is wholly dependent on the quality of both the stock and recruit estimates. If the installation of the broodstock collection fence will continue to preclude the collection of reliable spawner escapement data, as was the case in 2018, then we will not have reliable data for updating the stock-recruitment curves going forward. If the operation of the fence can be compatible with the MON-3 spawner escapement monitoring (i.e., through effective collaboration between these programs) such that reliable estimates are still possible then these curves can continue to be updated.

Coho fry abundance increased under the Trial 1 and Trial 2 treatments relative to the Trial 0 pre-flow period (Figure 3.25). The magnitude in the shift of the stock-recruitment curve for coho fry (e^{λ}) was 2.9, 2.3, and 0.35 for Trial 1, Trial 2, and the high flow (Trial 3) 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. The 2019 datapoint for coho was well beyond the initial slope and within the range of Trial 2 estimates for spawner stock size but was below any of the Trial 2 estimates for juvenile recruitment and much closer to the curves for Trial 0 and Trial 3. This suggests that spawner escapement was not the limiting factor for juvenile recruitment in 2019 and suggests that other factors (such as in-river habitat conditions or food source availability) were more significant.

In fact, there has been no indication from the shape of the various stock-recruitment curves in Figure 3.25 that coho fry production has been limited by escapement (i.e., under-seeded) in any of the study years to-date as almost all data points are near or on the asymptote of the curves. For example, escapements in 2005 and 2009 were low and close to the origin but fry production associated with these escapements was high. These points 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 hit the boundary of our maximum assumed value (1500 egg/female x 0.5 females/total spawners x 0.5 egg-fry survival rate = 375 fry/spawner) and would be unrealistically steep if we had not constrained this parameter.

Assuming a lower maximum initial slope (e.g. 37.5 fry/spawner based on a 0.05 egg-fry survival rate) constrains the curves to a much greater extent (Figure 3.26). In this case, there are many data points that have escapements that are less than required to maximize fry production. These more constrained curves paint a different picture yet provide a near equivalent fit to the data. The difference in log-likelihood measuring the fit of the curves in Figure 3.25 and Figure 3.26 is less than 2 units and therefore one set of curves cannot be considered more reliable than the other based on fit alone. The stock-recruitment curve in Figure 3.26 implies that the population has been under-seeded. In this case poor fry production under recent high flow years could be partially attributed to low escapement (as it would be for several Trial 1 and 2 years also). More data are required (i.e., at low escapement levels) to better define the initial slope of the stock-recruitment relationship to strengthen inferences about potential spawning stock limitation and determine which set of curves are a more reliable indicator of flow effects on coho fry production in the LBR.

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 0.50 egg-fry survival rate, Figure 3.27). The stock-recruitment λ values indicate that recruitment under the Trial 1 and Trial 2 flow treatments and during the high flow (Trial 3) years were 0.74-fold, 0.46-fold, and 0.44-fold lower than under the pre-flow conditions (Figure 3.27). Owing to the steep initial slope there is no indication that escapement has been limiting fry abundance, and the 2019 datapoint was within a cluster of datapoints from the previous flow trials that were near the origin on the x-axis (e.g., brood years 2008, 2009, 2011, 2013 and 2017). 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 0.05 (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 3.28). Yet, again, this more constrained curve provides a near equivalent fit to the data (the likelihood difference between fits is less than 2 units).

Given the current 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 data are not yet 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*= 19 data points for coho and *n*= 21 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 (Section **Error! Reference source not found.**). 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. 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 has been a less significant driver than flow treatment effects on juvenile production for these species.

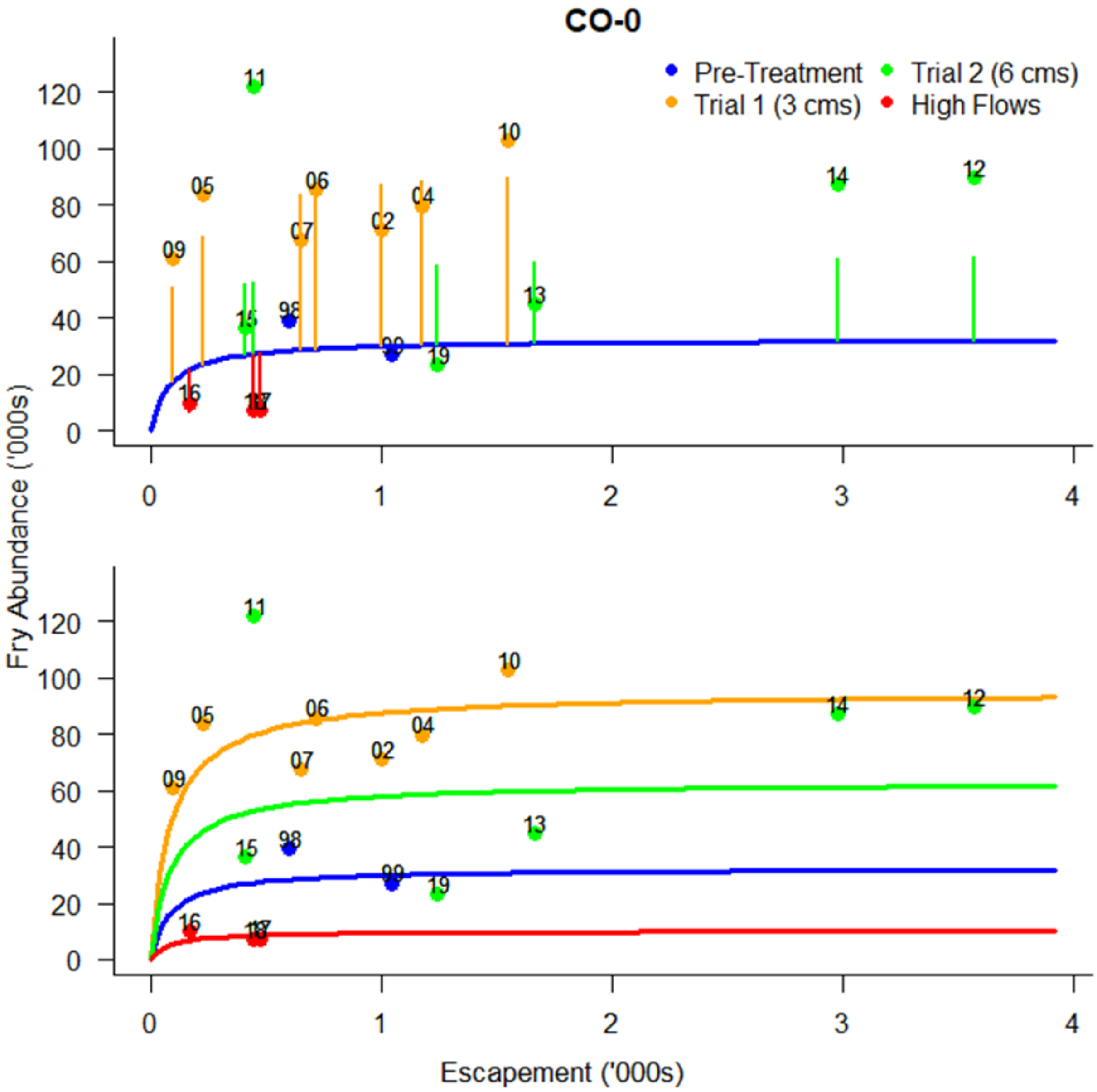


Figure 3.25 **Spawner-fry coho Beverton-Holt stock-recruitment curves fit assuming a maximum initial slope of 375 fry/spawner (50% egg-fry survival rate). Points show annual estimates of escapement and fry abundance with the label beside each point showing the recruitment year. The blue line in the top plot shows the base stock-recruitment curve under pre-flow conditions (Trial 0). The vertical lines in the top plot show the shift of the base stock-recruitment curve for the other three flow treatments. The bottom plot shows the treatment-specific stock-recruitment curves (e.g. the curve that results from drawing a line through the ends of the vertical lines in the top plot).**

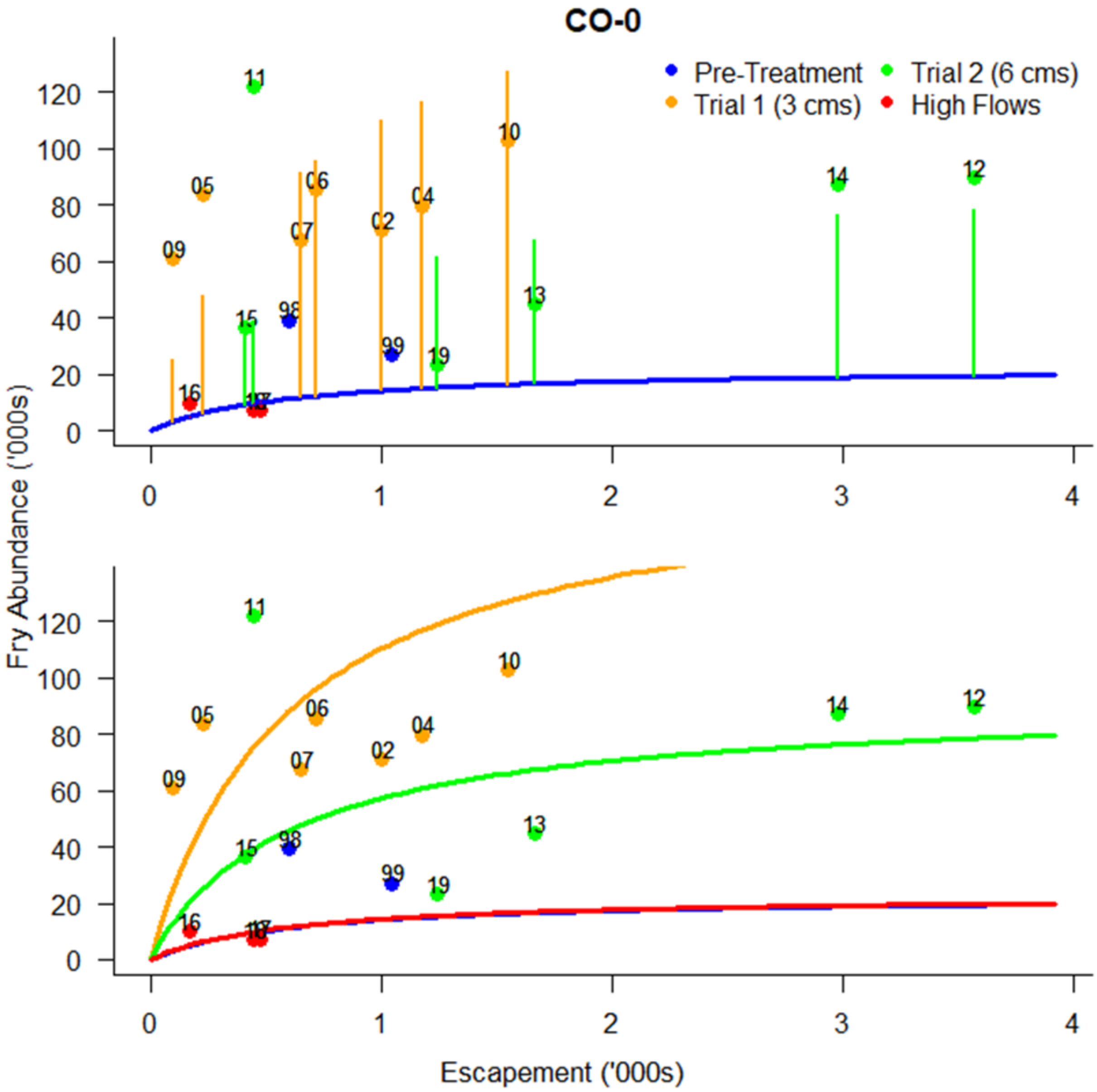


Figure 3.26 **Spawner-fry coho Beverton-Holt stock-recruitment curves fit assuming a maximum initial slope of 37.5 fry/spawner (5% egg-fry survival rate). See caption for Figure 3.25 for additional details.**

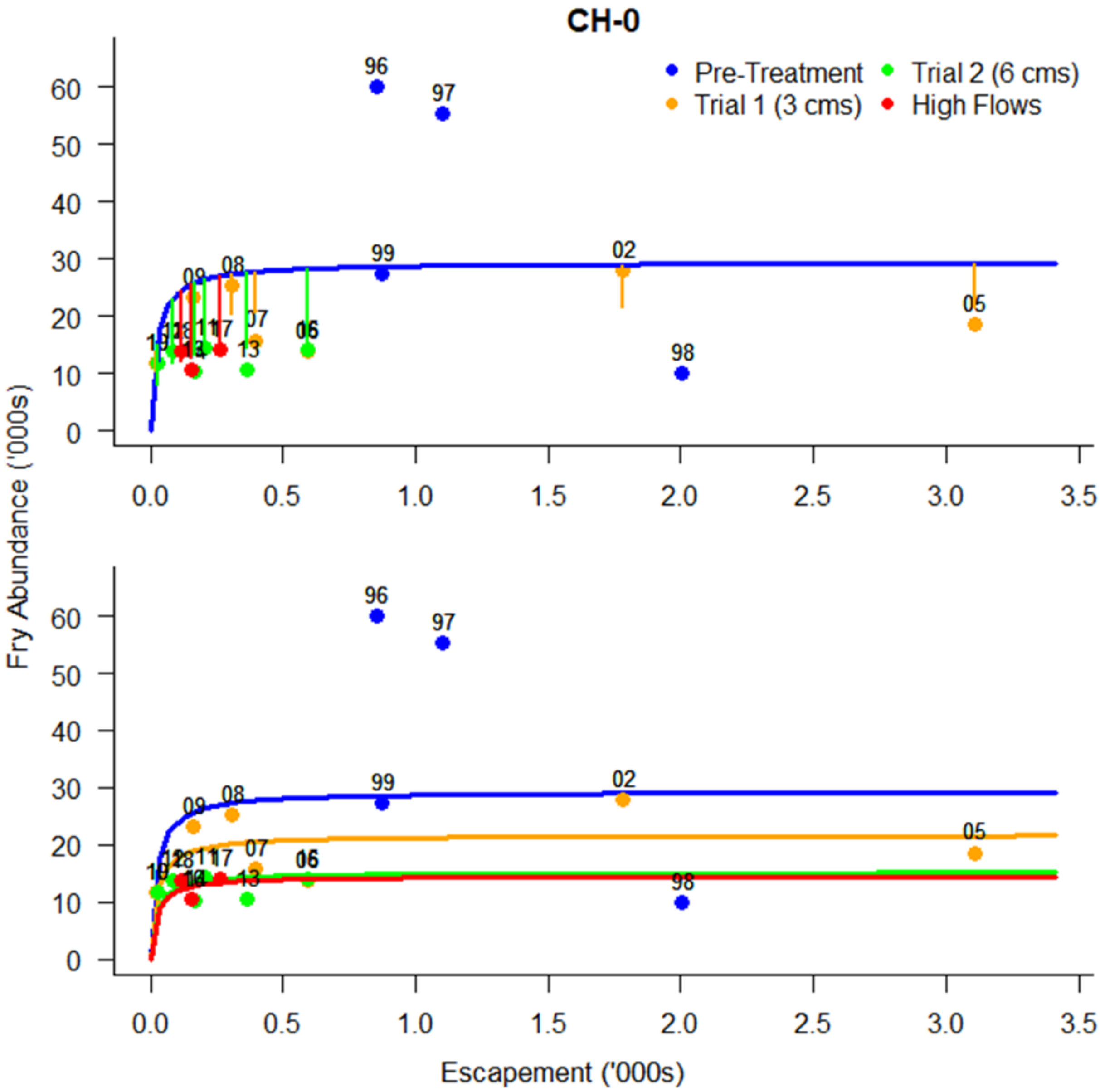


Figure 3.27 **Spawner-fry chinook Beverton-Holt stock-recruitment curves fit with a constraint that assumes a maximum egg-fry survival rate of 50% (maximum initial slope of 1250 fry/spawner). See caption for Figure 3.25 for details.**

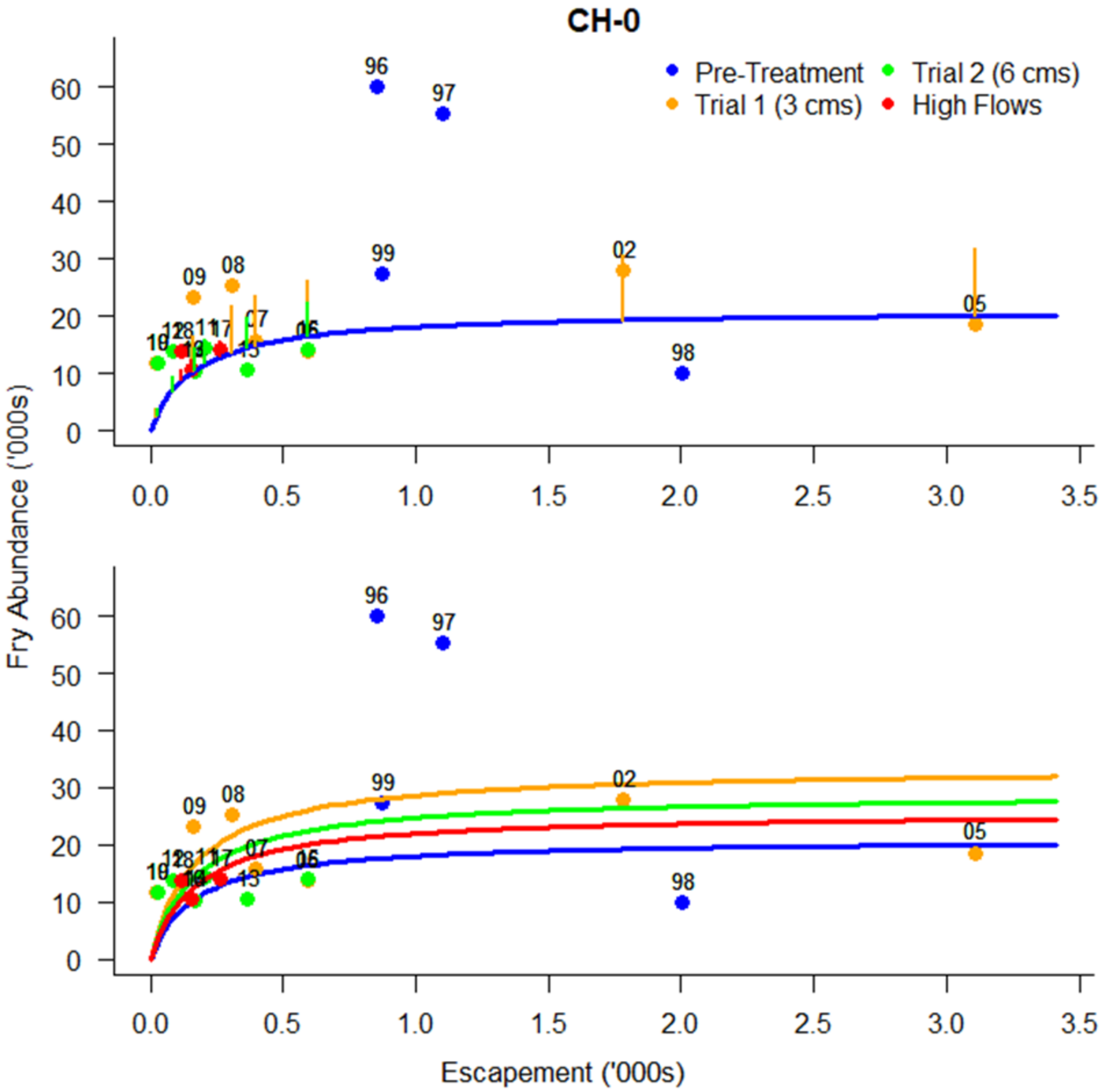


Figure 3.28 **Spawner-fry chinook Beverton-Holt stock-recruitment curves fit with a constraint that assumes a maximum egg-fry survival rate of 5% (maximum initial slope of 125 fry/spawner). See caption for Figure 3.25 for details.**

3.4.4. 2019 Hatchery Chinook Releases

A total of 3,189 hatchery-reared chinook fry were released into the Lower Bridge River in Reach 4, a short distance below Terzaghi Dam (~Rkm 40.9) on 20 August, 2019. During stock assessment sampling approximately two weeks later (29 August to 18 September 2019), 14 marked chinook fry were recaptured. There was evidence for some degree of dispersal from the release site at the time of the recapture events. Eleven marked fish were recaptured within Reach 4 (79% of recaptures; 0.8 to 3.9 km below the release site), two marked fish were recaptured in Reach 3 (14% of recaptures; 5.0 to 7.1 km below the release site), and one marked fish was recaptured in Reach 2 (7% of recaptures; 19.1 km below the release site). However, for the following analyses we had to assume site closure (i.e., that marked fish remained within the study area), which was at least partially supported by the high proportion of recaptures within reaches 3 and 4.

The release of hatchery-produced age-0 chinook in 2019 provided an opportunity to 1) estimate their survival rate from release to the stock assessment, or 2) check the estimated capture probability for naturally produced juveniles during the stock assessment. Survival rate was estimated assuming the capture probability of naturally- and hatchery-produced juveniles are the same and therefore allows an estimate of survival between time of release and the stock assessment. Capture probability was estimated assuming 100% survival between the release and the stock assessment to determine the capture probability of hatchery-produced fish, which can be compared to the estimated value for naturally-produced fish as determined by the stock assessment data and the Bayesian model (Appendix B). Both methods provide an estimate of hatchery-produced abundance at the time of the stock assessment that in turn can be used to determine the contribution of hatchery fish relative to the size of the natural population at the time of the fall stock assessment.

Method 1 – Estimate based on variable survival: 3,189 hatchery-produced age-0 chinook salmon were released into the plunge pool on 20 August 2019. 14 of these fish were recaptured during the fall stock assessment approximately 3 weeks later (Table 3.9). If we assume that the overall capture probability of these hatchery-produced fish is the same as naturally produced juveniles, the abundance of hatchery fish during the stock assessment can be determined. To do this, we divided the estimated total abundance of naturally produced chinook fry across reaches 2–4 (11,700) by the catch (128). The resulting abundance/catch ratio for naturally produced fish (An/Cn) of 91 indicates that the capture of 1 chinook fry represents 91 fish present. The overall reach 2–4 capture probability is simply the inverse of 91 or 0.011 (1.1%). Multiplying the An/Cn ratio by the number of hatchery fish that were caught (Cn) results in an estimate of 1,280 present during the stock assessment survey (i.e., Ah = Ch * An/Cn). If the assumption of equal capture probability is correct, hatchery additions therefore increased the abundance of the naturally produced chinook juveniles at the time of stock assessment by ~11%. The estimated survival rate between release and the stock assessment is 40% given a total release of 3,189 and the estimate of 1,280 during the stock assessment.

Method 2 – Estimate based on variable capture probability: Here we assume that survival of hatchery fish between release and stock assessment is 100%. Thus 3,189 hatchery chinook were present during the stock assessment (Ah). A total of 14 hatchery fish were caught during the stock assessment (Ch), equivalent to a reach 2–4 capture probability of 0.44% (Ch/Ah; Table 3.9). The capture probability for naturally produced juvenile chinook was 1.1% which is 2.5-fold higher than for hatchery-produced fish. If all hatchery fish survived between release to the stock assessment, hatchery contributions increased the abundance of juvenile chinook by 27% (100*3,189/11,700).

Of these two estimates, it is more likely that the survival rate of released hatchery Chinook was ~40% and that the contribution of hatchery fish to the population was ~11% (as suggested by Method 1). Large post-release mortality for hatchery-produced fish is expected (Melnychuk et al. 2014). In addition, the difference in estimated capture probability between naturally- and hatchery-produced juvenile chinook (as suggested by Method 2) is difficult to explain given that they were of similar size when compared and were found in similar habitats.

Table 3.9 Estimates of the contribution of hatchery-produced chinook fry to the natural population at the time of the fall standing stock assessment in 2019. Method 1 assumes equal capture probability for naturally- and hatchery-produced juveniles and therefore estimates the abundance of hatchery-produced juveniles at the time of the stock assessment. Method 2 assumes 100% survival between release and the stock assessment, and therefore allows capture probabilities for hatchery- and naturally produced juveniles to vary. Cells highlighted in grey denote that numbers are derived from calculations.

Method 1: Estimate based on variable survival
Assumption: Equal Capture Probability of Wild/Hatchery Fish

	Natural	**Hatchery**
Catch (all reaches)	128	14
Abundance (all reaches)	11,700	1,280
Abundance/Catch (A/C)	91	
Hatchery chinook released		3,189
Survival from release to stock assessment		40%
Increase in CH0 abundance from hatchery fish at stock assessment		10.9%

Method 2: Estimate based on variable capture probability
Assumption: 100% Survival of hatchery fish since release

	Natural	**Hatchery**
Catch (all reaches)	128	14
Abundance (all reaches)	11,700	3,189
Abundance/Catch (A/C)	91	228
Increase in CH0 abundance from hatchery fish at stock assessment		27.26%

3.5. WUP Ramp Down Monitoring and Fish Salvage

In the tables and figures throughout this section, comparable ramping information from the 2016 to 2018 high flows 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 2019 results, for reference.

Ramp downs from peak flows (i.e., between 15.2 and 3.0 $m^3 \cdot s^{-1}$) occurred on eight days between 1 and 19 August 2019, representing a total flow reduction of 12.2 $m^3 \cdot s^{-1}$ across that period (Figure 3.29, Table 3.10 and Table 3.11). 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 on 1 and 2 October 2019. The timing of ramp down dates in 2019 was comparable to the usual timing from the previous Trial 2 years (2011 to 2015). For additional information on flow and stage changes for each rampdown event, refer to the tables in Appendix D. Total stage change at the 36.8 km compliance location was 60 cm, and mean daily stage change rate implemented was 1.3 cm/hr (however, maximum *hourly* stage change rate was 4.6 cm/hr). The implementation of some higher hourly ramp rates in 2017, 2018 and 2019 (compared to past years) meant that the reduction of flows could be completed over a shorter timeframe (i.e., fewer hours of ramping per day).

Based on stage monitoring conducted during ramp events by the field crews and the hourly stage elevation data recorded by loggers deployed in reaches 2, 3 and 4, a gradient of stage change across the length of the study area was evident (Table 3.10). Within the flow range released from the dam in 2019 (max = 15 $m^3 \cdot s^{-1}$; min = 1.5 $m^3 \cdot s^{-1}$), the total stage change was 96 cm at the Terzaghi Dam plunge pool (Reach 4; Rkm 40.9), 60 cm near the Reach3/Reach 4 boundary (a.k.a. the compliance location; Rkm 36.8), 51 cm at the fish fence site (bottom of Reach 3; Rkm 26.1), 15 cm at the horseshoe bend (Reach 2; Rkm 23.6), and 25 cm at Camoo (Reach 2; Rkm 20.0). The gradient was similarly evident on each individual ramping date. The substantial reduction in stage elevation changes in Reach 2, relative to reaches 3 and 4, highlights the moderating effect of the Yalakom inflows on stage changes associated with flow reductions at the dam (within the Trial 2 hydrograph range). This finding supports the understanding that fish stranding risk is lower in reaches 1 and 2 due to the moderated stage changes and that fish salvage at discharges ≤ 15 $m^3 \cdot s^{-1}$ is not required.

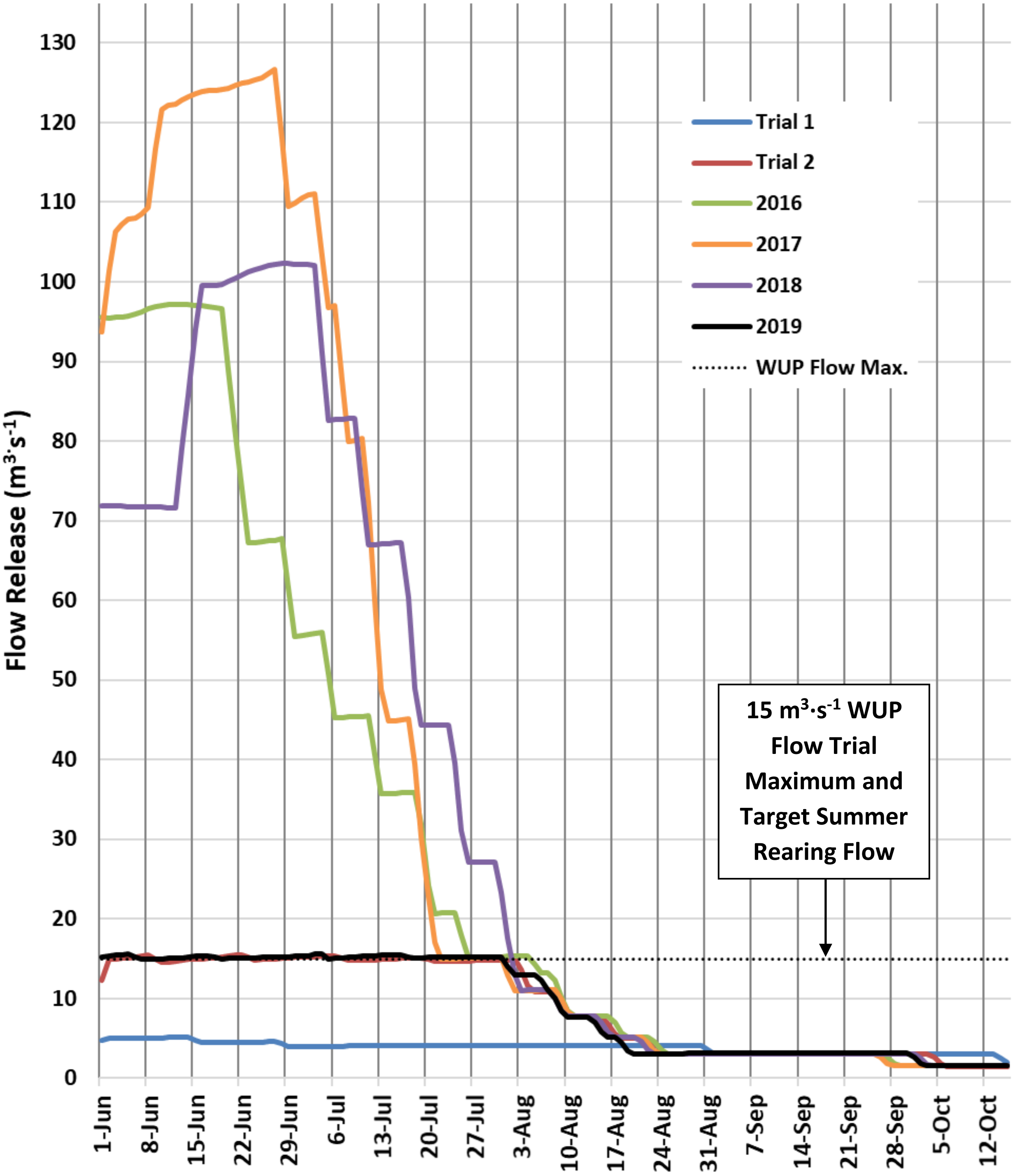


Figure 3.29 Schedule of flow releases and ramp downs from the peak period to the start of the fall low flow period in 2019. For reference, WUP Trial 1 and 2 flow releases as well as the 2016–2018 high flow years are shown for the same period.

Table 3.10 Summary of stage changes at available monitoring locations in reaches 2, 3 and 4 for each rampdown event in 2019.

Year	Date	Event #	Start Flow ($m^3 \cdot s^{-1}$)	End Flow ($m^3 \cdot s^{-1}$)	Flow Change ($m^3 \cdot s^{-1}$)	Stage Change (cm)				
						Plunge Pool (Rkm 40.9)	Top of Reach 3 (Rkm 36.8)[a]	Bottom of Reach 3 (Rkm 26.0)	Horseshoe Bend (Rkm 23.6)	Bottom of Reach 2 (Rkm 20.0)
2019	1 Aug	1	15.2	12.9	-2.3	-11	-6	-5	-3	-4
	6 Aug	2	13.0	11.1	-1.9	-11	-5	-6	-1	-3
	8 Aug	3	11.1	9.3	-1.8	-9	-5	-5	-2	-2
	9 Aug	4	9.3	7.7	-1.6	-9	-5	-5	-1	-3
	14 Aug	5	7.7	6.4	-1.3	-8	-5	-4	-1	-3
	15 Aug	6	6.4	5.1	-1.3	-9	-6	-6	-2	-3
	18 Aug	7	5.1	4.1	-1.1	-10	-5	-5	-1	-2
	19 Aug	8	4.1	3.0	-1.1	-9	-6	-6	-1	-2
	1 Oct	9	3.2	2.1	-1.0	-11	-8	-6	-1	-2
	2 Oct	10	2.1	1.5	-0.6	-10	-6	-5	-1	-1
WUP Rampdown Summary			**15.2**	**1.5**	**-13.7**	**-96**	**-60**	**-51**	**-15**	**-25**

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

Table 3.11 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 (2016–2019). Note: Flow releases did not exceed the WUP flow targets in 2019 so high flow ramp downs were not required that year. For more details on individual events refer to the tables provided in Appendix D.

Period	Year	Month(s)	# of Ramping Days	Total Flow Reduction ($m^3 \cdot s^{-1}$)	Total Stage Change (cm)	Maximum Daily Rate (cm/hr)
High Flow Ramp Events (>15 $m^3 \cdot s^{-1}$)	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 $m^3 \cdot s^{-1}$)	**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

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 3.30).

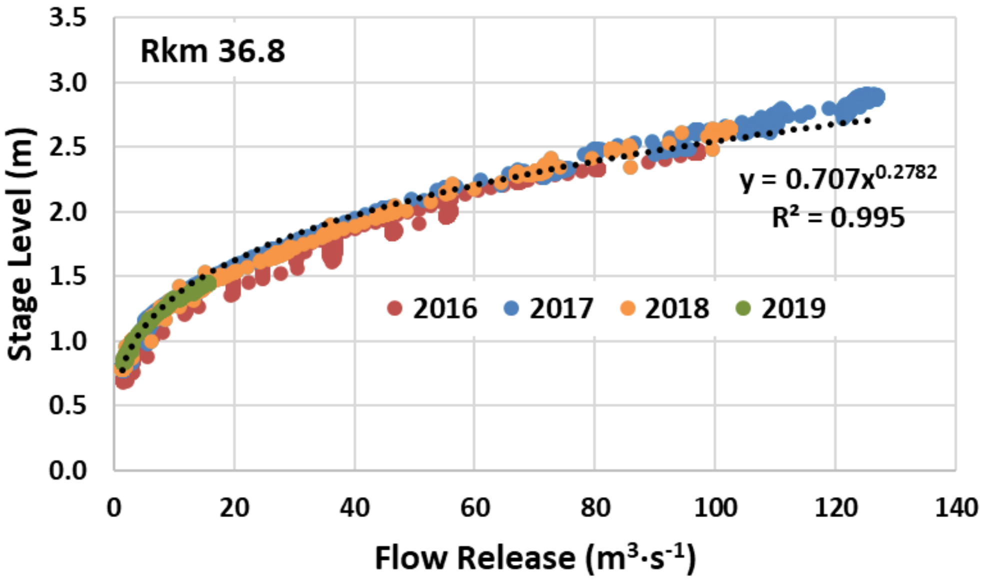
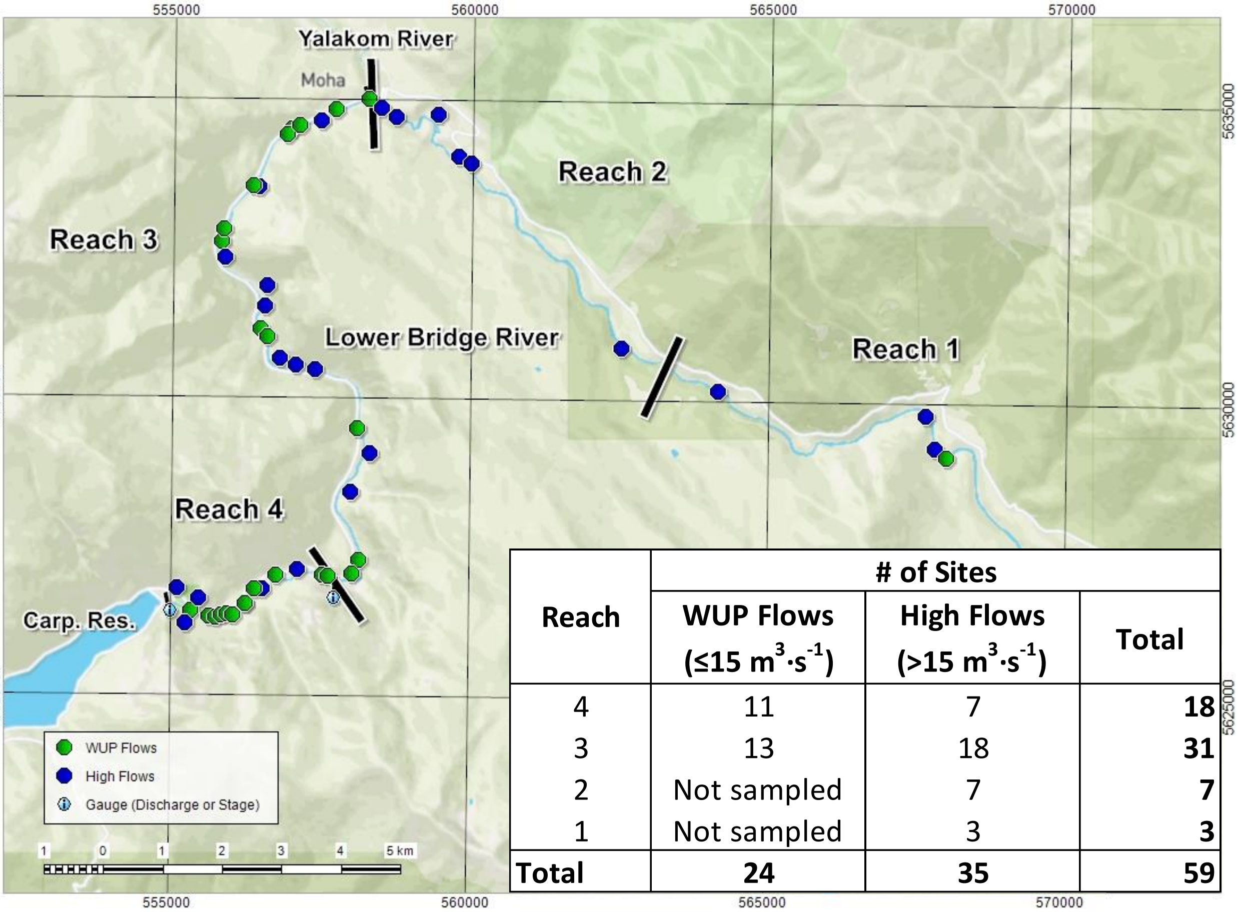


Figure 3.30 Discharge-stage relationship at 36.8 km (the compliance location) across the range of flows observed across all flow treatments. Separate data points for each high flow monitoring year (2016-2018) are shown.

The curve drawn through the points has a good fit ($R^2 = 0.995$), such that the associated equation ($y = 0.707x^{0.2782}$) may be useful for predicting stage changes for particular flow changes within this range. Stage values for discharges between 10 $m^3 \cdot s^{-1}$ and 60 $m^3 \cdot s^{-1}$ tended to be a bit lower in 2016, possibly due to some channel changes at the gauging location that have occurred with the high flows since then, so the current curve is based on the 2017, 2018 and 2019 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 surveys conducted during the 2019 flow ramp downs, the incidence of fish stranding was documented at 4 new locations relative to the 20 locations that had been identified for the 15 to 1.5 $m^3 \cdot s^{-1}$ flow range during the previous Trial 2 and 3 years (2011 to 2018), bringing the total to 24 unique locations (in reaches 3 and 4 only; Figure 3.31). However, active fish salvaging was only required at 18 of the 24 sites in 2019. Crews noted that channel morphology or flow conditions had changed at several locations following the years of high flows, which resulted in the identification of the new sites 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 was not required at these sites in 2019 since flows did not exceed 15 $m^3 \cdot s^{-1}$. Two of the new sites at flows ≤15 $m^3 \cdot s^{-1}$ were in Reach 4, and two were in Reach 3.



Reach	# of Sites		
	WUP Flows ($\leq 15\ m^3 \cdot s^{-1}$)	High Flows ($> 15\ m^3 \cdot s^{-1}$)	Total
4	11	7	**18**
3	13	18	**31**
2	Not sampled	7	**7**
1	Not sampled	3	**3**
Total	**24**	**35**	**59**

Figure 3.31 Survey area map for ramp monitoring and fish salvage on the Lower Bridge River showing fish salvage locations for WUP flows (green dots), including 2019 sites, and Modified Operations high flows (blue dots), which were not sampled in 2019. 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 is also included (inset).

Fish salvage numbers for the ramp downs across the Trial 2 range ($\leq 15\ m^3 \cdot s^{-1}$), including 2019 data points, were generally higher relative to the results for the high flow range ($> 15\ m^3 \cdot s^{-1}$; Figure 3.32). 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 2017–2019 results, these incidental catches were not included in the analyses.

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 $m^3 \cdot s^{-1}$, which was again apparent from the 2019 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 (see Section 3.4.2). Relative to the Trial 2 averages, abundance of coho and steelhead fry was down by 90% and 70%, respectively, during the high flow years. 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.

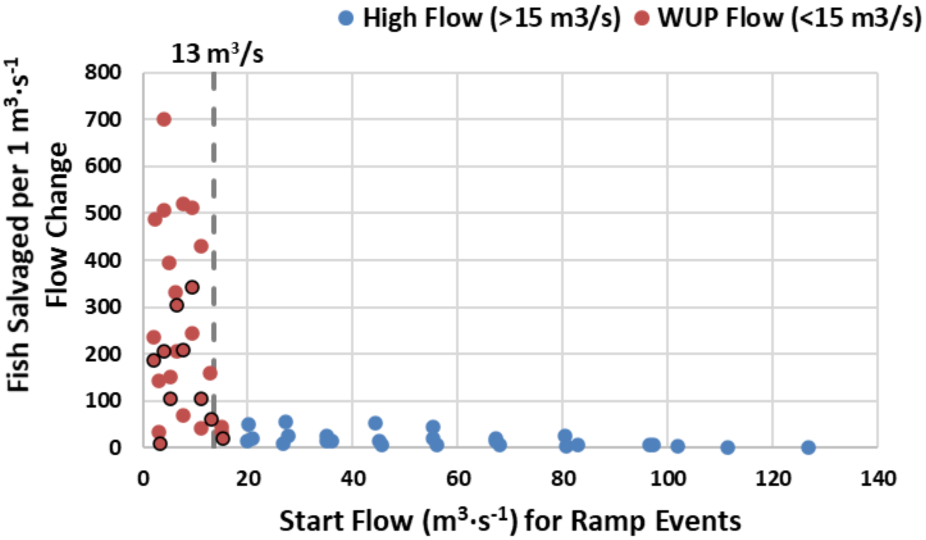


Figure 3.32 Relative differences in number of fish salvaged per increment of flow change for ramp downs from high flows (>15 $m^3 \cdot s^{-1}$) versus Trial 1 and 2 flows (≤15 $m^3 \cdot s^{-1}$). The vertical dashed line represents the approximate flow threshold (~13 $m^3 \cdot s^{-1}$) where the apparent break between high stranding risk and moderate or low stranding risk occurs. Note: values do not include incidental catches. Circles with black border represent 2019 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), primarily due to the addition of stranding site reconnaissance and salvage surveys in reaches 2 and 1 (Table 3.12). Note: only the most recent high flow year is presented for comparison because stranding area changed to some degree with each consecutive high flow event. 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. Under the trial flow range (≤15 $m^3 \cdot s^{-1}$) in 2019 when only reaches 3 and 4 were surveyed, the total stranding area was 9,021 m^2 (compared to 13,992 m^2 for those two reaches at the high flow range), and stranding area was again more prevalent in Reach 3 than Reach 4 (7,163 and 1,858 m^2, respectively).

Across the high flow range (>15 $m^3 \cdot s^{-1}$) in 2018, the highest proportion of salvaged fish per stranding habitat area was in Reach 3 (~8 fish per 100 m^2; Table 3.12). The values for the other reaches were relatively small (≤3 fish per 100 m^2). Within the Trial 2 flow range (≤15 $m^3 \cdot s^{-1}$) in 2019, fish stranding densities were greater, and the highest proportion was in Reach 4 followed by Reach 3 (30 and 20 fish per 100 m^2, respectively). These values were lower than those 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 trial flow range.

Table 3.12 Summary of fish stranding area and numbers of fish salvaged by reach for 2018 high flow (>15 $m^3 \cdot s^{-1}$) and 2019 trial flow (≤15 $m^3 \cdot s^{-1}$) ranges. Note: there was no data for fish stranding or salvage in reaches 1 and 2 under the trial flows.

Flow Range	Reach	# of Sites	Area (m^2) (% Contribution)	# of Fish	# of Fish per 100 m^2
2018 High Flows (>15 $m^3 \cdot s^{-1}$)	4	7	4,887 (7%)	125	3
	3	11	9,105 (14%)	710	8
	2	4	22,900 (34%)	551	2
	1	3	30,000 (45%)	413	1
High Flow Totals		**25**	**66,892**	**1,652**	**3**
2019 Trial Flows (≤15 $m^3 \cdot s^{-1}$)	4	6	1,858 (21%)	558	30
	3	12	7,163 (79%)	1,463	20
	2	------ No data ------			
	1	------ No data ------			
Trial Flow Totals		**18**	**9,021**	**2,021**	**22**

With the benefit of fish salvage crews on the ground, some higher ramp rates (up to 4.6 cm/hr) were implemented again in 2019 (as in 2017 and 2018). In the past, most ramp rates conformed to the ≤2.5 cm/hr threshold specified in the Water Use Plan (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 and 2018, 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 3.33). 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 in 2019 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 – 2019 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.

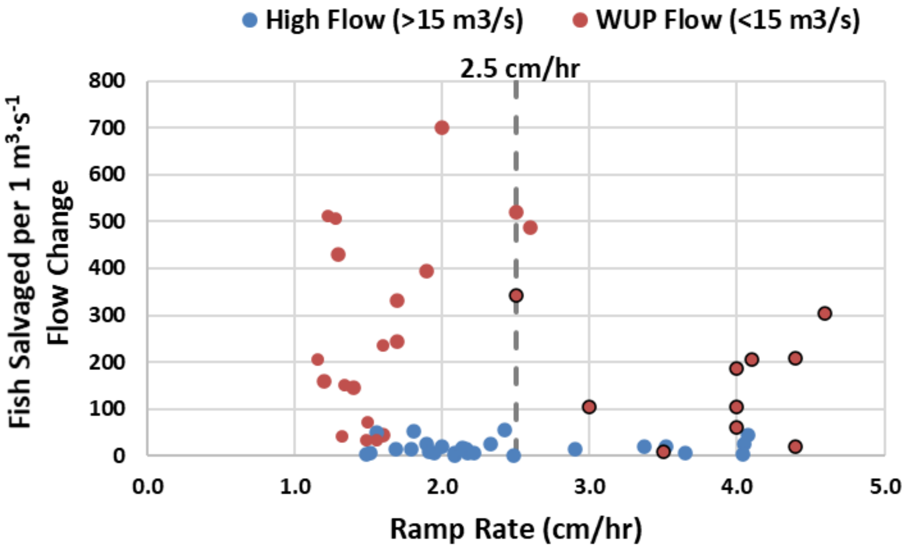


Figure 3.33 Relative incidence of fish stranding per increment of flow change according to different ramping rates under high flow (>15 $m^3 \cdot s^{-1}$; blue circles) and trial flow (≤15 $m^3 \cdot s^{-1}$; red circles) ranges. The vertical dashed line depicts the ramp rate (2.5 cm/hr) specified in the WUP when fish salvage crews are not present. Circles with black border represent 2019 data; Plain circles are data from past years.

Inclusive of the new sites identified in 2019, the proportions of identified stranding sites on river left (79%) versus river right (21%) 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 3.13). 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 45% on river left and 55% on river right based on the new site reconnaissance conducted by staff from Coldstream Ecology Ltd. and Xwísten during 2016 – 2018.

Table 3.13 Proportions of sites on the river left bank versus the river right bank for trial 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 high flow years).

Flow Range	Left Bank		Right Bank	
	n	%	n	%
Trial Flows (≤15 $m^3 \cdot s^{-1}$) *Reaches 3 & 4 only	19	79%	5	21%
High Flows (>15 $m^3 \cdot s^{-1}$) *New Site Reconn.	8	45%	10	55%
All	**27**	**64%**	**15**	**36%**

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 under high flows (Sneep et al. 2018; Sneep et al. 2019), and again under Trial 2 flows in 2019. Coho made up 50% of the total catch (Table 3.14), a substantial increase from the 18% contribution of coho during 2018 ramp downs (across the full ramp down range from 102 to 1.5 $m^3 \cdot s^{-1}$). This was likely due to the increase in coho abundance in 2019 (see Section 3.4.2). The contribution of mykiss fry to the salvage totals was similar between 2018 and 2019 (56% and 47%, respectively) – the slight decrease in 2019 was mostly due to the increase in the proportion of coho fry. 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 3.14 Summary of numbers of fish salvaged by species-age class and reach under WUP flow ramp downs (≤15 $m^3 \cdot s^{-1}$) in 2019.

Species	**WUP Flows (≤15 $m^3 \cdot s^{-1}$)**	
	Reaches 4 & 3	**Reaches 2 & 1**
Chinook	13	-
Coho	1012	-
Mykiss	948	-
Other spp.	48	-
All	**2,021**	**-**

The least abundant of the target salmonid species in the salvage results were chinook fry, which were encountered slightly more frequently in Reach 4 (*n*= 8) than in Reach 3 (*n*= 5) but contributed only 0.6% to the total number of fish salvaged in 2019 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 **Error! Reference source not found.**).

Other species in the 2019 fish salvage catches were: bull trout (*n*= 2), mountain whitefish (*n*= 2), redside shiner (*n*= 36), and sculpin spp. (*n*= 8). The bull trout and mountain whitefish were exclusively salvaged in Reach 3 and the redside shiner and sculpin spp. were most prevalent in Reach 4. The low numbers of these species in the ramp down results relative to the target salmonid species was likely due to lesser abundance, lower proclivity to stranding, or a combination of both. For the specific catch totals by species for each rampdown event, refer to Table C5 in Appendix D.

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 2019 added another set of results for the Modified Operations period, which started in 2016. However, flows in 2019 followed the Trial 2 hydrograph throughout the year, rather than greatly exceeding 15 m^3/s in the spring, which characterized the 2016-2018 Modified Operations years. One of the goals of monitoring in 2019 was to document how physical conditions in 2019, under a return to the Trial 2 hydrograph, compared to the previous Trial 2 years (i.e., 2011 to 2015) when flow releases were equivalent, as well as the preceding high flow conditions.

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 that were present in 2019 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 3.6) 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 (i.e., according to the Trial 2 hydrograph) in 2019 cannot be considered equivalent to the pre-high flow Trial 2 conditions even though the flow releases were equivalent. However, with only one year of post-high flow data from 2019 available to-date, understanding the linkages between these impacts and aquatic ecosystem recovery have just begun to be explored.

Estimated 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, and this was again evident in 2019. As such, the estimated discharge rates for sites in Reach 2 were substantially higher (by 1.3 to 4.1x 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 (including 2019), 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.

Under the Trial 2 hydrograph in 2019, river stage elevations varied by a total of 0.60 m between the spring peak and the winter low flows at the Rkm 36.8 monitoring location, and there was a gradient of effect with distance from the dam. Total stage changes across the Lower Bridge River were 96 cm at the Terzaghi Dam plunge pool (Reach 4; Rkm 40.9), 51 cm at the fish fence site (Reach 3; Rkm 26.1), 15 cm at horseshoe bend (Reach 2; Rkm 23.6) and 25 cm at Camoo (Reach 2; Rkm 20.0). Under the high flows from 2016 – 2018, the added discharge increased river stage by between 1.08 – 1.42 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. However, it was not possible to measure depths and velocities in mid-channel at the high flows 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 2D model outputs.

The water temperature profiles by reach during 2019 generally followed the patterns observed in other Trial 2 years: 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, there were a couple of notable differences: temperatures were generally cooler (by 1–3°C) than the other Trial 2 years in February; and temperatures were warmer than most other Trial 2 years in spring and summer, which was most evident in reaches 3 and 4. The cause of the cooler water temperatures in February 2019 were attributed to ambient temperature influence since these effects were apparent in all reaches and the Yalakom River (Appendix C). Mean monthly air temperatures (recorded in Lillooet, BC) were colder in February 2019 than they were in that month during the previous Trial 2 period (2011 – 2015).

The warmer temperatures in spring and summer may 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 temperatures in June to September 2019 were not notably warmer than this period during the previous flow trials. This effect was also observed during the Trial 3 high flow years and to a greater extent (Sneep et al. 2019). Nonetheless, temperatures in spring and summer 2019 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).

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?

In the process of making structured decisions about what are preferred flows compared to others 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. The term, "river health" can be ambiguous (Boulton 1999), but in previous explanatory attempts, it 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, Dokulil 2003, 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.

The analytical foundation for testing effects of flow on the benthic communities was analysis of variance blocked among three categorical variables: Trial, Reach, and Pinks presence/absence. "Trial" was of particular interest because its three levels defined three very different flow regimes regulated by flow release from the Terzaghi Dam over years (Figure 1.4). A common approach for testing treatment effects on a large river is a Before After Control Impact design (Stewart-Oaten et al. 1986) in which a response variable is measured in years before and while (or after) a treatment is applied in a control and treatment reach (Johnston et al 1990, Rosario and Resh 2000, Smokorowski and Randall 2017). Statistical tests are then run to determine if the mean difference in the response variable between the control and treatment reach in the "before" years is different from the mean difference between the two reaches in the "after" years. Years are replicates in this layout. If the test shows a statistically significant difference, a conclusion is that treatment affected the response variable. This layout was not possible for the Bridge River because there was no suitable control reach (the entire set of reaches received the same flow treatment). As a result, we were limited to testing a Trial effect in a before after layout. 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 consider 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.

Wetted habitat area increases logarithmically with flow in the Bridge River (**Error! Reference source not found.**). 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 base flow in the fall among years, there was little value in considering wetted area adjustments. Those consistent fall flows produced similar wetted areas among biotic measurements among Trials and Reaches.

Common benthic algae in the Lower Bridge River including diatoms, blue greens, and chlorophytes 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 Bridge River samples has wide-ranging environmental tolerances (Ponader and Potapova 2007). Other 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 extremely 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).

The Bridge River benthos including taxa from the 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. (2007a), Rader and Belish (1999).

Lack of a significant Trial and Reach effect on periphyton PB shows that spring and summer flow associated with Trial was not a factor affecting periphyton biomass accrual in the fall in any of the reaches. No difference in PB between 2019 and Trial 2 years is consistent with this finding. Flow is a product of water depth and velocity. Water depth can influence light attenuation and thus 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 and light attenuation from glacial turbidity declined during the fall, which is typical in the Bridge River. With a small range of water depths, difference in light limitation of algal growth between Trials would not be expected. Water velocities at the plates were in the range where periphyton accrual can respond positively to change in velocity (~0.1 – 0.5 $m \cdot s^{-1}$, Stevenson 1996). At those velocities, nutrients are supplied to most cells in the algal mat (Townsend et al 2012) and shear is not enough to cause sloughing of the biofilm (Rinke et al. 2001). Hence, physical conditions at the samplers were not enough to cause change in periphyton accrual during sampling in the fall between Trials.

Water velocity during the spring and summer high flows could modify amounts of benthic algal biomass (e.g. Townsend et al. 2012, Francoeur and Biggs 2006, Wellnitz and Poff 2006, Rinke et al 2001). The flow – biomass relationship can have a threshold wherein variation in flow at some low flows 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 tightly adhered taxa such as *Nitzschia* sp. may be more easily sloughed. Some form of step response likely occurred in the Bridge River during the spring and summer periods during relatively high flows that defined Trial with those flows increasing in progression from Trial 1 to the high flow years of 2016 - 2018. 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 or temporary equilibrium that is dominated by cellular growth fixed by light, temperature, and nutrient supply with influence by cell loss can take two months (Grimm and Fisher 1989) with shorter times associated with reduced nutrient limitation (Perrin et al 1987, Bothwell 1989). The several weeks at low base flow in the late summer and fall during sampler incubations would have been sufficient for algal biomass to recover, yielding amounts that were similar among all Reaches and Trials. This process shows resilience to disturbance in the Bridge River, particularly by diatom dominated periphyton communities. They may be scoured at extreme flows but recover quickly, which is a trait found in other rivers (e.g. Grimm and Fisher 1989, Tornes et al. 2014). For the Bridge River, it means that extreme physical events only exert a short - term change to autotrophy and do not have impacts on biomass over much more than a couple of months as periphyton production recovers after disturbance.

The Bridge River benthic invertebrates did not show this same resilience. 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 the flows of Trial 2 and high flows of Trial 3. This finding shows that Trial 3 flows exceeded capacity of physical habitat to sustain viable conditions for benthos and that flows during Trials 1 and 2 were more suitable for protection of habitat to support benthic invertebrates. Benthos recovery did not occur between the time of high flows in spring and summer and the time of sampling in the fall during Trial 3 years. Furthermore, benthos showed weak recovery during the lower flows of 2019, a full year after the 2018 high flows. This apparent lack of resilience was surprising given that benthic invertebrates can fully recolonize substrata in two weeks to a month following disturbance (Figueroa et al. 2006, Mackay 1992) although more than a year may be needed for full recovery of community diversity (Chapelsky et al. 2020).

Several factors may contribute to this weak recovery. First, 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 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 3.3). Coarse substrata may trap less fine particulate organic matter (FPOM) that comes from periphyton and allochthonous detritus compared to absence of coarsening and limit the availability of food for many benthic invertebrates (Bundschuh and McKie 2016). The 2019 differences among particle sizes between reaches, 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 based on the Wentworth scale (Bunte and Abt 2001). All these sizes were suitable for colonization by benthic invertebrates (Williams and Mundie 1978) or were at the small end of optimum sizes (Quinn and Hickey 1990). They certainly were not larger than ideal sizes, which might be expected if the high flows mobilized and exported small particles. They also are considered suitable to support spawning by anadromous salmon (Davey 2019; White et al. 2019). The lack of a Reach effect on benthos metrics among Trials is evidence that these particle size differences between reaches were not enough to cause change among the invertebrate assemblages between different places in the river. Loss of large woody material during the high flow years may also reduce 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 scour and removal of riparian vegetation in Reaches 4 and 3 during Trial 3 shows that large organic matter was moved downstream but would be expected to remain at some other place in the river as part of benthic habitat. 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) as part of serial discontinuity (Ward and Stanford 1995). With physical removal of benthos during the 2016 – 2018 high flows, this discontinuity may cause 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. 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 benthos metrics suggests that supplemental drift from the Yalakom River was not enough to change recruitment between reaches upstream and downstream of the Yalakom – Bridge confluence.

None of these physical processes provide convincing explanations for lack of recovery of invertebrate density and diversity during fall base flows after spring to summer high flows during Trial 3 and the poor recovery even one year after the 2018 high flows. Substrata mobility during high flows did not change the particle size distributions enough to reduce suitability for benthos. Trapping of FPOM that provides important food for benthos would be expected to remain effective in the gravels that remained after the high flow events. Much of the large woody debris that provides complex invertebrate habitat and surfaces for grazing was moved by the high flows but remained in different places in the river, thus continuing its functional role in stream metabolism. Serial discontinuity may play a role in slowing recruitment following the high flow disturbances but finding no effect of the Yalakom on benthos metrics (no Reach effect in the Bridge River) shows recruitment may not be limiting community development.

Finding NH_4-N concentrations greater than 20 $\mu g \cdot L^{-1}$ in the absence of anthropogenic pollution was surprising because under well oxygenated conditions in the presence of organic matter as in the Bridge River, NH_4-N is transient in its typically rapid oxidation to NO_3-N in forest streams and aquatic ecosystems of the temperate northwest 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 5 $\mu g \cdot L^{-1}$ in streams. 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 its biological availability in solution. Given these conditions, the high NH_4-N concentrations of 28 – 70 $\mu g \cdot L^{-1}$ that were found during Trials 1 and 2 in Reaches 2 and 3 shows there must be an anomalous NH_4-N source at those places and times.

While there were no known processes in the Bridge River drainage that could explain the NH_4-N patterns, we note that concentrations of NH_4-N, SRP, and to a smaller extent, NO_3-N matched the timing and places of spawning by pink salmon. Highest concentrations always coincided with presence of pink spawners in the odd years while lowest concentrations always occurred in even years when pinks were absent. This strong coincidence shows that spawning pink salmon must have been a source of the anomalous nutrient concentrations. It follows that nutrient concentrations may be a rough indicator of pink run size. High concentrations may be expected during a strong run while low concentrations may be associated with a weak run. If this hypothesis is correct, the nutrient data suggest that run size was strong during the Trial 1 and 2 years but weak during Trial 0 and particularly during Trial 3. The high nutrient loading in odd years of Trial 1 and 2 would have benefitted biological production while lower nutrient loading in even years of Trials 1 and 2 and all years of Trials 0 and 3 would have supported less production. Our conclusion is that spawning Pink salmon introduced a nutrient load to the Bridge River. Coincidental positive response by periphyton PB and benthos density shows a trophic upsurge that is well known in other salmon streams (Wipfli et al 1998, Johnston et al 2004, Ruegg et al.

2012, Harding et al 2014). It is driven by lowering nutrient deficiency (Bothwell 1989, Perrin et al 1987) that propagates through the food web and produces overall increased biomass of invertebrates that are fish food organisms (Johnston et al 1990, Perrin and Richardson 1997, Harvey et al. 1998, Ardon et al. 2020).

These time course shifts in nutrient supply and trophic response present an explanation for the poor recovery of benthos in the fall following the high spring and summer flows in 2016 – 2018 and in 2019 when flows were low and more suitable for benthic invertebrates as in Trial 2. Highest densities of invertebrates occurred during Trials 1 and 2, coinciding with highest concentrations of the macronutrients driven by decomposition of Pink salmon carcasses. Of the two DIN species (NH_4-N and NO_3-N), NH_4-N was most important in this association because it is energetically favoured for uptake by algae compared to NO_3-N. NH_4-N was more reactive to the presence/absence of pinks than was NO_3-N, which is complicated by nitrification reactions and thus is less useful for looking at the Pink nutrient interactions. The most reactive form of phosphorus was SRP, which is the most bioavailable form of phosphorus that can be measured using wet chemistry. Lowest densities of benthic invertebrates occurred in Trial 3 when low NH_4-N and SRP concentrations inferred weak Pink runs. The very low SRP concentrations in flow release from the Terzaghi Dam combined with little to no Pink effect on nutrient concentrations produced the highest potential nutrient deficiency for algal growth and thus food web production during the modified operations years (2016 – 2019) compared to the other flow trials. It culminated during the high flow years (2016 - 2018) when there was lowest invertebrate density and lowest amount of food for fish among Trials. While the high flows would have scoured benthic biota like during a flood event (Robinson 2012), recovery of that biota was strongly limited by newly low nutrient concentrations. SRP concentration was particularly low in Reach 2 due to inorganic N loading from the Yalakom River, which shifted N:P ratios in Reach 2 upwards and into a range showing extreme phosphorus deficiency for algal growth. This condition was new and not observed in the earlier trials. Those SRP concentrations and N:P ratios were similar to those commonly associated with streams known to respond strongly and positively to nutrient addition (e.g. Johnston et al 1990, Perrin and Richardson 1997, Harvey et al. 1998, Ardon et al. 2020). We find that low nutrient concentrations from low pink runs and declining phosphorus loading from Carpenter Reservoir combined with scour during the high flows produced the low benthos densities during Trial 3.

The same process applies to observations of low benthic invertebrate density and diversity in 2019. The NH_4-N and SRP concentrations in 2019 corresponded with those of other Pink years (odd years are Pink years) but they were at the low end of those found during Trials 1 and 2, potentially indicating a low Pink run. Benthic invertebrate density and diversity was equally low, again matching the interannual variation in nutrient concentrations driven by Pink salmon spawning. Given this evidence of bottom – up driven food web response, it is not surprising that benthos density was relatively low in 2019. Nutrient limitation driven by Pinks prevented production of periphyton and benthic invertebrates from reaching amounts of biomass and density that was found during Trials 1 and 2 when Pink driven nutrient concentrations were higher. The continuous decline in soluble phosphorus concentrations in water released from the dam to present values less than 1 µg·L^{-1} would have exacerbated this condition.

There is uncertainty about cause of declining SRP concentrations and rising molar N:P over the past 10 years in water released from Carpenter Reservoir, a trend that is increasing potential phosphorus limitation mainly in Reach 4 of the Bridge River (Figure 3.6). 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. 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 larger influence (Hood and Scott 2008). Further investigation is required to determine if these or other explanations are plausible (see Section 5 Recommendations).

Little has been mentioned of the observations from Reach 1 that was sampled for the first time in 2019 because the single year of data provides only one replicate of each variable within our design. Without replication, few conclusions or interpretations can be made but some initial observations are as follows. Nutrient concentrations were relatively high (Table 3.3), which supported high algal cell densities (**Error! Reference source not found.**) and a high mean PB of 18.4 µg chl-a·cm^{-2} among plates from all stations (not reported in results). These values are consistent with presence of spawning pink salmon. In contrast, benthos densities were lowest on record from all of the upstream reaches (**Error! Reference source not found.**). No conclusions can be drawn from this anomaly, but preliminary insight is Reach 1 had the lowest temperatures among all Trials and Reaches, driven by low temperature in the Yalakom River and downstream cooling. Benthos density was equally low in the Yalakom in 2019, corresponding with these low temperatures. At the mean temperature of 5.2^{o}C in Reach 1, benthos metabolism can be much reduced, leading to low growth rates (Bale and Hayward 2010). Suspension of development with formation of cocoons as in many chironomids or formation of other protective structures may also occur under these low temperatures (Lencioni 2004). Low temperature also limits growth rate among stoneflies (Lillehammer et al. 1989). Temperature inhibition is less of a factor among periphyton that are primarily limited by nutrient supply and only secondarily by temperature (Bothwell 1988). If low temperature limiting Reach 1 invertebrate production is correct, Reach 1 may function differently than the upstream reaches where fall temperatures are much higher. Low temperature may reduce grazing rates and allow periphyton biomass to be greater than at higher temperature. Sampling in additional years using the present layout is needed to address this uncertainty.

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 2019 fish sampling data added another set of results for the Modified Operations years which started in 2016, but were considered a stand-alone year in terms of the analyses since salmonid fry (Age-0+) sampled in 2019 were spawned, incubated, and reared entirely under the low flow conditions (based on the Trial 2 hydrograph) and it was the first year of post-high flow results. Conversely, the other Modified Operations years (2016 – 2018; Trial 3) were characterized by high flows that 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 of each species and age class were almost always highest (or among the highest) in each reach during the Modified Operations years (2016 – 2019) compared to the previous Trial 2 years (2011 – 2015). However, it should be noted that there was considerable overlap in the standard deviation error bars, 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 Mod. Ops. 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 from 2016 to 2019 and lower competition for the food resources that are 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.

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., Trial 3 and 2019). 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) may be the more likely explanation.

Overall, juvenile salmonid abundance and biomass were substantially reduced under the three years of high flows (Trial 3), compared to flow trials 1 and 2 and the pre-flow baseline period, and remained low despite a return to lower flows (based on the Trial 2 hydrograph) in 2019. Total abundance of juvenile salmonids (chinook, coho and steelhead 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 and High flow periods (means = ~189,000 and ~69,000 fish, respectively). Overall, the recruitment of juvenile salmonids was reduced by 70–80% 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. Total juvenile abundance for reaches 2, 3 and 4 was approx. 94,000 fish in 2019, which was an increase of approx. 15,000 – 31,000 fish relative to the three preceding high flow years (or 26,000 more than the Trial 3 average). However, the 2019 abundance was 82,000 – 240,000 fewer fish than the previous Trial 2 years (2011 – 2015), or 160,000 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 20-30% relative to the two flow trials. Steelhead 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 severely depressed due to early emergence effects on their survival or life history caused by the flow release (see Sneep and Evans 2020, in draft). Coho fry abundance was 10% of the Trial 1 and 2 numbers. Coho fry went from being the second most abundant species-age class, 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 3.5) 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 3.22) 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 years and in 2019 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 Trial 3 and in 2019 (except for Chinook fry in 2019), 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 Trial 3 (high flow) years to 2019 was entirely due to improved recruitment of mykiss fry and coho fry (i.e., by approx. 10,000–18,000 and 13,000–16,000 more fish relative to 2016 – 2018, respectively). Abundances of mykiss parr (8,000 fish) and chinook fry (12,000 fish) in 2019 were each similar to the estimates for these species-age classes from the Trial 3 years. The mykiss parr had recruited as Age-0+ fish under the last year of the Trial 3 flows in 2018, so they were not expected to have recovered this year. Chinook fry production has remained relatively low and stable for 10+ years (i.e., since the end of the Trial 1 period). Although, relative to the Trial 2 abundances for mykiss fry (162,000 fish), mykiss parr (33,000 fish) and coho fry (76,000 fish), the recovery in the first year following high flows was quite limited (Table 3.6, Table 3.7 and Figure 3.20).

By reach in 2019, highest juvenile abundances for mykiss (fry and parr) and coho (fry) were in Reach 3. Chinook abundance was highest in Reach 2. Among the reaches, the modest increase in abundance of juvenile fish in 2019 was almost entirely attributable to Reach 3. Recovery of juvenile fish production in reaches 4 and 2 was negligible in the first year following high flows, except for coho which increased by approx. 4,000 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 3.5; White et al. 2019).

Interestingly, these reach-based recovery results for juvenile salmonids in 2019 also aligned with the benthic invertebrate and nutrient source information presented in response to MQ #2. Highest nutrient concentrations (i.e., SRP) and recovery of benthic invertebrate abundance post-high flows were in Reach 3, whereas nutrient concentrations were much lower in reaches 4 and 2, as were benthic invertebrate abundances. This information seems to suggest that potential food source dynamics and reach-based productivity metrics may be factors that explain why fish abundance improved in some reaches and not others. It suggests that food source improvement contributed to some fish abundance recovery in Reach 3, whereas food source constraints may have been a limiting factor to fish recovery in reaches 2 and 4.

The condition factor data suggested that food supply for fish may have limited recovery for all the reaches in 2019. Mean Fulton's K score was lower in each of the study reaches in 2019 than during the Trial 2 years, and values in Reach 3 were generally the same or lower than the other study reaches for each species (Figure 3.19). We would expect that if benthic invertebrate abundance reflected food source availability, then fish in Reach 3 would have been the largest or in the best condition, which was not the case in 2019. Conversely, it is possible that higher food availability was contributing to better survival in Reach 3 in 2019, but it was density-dependent factors that were holding mean weight and condition factor down in that reach. Mean condition factors were higher during the previous Trial 2 years when fish abundances were 2- to 5-fold higher by reach; however, benthic invertebrate abundances were also substantially higher overall in those years (Figure 3.13). The results point to potential food source limitation hindering recovery in 2019; however, with only one year of post-high flow data to consult, this conclusion should still be considered preliminary at this point.

Adult salmon escapement estimates were provided by the BRGMON-3 program in order to evaluate stock-recruitment relationships according to flow release treatments and determine if 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).

The stock-recruitment data point for 2019 (i.e., 2018 spawners vs. 2019 juvenile recruits) was added to the plots for chinook and coho as a stand-alone point and was not factored into any of the existing curves since it represented the start of a new treatment (i.e., the post-high flow period). As such, we cannot draw any new conclusions other than that the 2019 datapoint for chinook reflected a very low spawner estimate in 2018 (n= 25; 95% confidence intervals: 14–44); though the escapement estimate may have been biased low due to the effect of fish fence operations on the spawner surveys that year (White et al. 2019). The spawner estimate was close to a cluster of other low escapement values near the origin of the x-axis, and the juvenile recruitment estimate was similar to all of the other values for trials 2 and 3 on the y-axis.

The 2019 stock-recruitment data point for coho was approx. 37,000 fry below the asymptote of the Trial 2 curve despite a sizeable spawner return in 2018 (*n*= 1,245; 95% confidence intervals: 882–1,627), indicating that recruitment was poor for that spawner stock size compared to the Trial 2 years. Continued data collection (at different levels of escapement) are required to better clarify the initial slope of the stock-recruitment relationships which would help us differentiate among the curves with equivalent fit and strengthen inferences about whether spawning stock size has limited chinook and coho recruitment during any of the monitoring years. Nonetheless, these data are useful for understanding the differences in productive capacity (asymptote of each curve) of the study area for each flow treatment, which provides the same conclusions as comparison of the mean juvenile abundances across trials.

Salmonid abundance data were collected in Reach 1 for the first time in 2019. Results for this year highlighted that all of the target species and age classes were present in the reach, of which mykiss fry were the most abundant (~17,000 fish), followed by coho fry (~10,000 fish), chinook fry (4,000 fish), and then mykiss parr (~1,000 fish). The patterns of abundance among the species in Reach 1 were most similar to their relative contributions in Reach 2 during 2019. The total for all species and age classes (~32,000) was 1/3 of all species in reaches 2, 3 and 4 combined (~94,000). However, due to the extensive length of Reach 1 (half of the total study area length), the lineal densities for all species/age class groups were lowest in Reach 1.

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 in 2019 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. 2019; McHugh and Soverel 2017; Sneep 2016). Ramping across this range in 2019 conformed well to the timing and shape implemented during the Trial 2 years (2011 – 2015; Figure 3.29).

Discharge Effect

The 2019 results reaffirmed that 13 $m^3 \cdot s^{-1}$ is the approximate flow threshold below which fish stranding risk tends to increase from low to moderate or high (as defined in the LBR fish stranding protocol (Sneep 2016)). 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, and 4.6 cm/hr in 2019; Table 3.11) to reduce flows more quickly without increasing fish stranding risk significantly (based on results for 2016 – 2019). 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 2019, 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.

Reach Effect

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 2019 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. The amount of fish stranding area was substantially greater in Reach 3 (7,163 m^2) versus Reach 4 (1,858 m^2) in 2019, and the numbers of fish salvaged were also larger in Reach 3 (1,463 vs 558); However, the fish stranding densities were highest in Reach 4 (30 fish per 100 m^2 of strand area), relative to Reach 3 (20 fish per 100 m^2).

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 – 2019 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 Table 3.10), 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.

Ramping Rate Effect
Ramping rates implemented in 2019 were between 0.7 and 4.6 cm/hr (stage reduction per hour at the 36.8 km compliance location). This represented the third year that rates above the ≤2.5 cm/hr WUP-referenced rate were specifically targeted. 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. Fish salvage results at the higher ramping rates within the Trial 2 range were higher than the results for the high flow range in 2016 – 2018. However, the incidence of fish stranding did not change relative to the identified risk for flows ≤15 $m^3{\cdot}s^{-1}$.

Currently the sample size for stranding monitoring at ramping rates >2.5 cm/hr is still relatively small. As was noted for the MQ above, juvenile fish abundances in 2017, 2018 and 2019 were low overall, which could have confounded the incidence of stranding despite the higher rates in each of those years. However, 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 further test 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 reduce uncertainty about the specific effects of higher ramp rates across the different high flow levels.

River Bank Effect
The distribution of sites between river left and river right was not equal for flows within the Trial 2 range. Based on the 2019 salvage survey data, the distribution of sites was 79% on river left and

21% on river right for ramp downs at flows ≤15 $m^3 \cdot s^{-1}$. Across the high flow range, based on site reconnaissance surveys, 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; km 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.

Opportunities to minimize or mitigate the risk of fish stranding during ramp downs

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 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 in the 2016-2018 high flow years, 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 to 2018, 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 4 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 – 2019) 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.

4.5. 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?

A response to this question based on the current set of information available from BRGMON-1, as well as other studies, was provided in 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 2020, *in draft*). Since this Management Question was one of the focusses of that document, and there was additional context and detail provided within it (beyond what is in this report), we have not repeated that information here. Please refer to the Memo Report, which serves as a supplement to this report, for more information pertaining to this management question.

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?

A response to this question based on the current set of information available from BRGMON-1, as well as other studies, was provided in 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 2020, *in draft*). Since this Management Question was one of the focusses of that document, and there was additional context and detail provided within it (beyond what is in this report), we have not repeated that information here. Please refer to the Memo Report, which serves as a supplement to this report, for more information pertaining to this management question.

5. Recommendations

In addition to the five recommendations provided in the Year 7 (2018) report, the following recommendations stem from the analysis and reporting of results for addressing the management questions up to, and including, Year 8 (2019):

1. There is much uncertainty about the role of changing melt patterns at the Bridge Glacier contributing to time course change in transport of nutrients, particularly phosphorus. This nutrient and others originate in geological formations that are eroded by the Bridge Glacier. They are transported downstream, supporting food webs and fish populations. Flow and water storage management can contribute to some change in system biogeochemistry (Limnotek 2019) but not to the degree seen in Figure 3.6. Larger more regional change may be active. To investigate the potential importance of these larger scale processes on nutrient availability for support of fish food webs, it is strongly recommended that monitoring of nutrient concentrations and transport be conducted over the continuum from the glacier including stations at the inflow to Downton reservoir, the Middle Bridge River, the inflow to Carpenter Reservoir, and the Carpenter Reservoir outflows. Some of this sampling has been done in past projects but it is not sufficient for tracking time course changes in system biogeochemistry that supports food webs and fish populations. New coordinated and longer-term sampling over the continuum is needed to assist with interpretations.
2. We recommend that nutrient addition be examined as a fish enhancement strategy in the Lower Bridge River to offset a progressive decrease in nutrient loading from upstream sources and diminished pink runs (relative to the Trial 1 and 2 years). This decline in nutrient loading has produced unprecedented nutrient deficiency in Reach 4 (due to declining nutrient loading from Carpenter Reservoir and potentially from further upstream) and in Reach 2 (due to high nutrient dilution from Yalakom River inflows). Evidence from many fertilization studies shows that addition of nutrients may increase biological production of the salmonid food web, leading to larger fish with greater survival and greater abundance.
3. To the extent possible, maintain a consistent hydrograph shape and flow magnitudes within the next few years to add additional post-high flow replication to the 2019 results. From a data interpretation perspective, too much variability in flow release conditions among consecutive years introduces constant change that adds to the variables already needing to be accounted for when analyzing the results. Replication is an important factor for building confidence in the results and understanding the degree of natural (versus operations-induced) variation among the flow treatments.
4. Preferably ensure that the collection of spawner escapement data for coho and chinook under BRGMON-3 can be unimpeded by broodstock collection fence operations to the extent possible. When the completion of spawner surveys are compromised due to fence

operations it increases bias and uncertainty in the stock recruitment relationships for assessing potential spawner stock size limitations. This issue may be mitigated by setting the fence up further away from the counter site at Rkm 26.1 (to reduce recycling over the counter) and allow for completion of the weekly spawner streamwalks throughout reaches 3 and 4 to ensure comparability of the data with the existing time series.

6. References Cited

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.

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., 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.

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.

Bale, J.S. and S.A.L. Hayward. 2010. Insect overwintering in a changing climate. The Journal of Experimental Biology. 213: 980-994.

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

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. 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. 2019. Scope of Services – 2019 Lower Bridge River High Flow Monitoring & 2019 Lower Bridge River High Flow Fish Salvage and Stranding Risk Assessment & 2019 (Year 8) BRGMON-1 – Lower Bridge River Aquatic Monitoring and Reporting. Prepared by Matt Casselman BC Hydro Fish and Aquatic Issues Team, April 2019 (Revised July 2019).

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.

Biggs, B.J.F., S.N. Francoeur, A.D. Huryn, R. Young, C.J. Arbuckle, and C.R. Townsend. 2000. Trophic cascades in streams: effects of nutrient enrichment on autotrophic and consumer benthic communities under two different fish predation regimes. Canadian Journal of Fisheries and Aquatic Science. 57: 1380-1394

Bothwell, M.L. 1988. Growth rate responses of lotic periphytic diatoms to experimental phosphorus enrichment: The influence of temperature and light. Canadian Journal of Fisheries and Aquatic Science. 45: 261-270.

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.

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.

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.

Clarke, K.R. and R.N. Gorley. 2006. Primer v6: User Manual/Tutorial. PRIMER-E Plymouth, UK.

Clausen, B. and B.J.F. Biggs. 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. *Freshwater Biology* 38: 327-342.

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

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.

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

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.

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.

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.

Ramos-Espinoza, D., M. Chung, G. Poole, C. Melville, and C. White. 2018. Lower Bridge River Adult Salmon and Steelhead Enumeration, 2017. Report prepared by Instream Fisheries Research Inc. for St'át'imc Eco-Resources and BC Hydro. 84p.

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.

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.

Gore, J.A. 1982. Benthic invertebrate colonization: source distance effects on community composition. *Hydrobiologia*. 94: 183-193.

Gore, J.A., J.B. Layzer, and J. Mead. 2001. Macroinvertebrate instream flow studies after 20 years: a role in stream management and restoration. *Regulated Rivers: Research and Management*. 17: 527-542.

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.

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.

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.

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

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., 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.

Kennan, J.G., K. Riva-Murray, and K.M. Beaulieu. 2010. Determining hydrologic factors that influence stream macroinvertebrate assemblages in the northeastern US. *Ecohydrol.* 3: 88-106.

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.

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.

Krebs, C.J. 1999. *Ecological Methodology*. Addison Wesley. Longman, NY.

Kurle, C.M. and B.J. Cardinale. 2011. Ecological factors associated with the strength of trophic cascades in streams. Oikos. 120: 1897-1908.

Lencioni, V. 2004. Survival strategies of freshwater insects in cold environments. Journal of Limnology. 63: 45-55.

Lillehammer, A., J.E. Brittain, S.J. Saltveit, and P.S. Nielsen. 1989. Egg development, nymphal growth and life cycle strategies in Plecoptera. *Holarctic Ecology.* 12: 173-186.

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

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

Marchant, R. and G. Hehir. 2002. The use of AUSRIVAS predictive models to assess response of lotic macroinvertebrates to dams in south-east Australia. *Freshwater Biology*. 47: 1033-1050.

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.

Melnychuk, M., Korman, J., Hausch, S., Welch, D., McCubbing, D., and C. Walters. 2014. Marine survival difference between wild and hatchery-reared steelhead trout determined during early downstream migration. Can. J. Fish. Aquat. Sci. 71: 831-846.

Mérigoux, S. and S. Doledec. 2004. Hydraulic requirements of stream communities: a case study on invertebrates. *Freshwater Biology* 49: 600-613.

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.

Ministry of Environment. 2009. The Canadian Aquatic Biomonitoring Network – Field Manual. Prepared by Ministry of Environment Science & Information Branch, Watershed and Aquifer Science Section for the Resource Information Standards Committee. Version 1.0. March 11, 2009.

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.

Nelson, S.M. and D.M. Lieberman. 2002. The influence of flow and other environmental factors on benthic invertebrates in the Sacramento River, U.S.A. *Hydrobiologia*. 489: 117-129.

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.

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.

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

Reynoldson, T.B., R.H. Norris, V.H. Resh, K.E. Day, and D.M. Rosenberg. 1997. The reference condition approach: a comparison of multimetric and multivariate approaches to assess water quality impairment using benthic macroinvertebrates. Journal of the North American Benthological Society 16: 833-852.

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.

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.

Slavik, K., B.J. Peterson, L.A. Deegan, W.B. Bowden, A.E. Hershey, and J.E. Hobbie. 2004. Long-term responses of the Kuparuk River ecosystem to phosphorus fertilization. Ecology. 85: 939-954.

Sneep, J., and M. Evans. 2020, *In Draft*. 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. March 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. 2012. 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.

Stanford, J.A., J.V. Ward, W.J. Liss, C.A. Frissell, R.N. Williams, J.A. Lichatowich, and C.C. Coutant. 1996. A general protocol for restoration of regulated rivers. *Regulated Rivers: Research and Management*. 12: 391-413.

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.

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.

Tronstad, L.M., B.P. Tronstad, and A.C. Benke. 2007. Aerial colonization and growth: rapid invertebrate responses to temporary aquatic habitats in a river floodplain. *Journal of the North American Benthological Society*, 26:460-471.

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.

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.

Wetzel, R.G. 2001. Limnology. Academic Press. San Diego.

White, C., D. Ramos-Espinoza, M. Chung, G. Poole, K. Cooke, and C. Melville. 2019. Lower Bridge River Adult Salmon and Steelhead Enumeration – Implementation Year 7 (2018). Ref. BRGMON-3. Report prepared for St’at’imc Eco-Resources by Instream Fisheries Research Inc.

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

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

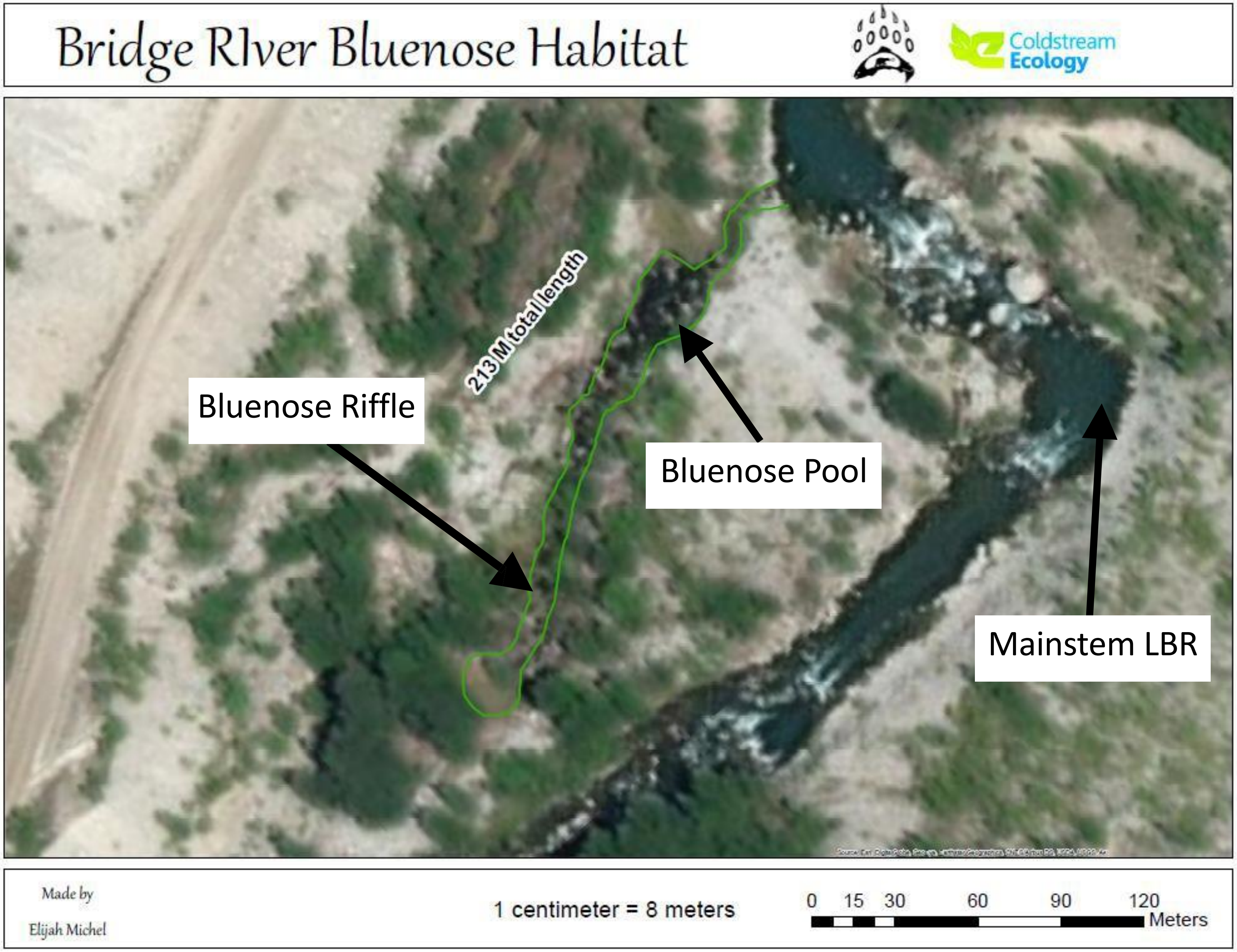
[a] UTM zone is 10U.

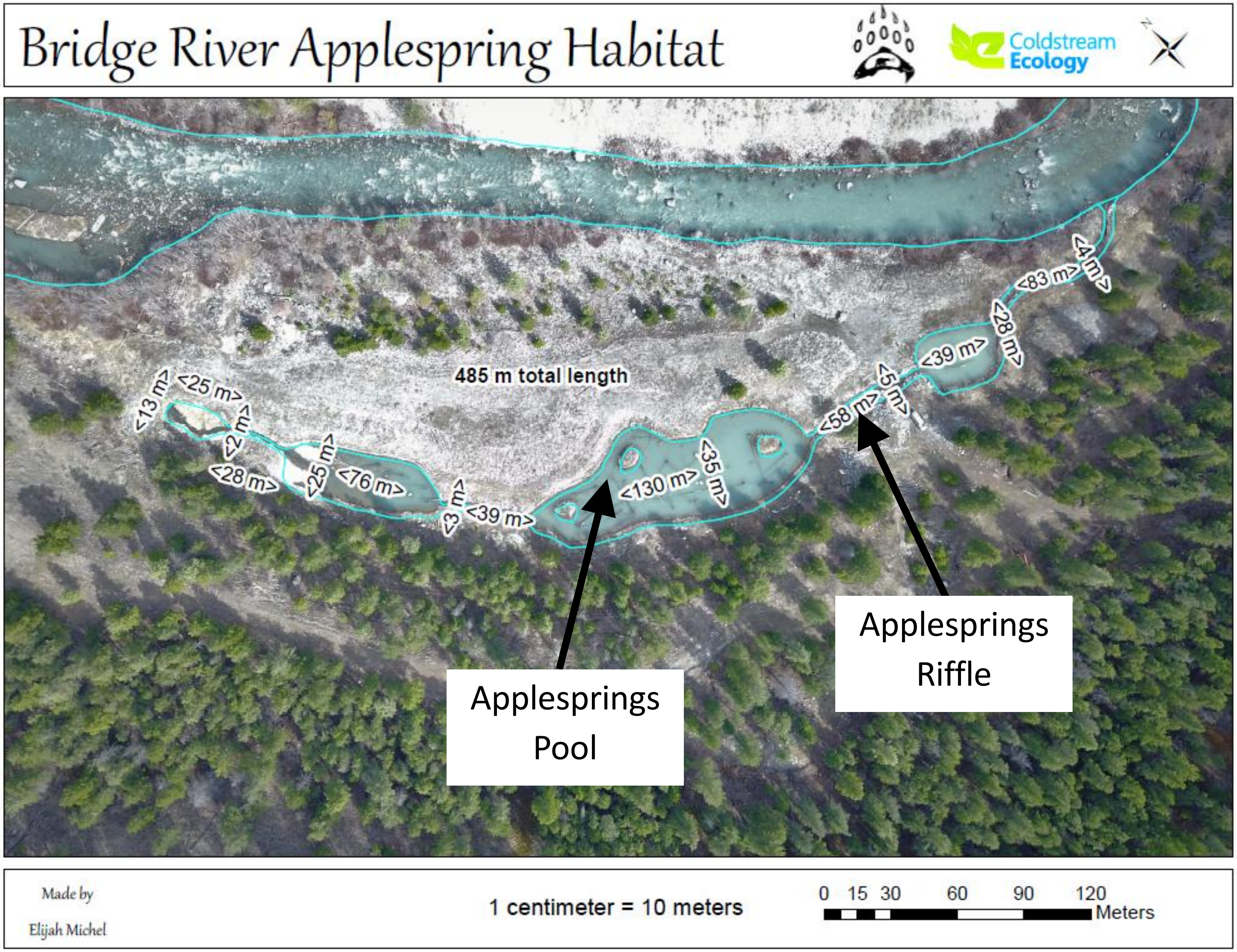
Off-channel Fish Sampling Locations

Field Component	Reach	Site	Approx. River Km	Bank	UTM Coordinates	
					Easting	Northing
Juvenile Salmonid Habitat Use	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
Offchannel 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
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
N
485 m total length
<13 m>
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<2 m>
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<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) $c_{y,i,j} \sim dbin(\theta_{y,i}, N_{y,i,j})$

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) $N_{y,i,j+1} = N_{y,i,j} - c_{y,i,j}$

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) $\theta_{y,i} \sim dbeta(\alpha_y, \beta_y)$

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

(4) $\log(\lambda_{y,i}) \sim dnorm(\mu_{\lambda_{y,r}}, \tau_{\lambda_{y,r}})$

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) $N_{y,i} = \lambda_{y,i} a_{y,i}$

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) $Ntot_{y,r} = Nss_{y,r} + Nus_{y,r} + Nuc_{y,r}$

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

(7) $Nss_{y,r} = \sum_i N_{y,r,i,1}$

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:

$$(8) \qquad Nus_{y,r} = \exp\left[\mu_{\lambda_{y,r}} + 0.5\tau^{-1}_{\lambda_{y,r}}\right](2l_r w_{y,r} - \textstyle\sum_i a_{y,i})$$

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).

$$(9) \qquad Nuc_{y,r} = \left(Nss_{y,r} + Nus_{y,r}\right)(1 - \rho_{f,r})$$

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 ($m_{f,r}$).

$$(10) \qquad h_{f,y} \sim dbin(\rho_{y,r}, m_{f,r})$$

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.

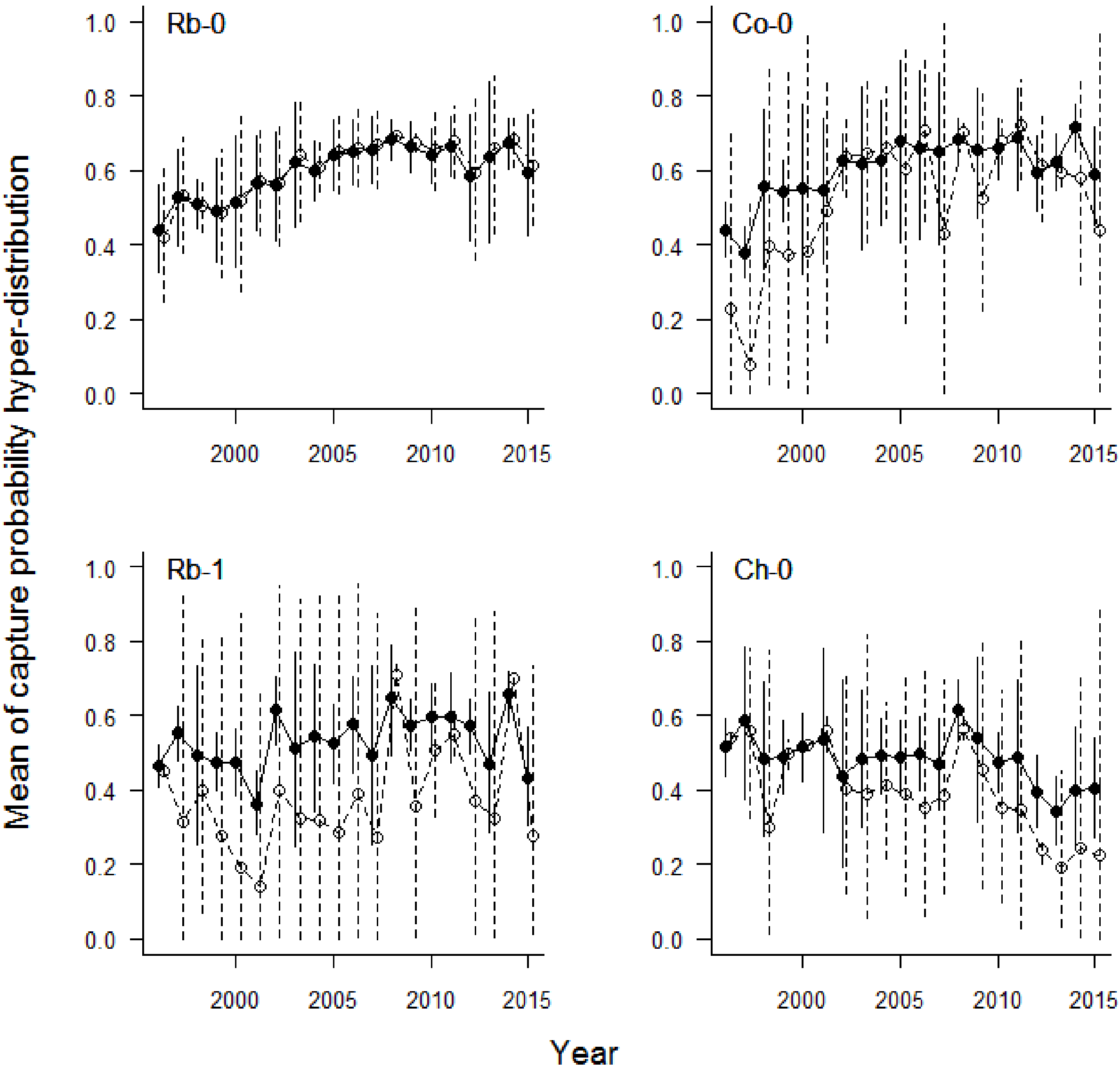


Figure B1. Annual estimates of the mean (with 90% credible interval) of the capture probability hyper-distribution (distribution of capture probability across sites) based on the HBM with more restrictive priors for the capture probability hyper-distribution (solid symbols). Also shown are estimates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

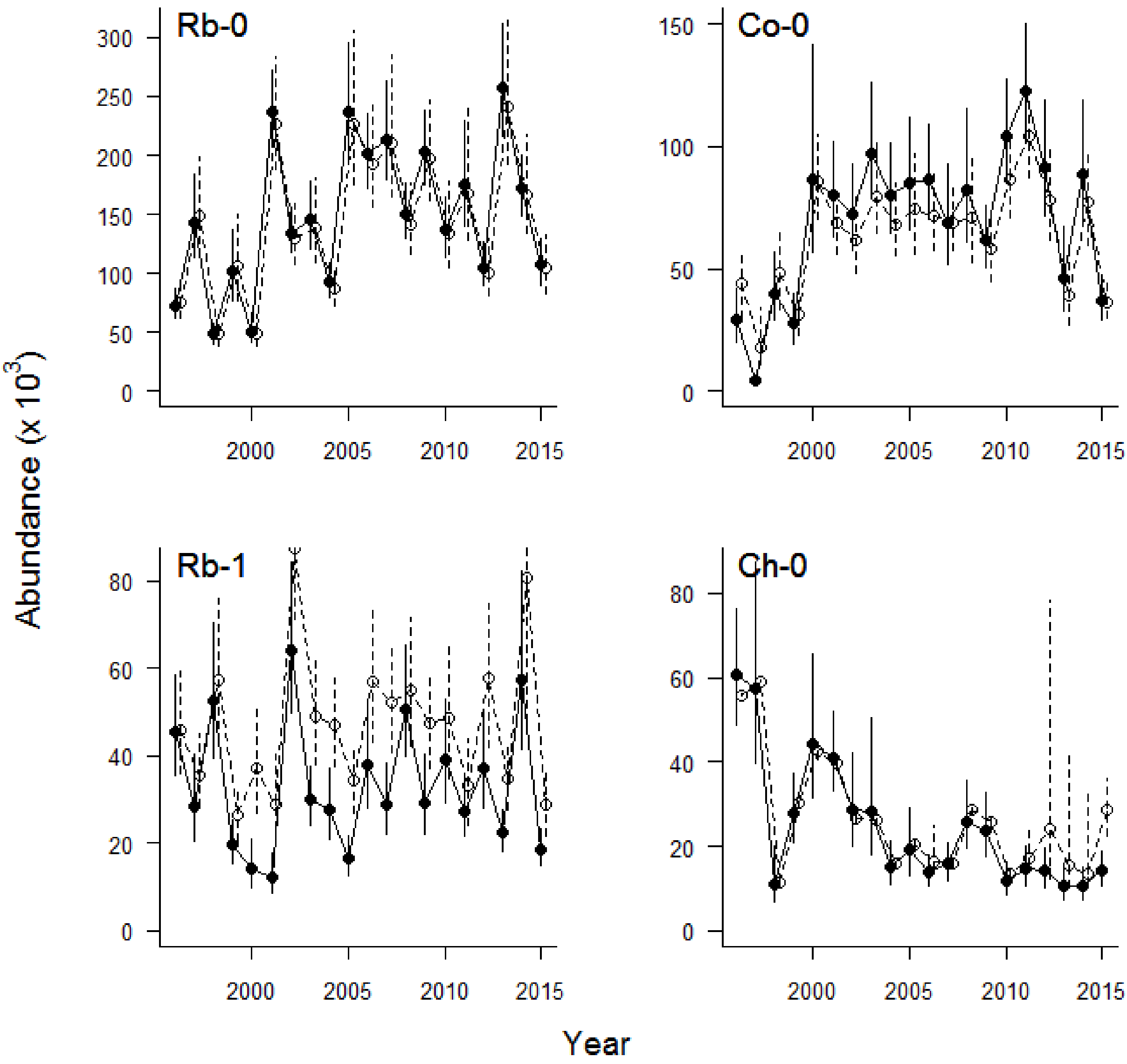
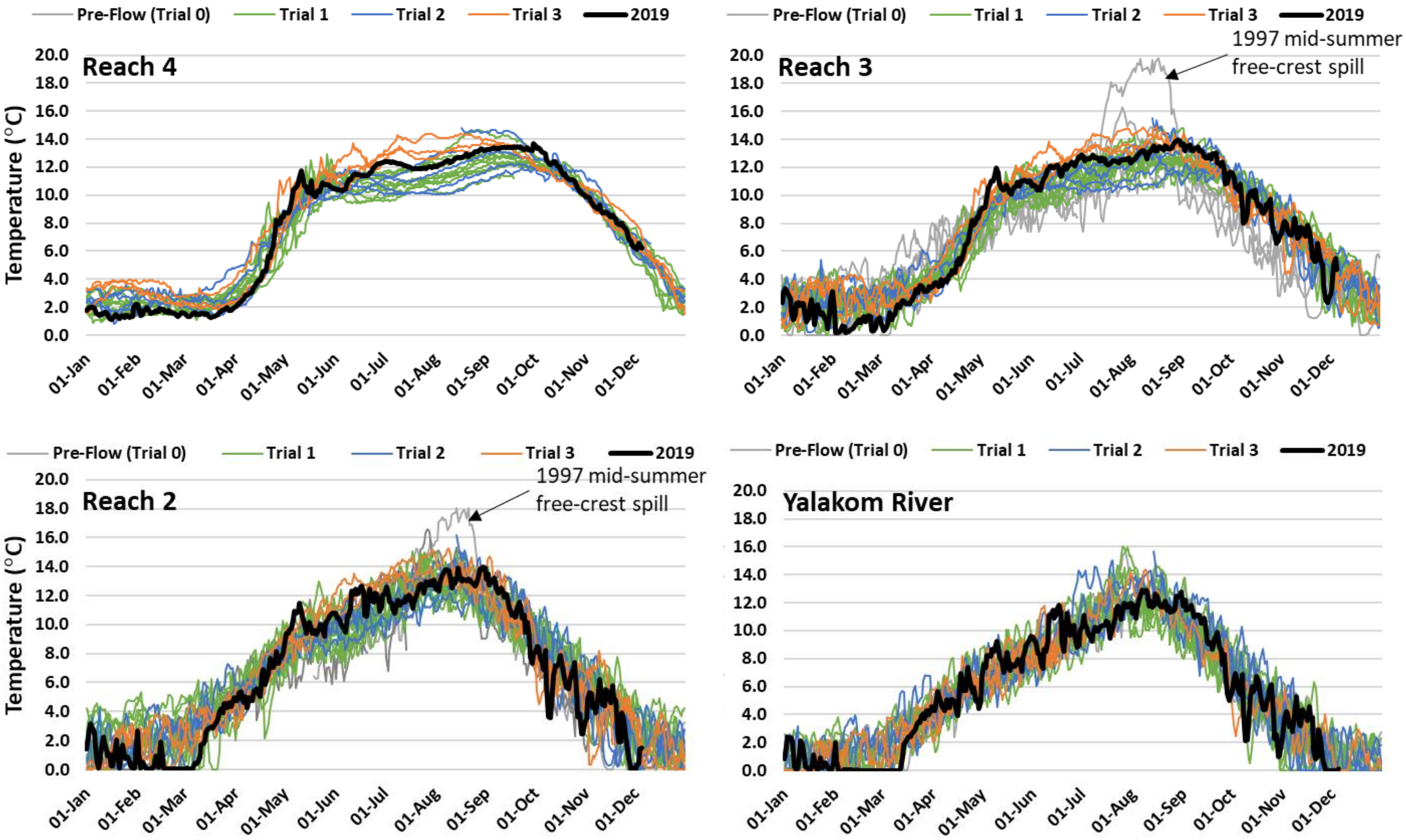


Figure B2. Annual estimates of abundance (all reaches combined) based on the HBM with a more restrictive prior (solid symbols). Also shown are estimates based on the uninformative priors used in Bradford et al. (2011, open symbols).

Appendix C – Mean Water Temperatures in the Lower Bridge River (by Reach) and the Yalakom River for each Flow Trial Year and 2019.



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

Table C1 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2019.

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)
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
WUP Rampdown Summary			**5**	**15.2**	**1.5**	**15.2**	**144**	**84**	**-60**	**-1.3 (Max.)**

Table C2 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 3.30 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 C3 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·s^{-1})	End Flow (m^3·s^{-1})	Flow Change (m^3·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 C4 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 C5 Fish salvage tallies by species, ramp date and flow range – 2019 results (reaches 3 and 4 only).

Species	Ramp Date and Flow Range ($m^3 \cdot s^{-1}$)										Species Totals
	1 Aug (15.2–12.9)	6 Aug (13.0–11.1)	8 Aug (11.1–9.3)	9 Aug (9.3–7.7)	14 Aug (7.7–6.4)	15 Aug (6.4–5.1)	18 Aug (5.1–4.1)	19 Aug (4.1–3.0)	1 Oct (3.2–2.1)	2 Oct (2.1–1.5)	
Chinook						5	3	5			**13**
Coho	26	71	118	270	136	253	33	79		26	**1,012**
Mykiss	18	17	69	278	135	131	76	137	9	78	**948**
Bull trout						2					**2**
Mountain whitefish				1		1					**2**
Redside shiner	2	29	3			1				1	**36**
Sculpin spp.										8	**8**
Daily Totals	**46**	**117**	**190**	**549**	**271**	**393**	**112**	**221**	**9**	**113**	**2,021**