Bridge River Project Water Use Plan

Lower Bridge River Aquatic Monitoring

Implementation Year 6

Reference: BRGMON-1

BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 6 (2017) Results

Study Period: April 1 2017 to March 31 2018

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November 7, 2018

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File no. BRGMON-1

April 2018

Executive Summary

A second year of high flow monitoring was conducted in 2017. The peak flow release from Terzaghi Dam was 127 $m^3 \cdot s^{-1}$ and average flows for the year were 19 $m^3 \cdot s^{-1}$. The high flow period began in the third week of May, peaked across the month of June, and was ramped back down by the third week of July (high flow duration = 59 days). Outside of the high flow period, the flow releases conformed to the Trial 2 hydrograph from the Lower Bridge River (LBR) flow experiment.

Increases in the maximum Terzaghi Dam discharge were expected to have impacts on the aquatic ecosystem in the LBR. In both the short- and long-term, high flows were anticipated to affect periphyton accrual and biomass, benthic invertebrate abundance and diversity, and juvenile salmonid growth and abundance, related to disturbance and changes in habitat suitability associated with the high flows. Monitoring in 2016 and 2017 was intended to characterize some of these effects in reaches 2, 3 and 4 in the first and second year of high flow implementation. Comparisons with previously monitored flow treatments are included.

The core methods (field and laboratory) employed for monitoring the effects of the Terzaghi flow releases in 2017 were generally consistent with those employed during the Trial 0 pre-flow (0 $m^3 \cdot s^{-1}$; 1996 to July 2000), Trial 1 (3 $m^3 \cdot s^{-1}$; August 2000 to 2010), Trial 2 (6 $m^3 \cdot s^{-1}$; 2011 to 2015), and other Trial 3 high flow (>18 $m^3 \cdot s^{-1}$; 2016) years. Four core monitoring activities were conducted: 1) continuous recording of flow release discharge, river stage and temperature; 2) assessment of water chemistry parameters, periphyton accrual, and aquatic invertebrate abundance and diversity during fall; 3) periodic sampling to monitor juvenile salmonid growth; and 4) a fall standing stock assessment to estimate the relative abundance and distribution of juvenile salmonids in the study area.

Some additional monitoring components to assess some of the short-term impacts of the high flows were also conducted in 2017. These activities included: kokanee entrainment surveys; high flow ramp down monitoring and stranding risk assessment; and sediment and erosion monitoring.

On balance, the net effect of the high flows in 2016 and 2017 was negative on virtually every major productivity metric in the Lower Bridge River study area compared to the results from the previous flow treatments (trials 0, 1, and 2). Following is a brief summary of the high flow (Trial 3) results based on the various aquatic monitoring components implemented:

- Due to the confined nature of the channel throughout most of the study area, and particularly in reaches 3 and 4, the flooding of the channel by the higher flows resulted in substantial increases in depth (1.43 m at km 36.8 above Trial 2 peak) and mid-channel velocities (unmeasured but anecdotal), which reduced the amount of suitable rearing habitat per wetted area;

- The higher flows introduced increased shear forces that mobilized sediments (both erosion and deposition) and are trending the channel towards a pre-regulation condition that retained fewer spawning-sized gravels. The high magnitude flows in 2017 reset the sediment mobility threshold values (to between ~20 and ~50 $m^3 \cdot s^{-1}$) from the values estimated for previous peak flow magnitudes. The thresholds vary according to location due to differences in localized resiliency (Ellis et al. 2018);
- Water temperatures remained elevated across the fall period, relative to the pre-flow regime, as has been reported for trials 1 and 2; These elevated temperatures accelerate incubation to emergence for chinook fry, particularly in Reach 4 and the top of Reach 3, and may reduce fry survival or limit spawning use of these otherwise potentially productive areas;
- Overall benthic invertebrate density declined by 64% following the high flows in 2016 and 2017 (relative to Trial 2 abundances) and all fish food organisms were affected;
- Low abundance of benthic invertebrate abundance at low base flows in the fall, approximately 3 months after peak flows timing in spring to early summer, means the effect of the high flows was sustained, suggesting poor recruitment from upstream sources (due to impoundment of the channel by the dam);
- Juvenile salmonid abundance (measured during the stock assessment sampling in September) was reduced by 70% compared to the Trial 2 average (reductions by species-age class were: -70% for steelhead fry, -70% for steelhead parr, equivalently low abundance for chinook fry, and -90% for coho fry);
- Juvenile salmonid biomass trends mirrored the trends in abundance since differences in mean size for each species and age class were generally not significant among the trials (or less so relative to the differences in mean abundance);
- Stock-recruitment curve for Trial 3 (high flows) suggests poorest recruitment of coho fry per spawner stock size for any of the flow treatments assessed, and equivalently low production of chinook fry (as the other trial flows); however, more years of data are required to inform the initial slope of the curves and reduce uncertainty;
- The high flows flood additional edge areas, including habitats that become isolated from the mainstem or dewater when flows are reduced, thereby adding to the total numbers of fish stranded across the lower flow ranges. However, the rate of stranding appears to be lower at flows above ~13 $m^3 \cdot s^{-1}$ than below.
- Entrainment of kokanee from Carpenter Reservoir into the Lower Bridge River channel occurred in both 2016 (*n*=83 observed) and 2017 (*n*=48 observed).

Results that noted positive effects or changes included:

- Increased wetted area, although the gain in wetted area per volume of discharge diminishes above ~7 $m^3 \cdot s^{-1}$, and increasing depth and velocities diminish the proportion of suitable rearing habitat area;

- The higher flows (i.e., increased river stage elevation and velocities) engaged sediment sources (e.g., fans) along the channel edge that resulted in additional recruitment from sources that were not mobilized at the lower trial flows. Ellis et al. (2018) state that the volume of sediment recruited from fans in 2017 was more than twice the recruitment volume observed in 2016;
- Warmer water temperatures during the spring and early summer period within optimal ranges for rearing may have benefited feeding and growth for juvenile fish that remained/survived following the peak flow period in reaches 2, 3, and 4;
- Improved periphyton growth following higher flows. For the fall sampling that occurred in all trials, algal cell density was the same between Trials 0 and 1 ($p=0.72$), it increased by more than two orders of magnitude in Trial 2 ($p<0.001$), and it increased by another four times in Trial 3 ($p<0.001$). All reaches contributed to the trial effect in the fall but Reach 4 contributed most and Reach 2 contributed least to the differences in algal cell densities between Trials 2 and 3;
- Higher mean size of juvenile steelhead, chinook and coho in each reach (though there was significant overlap in standard deviations in many cases). However, this was likely related to substantially reduced abundance since food sources (invertebrates) were also substantially reduced (see above). In other words, smaller mean size under previous flow trials was likely due to density-dependent factors when abundances were much higher.

Summary of BRGMON-1 Management Questions and Interim (Year 6 – 2017) Status

Primary Objectives	Management Questions	Year 6 (2017) 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.	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. Additional gains from higher flows are proportionally less substantial and reduce the suitability of mid-channel habitats by increasing flow velocities above suitable thresholds. • Higher flows introduced increased shear forces that mobilized sediments (i.e., erosion in some areas and deposition in others). Flow magnitudes in 2017 reset the sediment mobility thresholds to between ~20 and ~50 $m^3 \cdot s^{-1}$. High flows also recruited material from edge sources (e.g., the toe of fans). • Water temperatures under all trial flows were cooler in the spring and warmer in fall relative to the pre-flow profile. Under high flows in 2016 and 2017 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).
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.	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?	• Periphyton accrual (cell density per m^2), as measured in fall, was positively correlated with peak flow magnitude in spring/early summer. Under Trial 3, accrual was highest in Reach 4 and lowest in Reach 2. • Overall benthic invertebrate density declined by 64% following the high flows in 2016 and 2017 (relative to Trial 2 abundances) and all fish food organisms were affected. • Low abundance of invertebrates 3 months after the peak flow period suggested poor recruitment to offset losses (due to effects of channel scour, etc.) caused by the high flows. As observed in other impounded systems, it is likely that the dam has segregated the Lower Bridge River channel from upstream recruitment sources.
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 do changes in physical conditions and trophic productivity resulting from flow changes together influence the recruitment of fish populations in Lower Bridge River?	• Juvenile salmonid abundance was highest (overall) under the Trial 1 and 2 flow regimes (in general, production between them was near equivalent, but both impacted chinook recruitment). Relative to the previous flow treatment, the high flows in 2016 and 2017 reduced salmonid abundance by 70%. Reductions for steelhead and coho juveniles were between 70% and 90%. Chinook fry abundance remained low (equivalent to Trial 2). • Juvenile salmonid biomass trends mirrored those for abundance. • Based on stock-recruit analysis, production for chinook and coho is characterized by a different curve for each flow treatment. It is likely that habitats were fully seeded in most study years; however, more data are required to reduce uncertainty. • Higher mean weight of juvenile salmonids during the fall stock assessment period was observed for Trial 3 in each reach (although there was significant overlap in standard deviations). Lower fish abundance likely resulted in reduced competition for the available food resources.

Primary Objectives	Management Questions	Year 6 (2017) Results To-Date
	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 2017 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. • 2016 and 2017 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.
High Flow Ramp Down Monitoring and Stranding Risk Assessment	Is the stranding risk during ramp downs at flows >15 $m^3 \cdot s^{-1}$ different than the stranding risk during ramp downs <15$m^3 \cdot s^{-1}$?	• Response based on 2016/2017 results: Yes*****. • Above a threshold of ~13 $m^3 \cdot s^{-1}$, the fish stranding risk (per 1 $m^3 \cdot s^{-1}$ increment of flow change) was consistently low (or occasionally moderate). Conversely, below the 13 $m^3 \cdot s^{-1}$ threshold, the fish stranding risk was more consistently high. • This difference likely provides the opportunity to continue to implement (and monitor) faster ramp rates for higher flows (>13 $m^3 \cdot s^{-1}$) ***** Important caveat: juvenile fish abundance was substantially reduced overall in 2016 & 2017, which likely affected salvage results following high flows during those years.
	Is the stranding risk equal across reaches of the Lower Bridge River?	• Response based on 2016/2017 results: No. • Under previous flow trials (≤15 $m^3 \cdot s^{-1}$), differences in the number of fish salvaged (per 100 m^2) among reaches was significant. Reach 4 densities were more than double Reach 3 densities. • Differences among reaches in the high flow range (>15 $m^3 \cdot s^{-1}$) were also apparent but they were smaller. Slightly higher densities were observed in reaches 2 and 3 than in reaches 1 and 4.
	Does the stranding risk change when the maximum hourly ramping rate is greater than 2.5 cm/hr?	• Response based on 2016/2017 high flow results: No*****. • At ramp rates up to 4.1 cm/hr implemented for the first time in 2017, there was no appreciable difference in fish stranding risk relative to lower rates (≤2.5 cm/hr) within the high flow ranges tested (80.4 to 67.1 $m^3 \cdot s^{-1}$, 67.2 to 55.1 $m^3 \cdot s^{-1}$, and 55.2 to 44.7 $m^3 \cdot s^{-1}$). • These results, while preliminary at this point, suggest there is opportunity to further test higher rates across the high flow range going forward. ***** Important caveat: the sample size for strand monitoring at ramping rates >2.5 cm/hr is small and abundance of juvenile salmonids in 2017 was low overall, which could have influenced results.
	Is the stranding risk equal on the left and right banks of the Lower Bridge River?	• Response based on 2016/2017 results: Yes***** for high flows (>15 $m^3 \cdot s^{-1}$); and No***** for flows <15 $m^3 \cdot s^{-1}$. • At high flows, site distribution was close to equal (40% river left; 60% river right), whereas at low flows, the distribution was more skewed (80% river left; 20% river right). We speculate that these differences at the lower flows are due to human-caused effects (e.g., river access, gold mining, gravel placements, etc.) on habitats at low elevations, rather than natural causes.
	What are the potential incremental impacts of	• As mentioned for several of the MQs above, the high flows in 2016 and 2017 resulted in much lower fish abundance across the study area than under any of the previous flow treatments (i.e., trials 0, 1, or 2).

Primary Objectives	Management Questions	Year 6 (2017) Results To-Date
	2016-2017 flows on fish stranding in the Lower Bridge River during summer ramp downs from high flows?	These substantially reduced numbers would most certainly have had an influence on the number of fish stranded in 2016 and 2017, particularly because coho and steelhead fry were most strongly affected by the high flows and these are the species and age class that are usually at greatest risk of stranding.
	What are the changes to the Adaptive Stranding Protocol recommended to manage fish stranding risks in the future?	• The high flow ramp down data from 2016/2017 provide an important supplement to the data that was available for the protocol (flows ≤20 $m^3 \cdot s^{-1}$) at the time it was written. • 2016/2017 ramp down monitoring provided some important learning about stranding risk according to flow rate, reach, side of the river, and ramp rates for high flows that had not previously been assessed. • Low fish abundance overall, and limited sample size at high flows or for faster ramp rates, precluded full certainty for answering some of the MQs at this stage. Further high flow ramp down monitoring is recommended. • Attempts were made to align the analyses and description of the results to facilitate incorporation of the high flow data into the protocol in future, if there is interest in doing so.
Lower Bridge River Sediment and Erosion Monitoring [Responses from: Ellis et al. 2018]	What is the post-2016 flow threshold that is likely to mobilize the remaining spawning size sediment?	• A single mobility flow threshold for the entire river was not identified, as mobility flow thresholds vary by substrate size and channel geometry. Some locations in Reaches 4 and 3 are more resilient against mobility than others as identified by higher mobility flow thresholds. • In general, flow thresholds in the range of 20 $m^3 \cdot s^{-1}$ to 50 $m^3 \cdot s^{-1}$ would be required to maintain the observed spawning-sized sediment in the Lower Bridge River. To maintain the spawning-sized sediment (observed pre-2016) would require a return to pre-2016 regulated flows. • After mobility is initiated, sediment transport rates increase steeply with flow. This implies that the shape of the hydrograph (for a given volume of flow) may have an important effect on total sediment transport. • If spawning-sized sediment is to be maintained during high flows similar to 2016-2017, the suggested approach would be to locate spawning habitat in more resilient locations (as identified by higher mobility flow thresholds), and to implement a hydrograph shape that limits mobility. • Residual uncertainties in the relationship between mobility and flow could be further addressed by monitoring the effects of different high flow durations / magnitudes / hydrograph shapes.

Primary Objectives	Management Questions	Year 6 (2017) Results To-Date
	What are the impacts of high flows on the level of sediment embeddedness and concentration of fines in the mesohabitats used by juveniles in the Lower Bridge River?	• Due to the small sample size (two sites), the embeddedness results are preliminary. These preliminary results suggest that pore sizes in the 2017 post-freshet condition became deeper, but the density of pores decreased. • Assuming a similar depth of erosion occurred at the embeddedness sites, it appears that sheltering in sediment may not provide adequate refuge for juvenile salmonids during high flows.

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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 once per decade (Figure 1.1). The flooding and subsequent dewatering associated with these events inevitably had impacts on the LBR ecosystem.

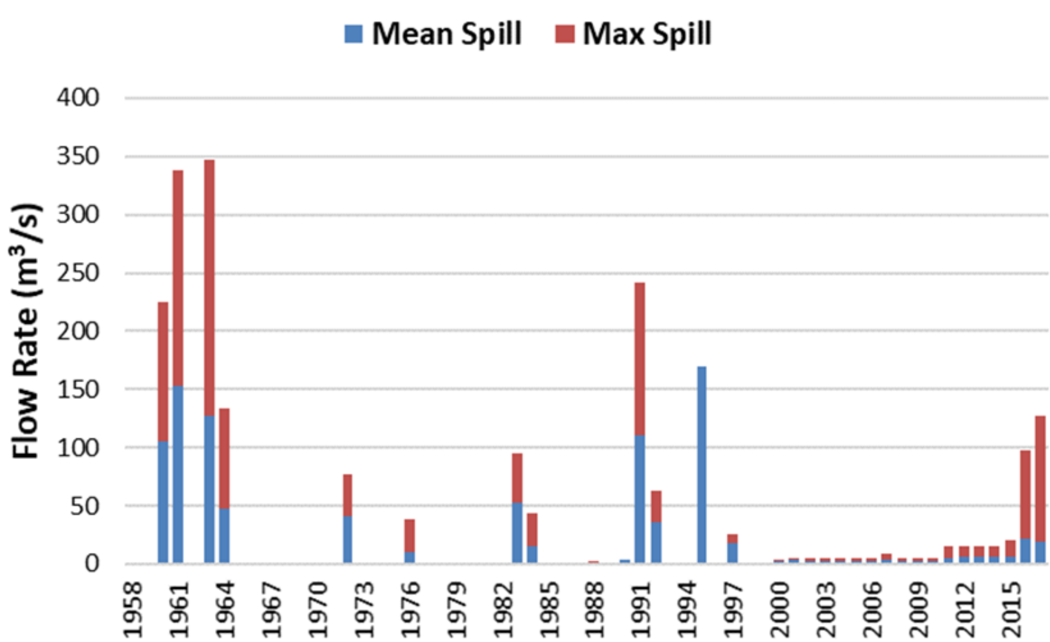


Figure 1.1 Frequency of spill and flow release events from Terzaghi Dam into the Lower Bridge River following impoundment in 1960.

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 significant 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 (Crane Creek Enterprises 2012; McHugh and Soverel 2016).

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.

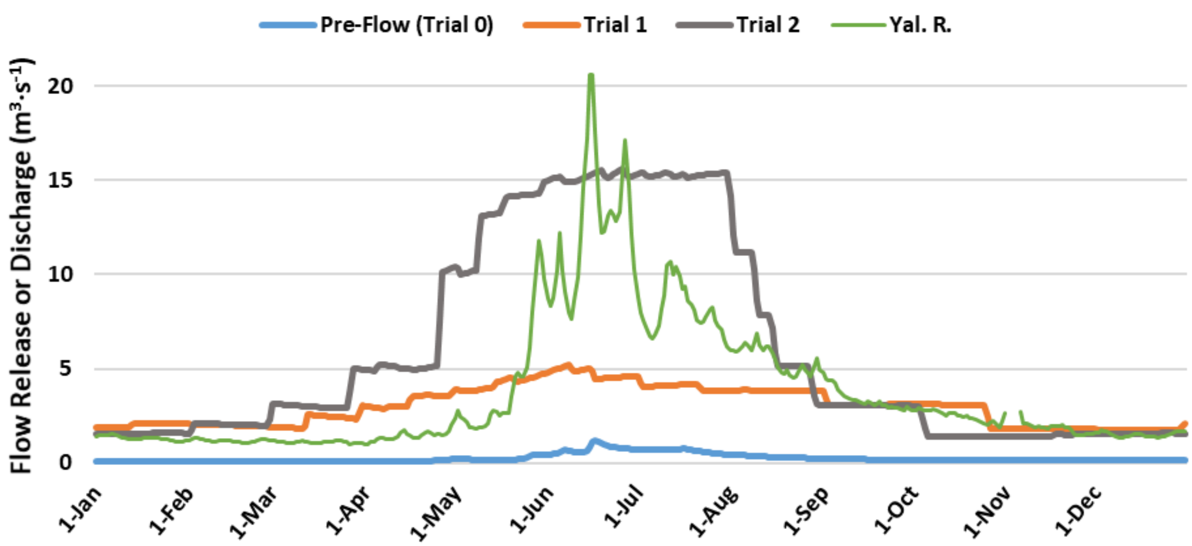


Figure 1.2 Mean daily releases from Terzaghi Dam for Trial 1 and Trial 2 during the flow experiment. Typical hydrograph shapes during the Pre-flow period and for the unregulated Yalakom River discharges are included for reference.

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. Additional High Flows

At some point during the 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) would need to be reduced for a period of years to mitigate the issues and allow for the affected infrastructure to be rebuilt or replaced.

The reduction of water storage and flow diversion above Terzaghi Dam meant that additional flow needed to be passed into the Lower Bridge River channel above the amounts prescribed for the flow experiment (described above). The delivery of the higher flows began in 2016 and continued in 2017. Mean annual flows from the dam were approximately 22 and 19 $m^3 \cdot s^{-1}$ (peak flows = 97 and 127 $m^3 \cdot s^{-1}$) in 2016 and 2017, respectively (Figure 1.3). These high flow years are referred to as “Trial 3” in the context of the benthos analyses within this report.

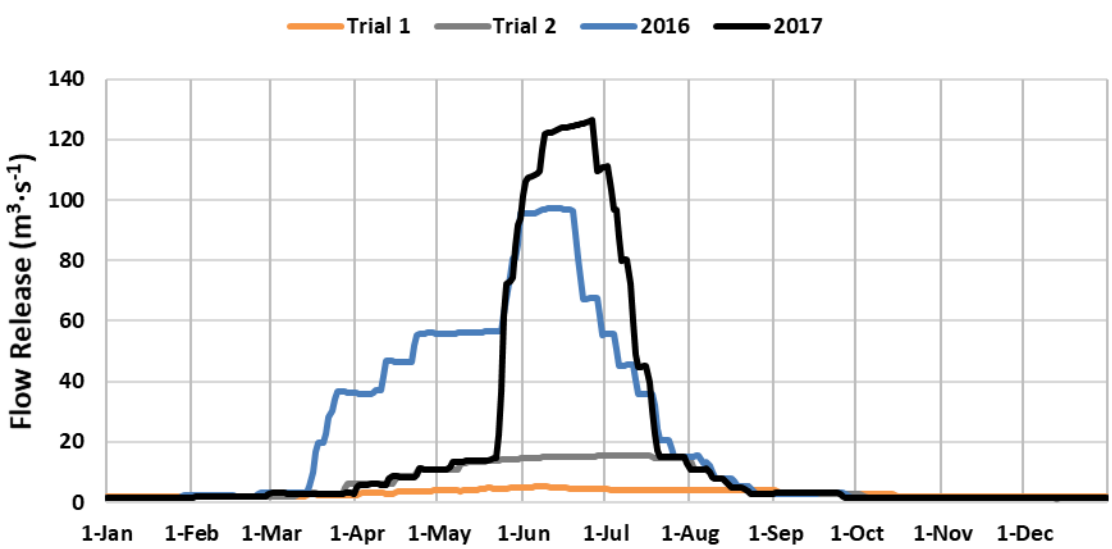


Figure 1.3 Hydrograph shapes for the high flows released from Terzaghi Dam into the Lower Bridge River channel in 2016 and 2017. Mean daily releases for the Trial 1 and 2 hydrographs are shown for context.

Peak flows in 2016 and 2017 were substantially higher than the Trial 1 and Trial 2 flow experiment hydrographs, but were within the range of spill flows from past events since the completion of Terzaghi Dam in 1960 (refer to Figure 1.1). The delivery of substantially higher flows in 2016 started in mid March, peaked in June, and returned to Trial 2 levels by the end of July (2016 high flow duration = 133 days). The high flows in 2017 had a higher peak, but a shorter duration relative to 2016: Flows increased above the Trial 2 hydrograph in the third week of May, peaked across the month of June, and was ramped back down to Trial 2 levels by the third week of July (2017 high flow duration = 59 days). The flow release during both high flow years was identical to the Trial 2 hydrograph shape from mid summer through fall and winter.

At least until the end of the current monitoring period (planned for 2021), spring flows could continue to be high and more variable across years than they were under the flow experiment trials. 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, reduce juvenile salmonid rearing habitat, 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 original objectives of the monitoring program were to reduce uncertainty about the expected long term ecological benefits from the release of continuous flows from Terzaghi Dam into the Lower Bridge River channel. This lack of certainty was an impediment to decision-making on an optimal flow regime and centred around the unknown effects of different flows on aquatic ecosystem productivity. A decision about flow release volumes and hydrograph shape based on invalid judgements would have implications for both energy production and the highly valued ecological resources of the Lower Bridge River. Therefore, the goal of the monitoring program was to resolve the uncertainty by the collection and analysis of scientifically defensible data.

1.4.1. Original (Core) Management Questions

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

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

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

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

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

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

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

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

   Inherent in the development of the Trial 2 hydrograph, was uncertainty regarding the risk of fish stranding given the relative magnitude of ramp-downs during the months when flows were reduced (i.e., August and October). Some information on the incidence of fish stranding between 8.5 and 2 $m^3 \cdot s^{-1}$ *had been documented during the Trial 1 period (Tisdale 2011a, 2011b). However, there was limited existing information on fish stranding in the discharge range from 15* $m^3 \cdot s^{-1}$ *to 8.5* $m^3 \cdot s^{-1}$ *and the types of habitats in this flow range. The collection of information on the risk of fish stranding at each stage of flow reduction between 15 and 1.5* $m^3 \cdot s^{-1}$ *will be useful for refining the descending limb of the Trial 2 hydrograph, or any alternative hydrograph that incorporates a similar flow range.*

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 in the context of the flow experiment.

1.4.2. Original (Core) Management Hypotheses

The original management hypotheses in the BRGMON-1 Terms of Reference were designed to use juvenile salmonid biomass as the primary indicator of the effect of the instream flow regime. Although originally conceived to apply to the 3 $m^3 \cdot s^{-1}$ (low flow) and 6 $m^3 \cdot s^{-1}$ (high flow) trials, these hypotheses can still be applied to the current higher flows by understanding them to mean that juvenile salmonid production (or other relevant metric as directed by the management questions) is either positively (H_O) or negatively (H_A) correlated with flow release volume from Terzaghi Dam. The management hypotheses are:

H_O: "High flow is better"
H_A: "Low flow is better"

1.4.3. Additional (High Flow) Management Questions

Due to the modified operations resulting from the La Joie Dam and Bridge River Generation issues, additional monitoring programs with new management questions were created to guide the short-term high flow monitoring programs and inform the LBR impact assessment and mitigation planning. This information will be used by the Technical Sub-Committee (TSC) charged with the monitoring and mitigation planning for the duration of the modified operations. As indicated in the BC Hydro Scope of Services document, it is noted that management questions have not been developed for the High Flow Monitoring component, a short-term program that examines water quality, erosion and other parameters exclusively during the high discharge periods.

High Flow Ramp Down Monitoring and Stranding Risk Assessment

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

1) Is the stranding risk during ramp downs at flows >15 $m^3 \cdot s^{-1}$ different than the stranding risk during ramp downs ≤15 $m^3 \cdot s^{-1}$?
2) Is the stranding risk equal across the reaches of the Lower Bridge River?
3) Does the stranding risk change when the maximum hourly ramping rate is greater than 2.5 cm?
4) Is the stranding risk equal on the left and right banks of the Lower Bridge River?

Two additional questions addressing stranding risk were created as part of the Emergence Timing, Residence, and Rearing Habitat monitoring program:

5) What are the potential incremental impacts of 2016-2017 flows on fish stranding in the Lower Bridge River during summer ramp downs from high flows?

6) What are the changes to the Adaptive Stranding Protocol recommended to manage fish stranding risks in the future?

Sediment and Erosion Monitoring

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

1) What is the post-2016 flow threshold that is likely to mobilize the remaining spawning size sediment?
2) What are the impacts of high flows on the level of sediment embeddedness and concentration of fines in the mesohabitats used by juveniles in the Lower Bridge River?

Emergence Timing, Residence, and Rearing Habitat

The flow release, drawn from the bottom of Carpenter Reservoir above Terzaghi Dam, has directly influenced the thermal regime of the LBR, affecting the incubation and emergence timing of Chinook salmon recruits under each of the flow trials to-date. In addition, the high flows delivered in 2016 and 2017 impacted juvenile salmonid rearing habitats by introducing higher velocities throughout more of the channel, and mobilizing sediment resulting in additional areas of scour and deposition. The effects of these changes were expected to include potential changes to rearing habitat area, displacement of fish out of the study area, and/or life history changes in the longer term. In response to (or anticipation of) these potential changes, the following management questions were developed:

1) What are the effects of warmer water temperatures associated with flow releases on the emergence timing of chinook and coho salmon in the Lower Bridge River?
2) What are the potential impacts of increased flows on fish rearing habitat in the Lower Bridge River?
3) What are the potential impacts of increased flows on residence and abundance of juvenile coho salmon and steelhead in the Lower Bridge River?
4) What proportion of the juvenile chinook salmon population rears in the Lower Bridge River, Fraser River or adopts an ocean-type life history?
5) Is the proportion of rearing in the Lower Bridge River, Fraser River and ocean influenced by flows in the Lower Bridge River?
6) How do temporary flow reductions during the freshet influence rearing habitat distribution/availability and use by juvenile fish?

1.4.4. Additional (High Flow) Management Hypotheses

Management hypotheses were created to accompany the management questions for the high flow ramp down monitoring and stranding risk assessment program (see above). These additional management hypotheses are:

H_{O1}: Stranding risk in the Lower Bridge River is independent of discharge.

H_{O2}: Stranding risk is equal across the reaches of the Lower Bridge River.

H_{O3}: Stranding risk is independent of the hourly ramping rate.

H_{O4}: Stranding risk is equal on the left and right banks of the Lower Bridge River.

1.5. Study Area

The Bridge River drains a large glaciated region of the Coast Range of British Columbia and flows eastward, eventually joining the Fraser River near the town of Lillooet. The river has been impounded by 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. 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 break designations are defined in Table 1.1. Monitoring for this program conformed to these reach break designations and has focused on the section of river between Terzaghi Dam and the bridge crossing upstream of Camoo Creek (i.e., reaches 4, 3 and 2). The overall study area is illustrated in Figure 1.4.

Table 1.1 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

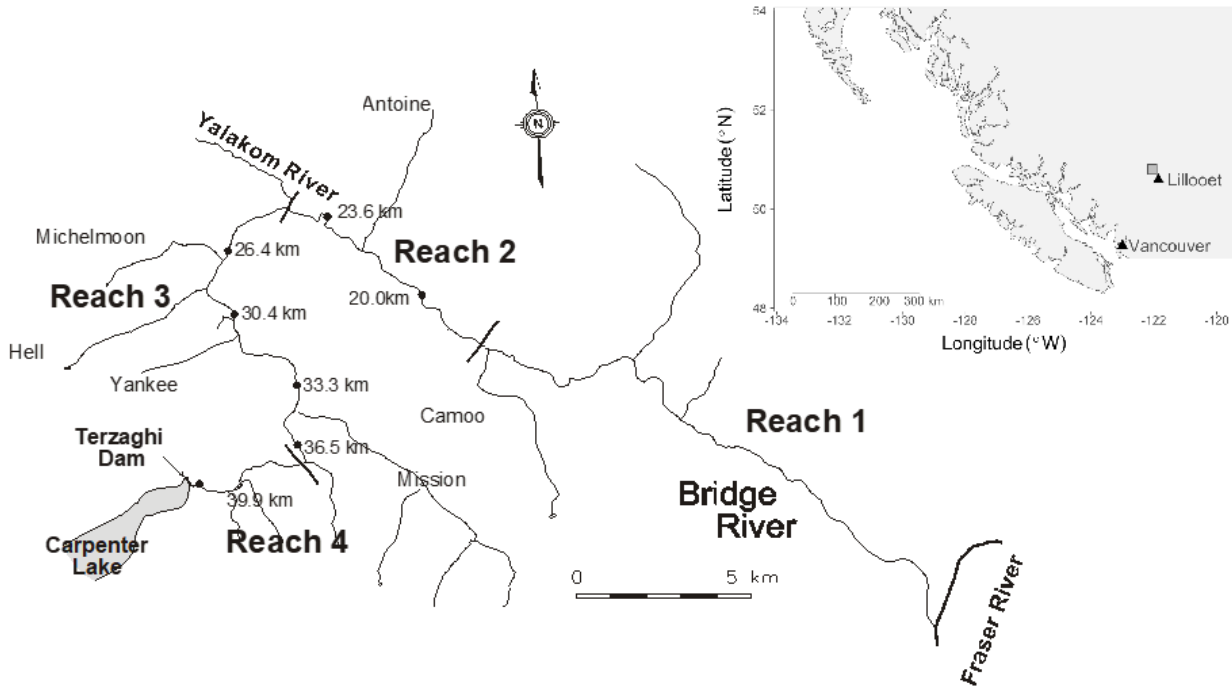


Figure 1.4 The Lower Bridge River downstream of Terzaghi Dam near Lillooet, British Columbia. Reaches are labelled 4 through 1 with increasing distance below Terzaghi Dam. Index sampling sites are labelled as distances upstream of the Fraser River and correspond to the following letters in some of the 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) and 20.0 km (G). The grey box in the top-right inset 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 are insignificant, so discharge is dominated by the release. Reach 3 was the groundwater- and tributary-fed reach extending down to the Yalakom confluence (length = 11.7 km). These inflow sources are relatively 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 most significant 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 = 4.3 $m^3 \cdot s^{-1}$). Other smaller tributaries 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

Field sampling in 2017 was conducted between February and December according to monitoring component (Table 1.2). 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 2017; however, comparisons or context from previous years and flow trials are included where relevant and available.

Table 1.2 Summary of data to be included in BRGMON-1 analysis and reporting for monitoring year 2017. Components that have prior years of data are noted.

Task	Components	2017 Period	Prior Years of Data[1]
Physical Parameter Monitoring	Water temperature; river stage; discharge	Year-round	1996 to 2016
Water Chemistry	Nutrients; alkalinity; pH	Oct to Nov	1996 to 2016
Primary & Secondary Productivity	Periphyton accrual; benthic invertebrate diversity & abundance	Oct to Nov	1996 to 2016
Juvenile Salmonid Growth	Monthly size data for juvenile salmonids	Apr; Aug to Nov	1996 to 2016
Juvenile Salmonid Abundance	Annual standing stock assessment	Sep	1996 to 2016
Habitat Surveys	Habitat unit classification & mapping	Aug to Sep	1996 to 2016
Ramp Down Monitoring	Stage monitoring; fish salvage	Jul to Sep	2011 to 2016
High Flow Monitoring	Kokanee entrainment; water quality sampling; sediment erosion & deposition; fish stranding site reconnaissance	Jun to Aug	2016
High Flow Ramp Down & Stranding Risk Assessment	Stage monitoring; fish salvage at flows >15 m^3/s	May to Jul	2016
Sediment & Erosion Monitoring	Sediment recruitment, movement & loss	Feb and Sep	2016
Emergence timing, residence & rearing habitat	Water temperature, emergence timing	Sep to Dec	

[1] Results of analyses for prior years of monitoring will only be included in this annual report where relevant for providing context to the 2017 results and where this could be supported by the project budget.

2. Methods

2.1. Core Monitoring Components

The purpose of the monitoring program was to document the effects of flow releases from Terzaghi Dam on key aquatic productivity metrics in reaches 2, 3, and 4 of the Lower Bridge River. Since a suitable control site was not available, the study design relies primarily on before-after comparisons among reaches within the study area. When the flow experiment and associated monitoring program were 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 Reach 2 were expected to be more muted. In other words, it was understood that differences or changes in measured variables in Reach 2 may result from factors other than (or in addition to) changes in the flow release from Terzaghi Dam.

The core methods employed for monitoring the effects of the Terzaghi flow releases in 2017 were generally consistent with those employed during the Pre-flow (Trial 0; 1996 to July 2000), Trial 1 (August 2000 to 2010), Trial 2 (2011 to 2015), and other High Flow (2016) periods. Four general monitoring activities were conducted: 1) continuous recording of flow release discharge, river stage and temperature; 2) assessment of water chemistry parameters, periphyton accrual, and aquatic invertebrate abundance and diversity during fall; 3) periodic sampling to monitor juvenile salmonid growth; and 4) a fall standing stock assessment to estimate the relative abundance and distribution of juvenile salmonids in the study area. Activities 1) to 3) were conducted at seven index sites located at approximately three kilometer intervals below 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)). Site A is located in Reach 4; sites B to E are in Reach 3; and sites F and G are in Reach 2. The fall standing stock assessment was conducted at 36 sites (during the Pre-flow period) and ~50 sites (during the two flow trials) distributed throughout the wetted portion of the study area.

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 in the fall only.

Field data collection for the Lower Bridge River Aquatic Monitoring Program (BRGMON-1) and most of the additional high flow monitoring components in 2017 were conducted by members of the Coldstream Ecology Ltd.-Bridge River Band Partnership. The 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.

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 benthos samples were collected	Target mean annual flow release from Terzaghi Dam ($m^3 \cdot s^{-1}$ ± SD)	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	0.25 ± 0.22
Trial 1	August 2000 – 2005	2, 3, 4	Spring Summer Fall	3 ± 5%	3.00 ± 1.03
	2006 – April 2011	2, 3, 4	Fall		
Trial 2	May 2011 – 2015	2, 3, 4	Fall	6 ± 5%	5.95 ± 5.39
Trial 3	2016 and 2017	2, 3, 4	Fall	No target [a]	20.2 ± 30.5

[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.

2.1.1. Discharge, River Stage and Water Temperatures

Discharge estimates were derived by a variety of means 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). Reach 3 flows were estimated using river stage data from pressure transducers located at the top and bottom ends of the reach (i.e., Rkm 36.8 and 26.1) that were converted to discharge using local rating curves. Reach 2 flows were the sum of the Yalakom inflows (based on data from Water Survey of Canada gauge 08ME025) and the estimated flows in Reach 3.

The relative stage of the river was continuously monitored and recorded at three stations (Rkm 20.0, 26.1, and 36.8) using PS9000 submersible pressure transducers (Instrumentation Northwest, Inc.) coupled to Lakewood 310-UL-16 data recorders. The stage data is logged every 15 minutes throughout the year, and the loggers are checked and maintained by Via-Sat Data Systems Inc. of Burnaby, BC. BC Hydro also maintains 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.

Water temperatures were recorded hourly throughout the year at each of the seven index sites using data loggers manufactured by Onset Computer Corporation (Model: UTBI-001). An additional temperature logger was deployed in the Yalakom River, approx. 100 m upstream of its confluence with the Bridge River. The temperature 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.

Surveys of hydraulic conditions were conducted at a range of discharges across the Pre-flow (Trial 0), Trial 1 and Trial 2 periods to evaluate the effects of the flow release on physical habitat conditions. The flow volumes assessed and survey dates are summarized in Table 2.2. There were replicate surveys completed at the Pre-flow, 1.5 $m^3 \cdot s^{-1}$, and 3 $m^3 \cdot s^{-1}$ release levels. Wetted widths and lengths of each habitat type (including cascades, runs, riffles, pools, rapids, and side channels) were measured with an optical rangefinder to assess changes in wetted area. Water depth and velocity (at 0.6 of depth) were measured using a top-set wading rod and velocity meter (Swoffer Instruments, Inc. Model 2100) at two or more locations along the thalweg in each habitat unit. Data were averaged by reach.

Table 2.2 Summary of flow release volumes and survey dates for habitat surveys conducted in the Lower Bridge River during the Pre-Flow, Trial 1, and Trial 2 periods.

Flow Period	Flow Release at Terzaghi Dam ($m^3 \cdot s^{-1}$)							
	0	**1.5**	**2**	**3**	**4**	**5**	**8**	**15**
Pre-Flow	Sep-96 Jul-00							
Trial 1			Oct-08	Oct-06 Oct-09 Sep-15[a]	Aug-00	Jun-07	Jul-07[a]	
Trial 2		Oct-13 Oct-14						Jul-14[a]

[a] Only reaches 3 and 4 were assessed during these surveys; Reach 2 was not completed or inaccessible due to unsafe wading conditions related to high flows.

2.1.2. Periphyton Biomass and Composition

Field Methods

Periphyton was sampled from riffles at each site (39.9 km (A), 36.5 km (B), 33.3 km (C), 30.4 km (D), 26.4 km (E), 23.6 (F) and 20.0 km (G)) (**Error! Reference source not found.**.4). Each site included three replicates. During Trial 0 only sites in Reaches 2 and 3 were sampled because Reach 4 was dewatered (Table 2.1). When the flow release began in August 2000, marking the beginning of Trial 1, sampling in Reach 4 began while sampling in Reach 2 and 3 continued. Sampling in all three reaches continued across Trials 2 and 3. In Trials 0 and 1, sampling occurred in spring (May and June), summer (August and September), and fall (October and November) while in Trials 2 and 3, sampling only occurred in the fall. The sampling locations had easy access and for consistency they were the same as those used for other ecological measurements reported by Bradford and Higgins (2001) and Decker et al. (2008).

Artificial substrata called “periphyton plates” were used to sample periphyton assemblages potentially supporting benthos in the river food web (Photo 2.1). Each plate was a 30 x 30 x 0.64 cm sheet of open celled Styrofoam (Floracraft Corp. Pomona Corp. 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 roughness, shape, and aspect of substrates.

Photo 2.1 Image of an installed periphyton plate.

Periphyton biomass was sampled weekly from each of three replicate plates that were installed at each site. The plates were submerged in riffles. In most years, water depth and velocity over each plate was recorded at the start of the sampling series when the plates were installed, and then again at the end prior to removal from the river. Each biomass sample consisted of a 2 cm diameter core of the Styrofoam and the 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. Each sample was packed on ice and frozen at the end of each sampling day at -15°C for later analysis. On the final periphyton sampling day of the series, one additional core was removed from each plate and preserved in Lugol's solution for taxonomic analysis. These samples were used to determine cell counts and biovolume per unit area for each of the identified algal taxa.

As in other flow trial years, periphyton samples in 2017 were collected ca. weekly at all seven index site locations, during one ca. eight-week monitoring series in the fall (i.e., 54 days between 6 October and 28 November 2017). A depth and velocity measurement was taken for each plate using a top-set wading rod and velocity meter manufactured by Swoffer Instruments, Inc. In 2017, these measurements were taken at the start of the sampling series only.

Laboratory Methods

The weekly periphyton biomass samples were submitted to ALS Environmental Laboratories where they were analysed for concentration of chlorophyll-a (also called chl-a) using fluorometric procedures reported by Holm-Hansen et al (1965) and Nusch (1980). The highest chlorophyll-a concentration from each plate was considered peak biomass (PB) for a given sampling time series. PB was the biomass metric used to define biomass accrued on a substratum. It was used along with other habitat attributes (Section **Error! Reference source not found.**) to find the most important variables contributing to variation in benthic invertebrate assemblages between trials.

The periphyton taxonomy samples were submitted to Danusia Dolecki at Limnotek for analysis. 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.

2.1.3. Habitat Attributes Potentially Driving Assemblages of Benthic Invertebrates

Attributes that may affect invertebrate assemblages were measured during times when invertebrate samplers were installed in the river. Algal peak biomass accrued during the sampling time series was one of these variables. It was measured using methods described in Section **Error! Reference source not found.**). Water temperatures were measured hourly throughout the incubation period by loggers deployed at each index site, and mean daily flow release from Terzaghi Dam was supplied by BC Hydro (see Section 2.1.1).

Conductivity and pH (measured with a handheld WTW probe), as well as concentrations of NH_4-N, NO_3-N, SRP, TDP, TP and total alkalinity (using standard grab sample methods) were measured at the start and end of each sampling time series. In 2017 this occurred on 2 October and 29 November. The grab samples were submitted to ALS Environmental in Burnaby, B.C. for analysis. The methodology employed for water sampling, as well as techniques used for laboratory analysis of nutrients, are described by Riley et al. (1997).

Water depth at each basket was measured using a top-set wading rod and velocity was measured using a Swoffer Instruments velocity sensor. The measurements were taken at the nose of the wire basket sampler at the start of the sampling series when the samplers were installed, and then again at the end prior to retrieval.

A binomial factor was used to identify sampling sites upstream and downstream of the Yalakom River, coded 0 and 1 respectively, to deduce influence of this tributary inflow on benthic invertebrate assemblages in the Lower Bridge River. Distance from river origin was measured using from digital maps. This distance was measured from the Terzaghi Dam to each sampling site in Reaches 4 and 3 and it was measured from the LBR-Yalakom River confluence for sites in Reach 2. This metric was a surrogate for potential recruitment of invertebrates from upstream.

Substrate composition data were retrieved from past habitat survey results (conducted throughout the study reaches during the Trial 0, 1, and 2 periods; refer to Riley et al. 1997 for a description of the habitat survey methods). For these surveys, the percent contribution of each substrate size class (i.e., boulder, cobble, gravel, and fines based on the scale described by Wentworth 1922) was qualitatively assessed for each habitat unit, including the index site locations. Comparable habitat surveys were not conducted in 2017 in order to update the substrate composition metrics since initiation of the Trial 3 high flows.

Each of these variables, described above, were considered potential candidates for use in analyses to determine the most important habitat attributes driving variation in assemblages of benthic invertebrates.

A potential biological driver variable was the presence or absence of spawning Pink salmon (*Oncorhynchus gorbuscha*) at a given site. This species of salmon typically returns to spawn on a bi-annual cycle with low abundance returns to the Fraser River watershed during even calendar years (e.g. 2012, 2014, 2016) and higher abundance returns during odd calendar years (e.g. 2011, 2013, 2015, 2017) (Crossin et al. 2003; Northcote & Atagi 1997). Annual pink salmon abundance during the study period was not available but given this bi-annual cycle, we accounted for the potential influence of pink salmon on the summer and fall benthic invertebrate community by coding pink salmon as a binomial factor, 0 for low abundance/even calendar years and 1 for higher abundance/odd calendar years. Summer and fall sampling directly overlapped with salmon spawning, which meant there was a pathway for direct effects of salmon-derived nutrients to the invertebrate community during these seasons. Spring sample timing preceded the salmon spawning period, so any influence pink salmon might have had would have been from the previous fall. The pink salmon coding was adjusted accordingly to account for the potential indirect effects to the spring invertebrate community.

2.1.4. Benthic Invertebrate Abundance and Composition

Field Methods

Three replicate benthic invertebrate samples were collected from the same sites and trial-season combinations used for the periphyton sampling (Table 2.1 and Section **Error! Reference source not found.**). 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 (**Error! Reference source not found.**.2), with 2 cm openings that was installed in the river for a period of approximately 8 weeks. The basket was similar to that shown by Merritt, Cummins, & Berg 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. To the extent possible, the sampling methods and equipment have remained consistent among all monitoring years to-date.

Photo 2.2 Basket sampler before installation in the Lower Bridge River

At the start of each colonization period, the baskets (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 (in 2017 this ran for 54 days from 6 October to 28 November). At the end of the sampling 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 Nitex screen (to remove excess water), transferred to a sample jar, and then preserved with a 10% formalin solution.

Following sample collection, the preserved invertebrates were submitted to Mike Stamford (Stamford Environmental) for sorting, identification (to Family), and enumeration.

Laboratory Methods

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 then washed through a coarse 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, Oligochaeta, Platyhelminthes, and Ostracoda. 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).

2.1.5. 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

During the flow experiment (Pre-flow, Trial 1 and Trial 2 periods), juvenile salmonids were collected during various sampling sessions spanning the growth season (e.g., April or May, June or July, August, September, and November) at each index site to enable analysis of spatial and temporal patterns of fish size (as a surrogate for growth information). During 2017, juvenile fish sampling was precluded during the high flow period (May to July), so size data were collected in August, September, October and November. The intention was to capture up to 30 fish from each juvenile age-class for the salmonid species available at the time of the surveys. Single pass, open-site backpack electrofishing was the sole fish sampling method employed to accomplish this goal. Forklength (mm) and weight (g) was recorded for each captured fish. However, sampling effort at each site in terms of sampled distance, area, or electrofishing seconds, was not recorded.

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 2.3). The selection of sampling sites was based on a habitat survey that was conducted in 1993 in reaches 2 and 3 that inventoried all major meso-habitat types. Eighteen sampling units in each reach were randomly selected 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 new sites were added to the upper region of Reach 3 in 1998. 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.

Table 2.3. 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)	# of Sites	Sampling Dates
1996	Trial 0 – Pre-Flow ($0\ m^3 \cdot s^{-1}$)	36	8 – 16 Oct
1997		36	2 – 13 Oct
1998		38	29 Sep – 9 Oct
1999		38	3 – 10 Sep
2000	Trial 1 ($3\ m^3 \cdot s^{-1}$)	50	30 Aug – 10 Sep
2001		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 [a]
2011	Trial 2 ($6\ m^3 \cdot s^{-1}$)	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 – High Flows (>$18\ m^3 \cdot s^{-1}$)	48	1 – 21 Sep
2017		49	5 – 20 Sep

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

At each site, the area to be sampled was enclosed with block nets constructed of 6 mm mesh. The average size of a sampled area was 97 m^2 (range: 20 to 273 m^2). Total catches were derived using a depletion method based on three or four passes of backpack electrofishing. A minimum of 30 minutes elapsed between passes. After each pass, captured fish were identified and forklength (nearest mm) and weight (0.1 g) of all salmonids were recorded before being released outside the enclosure. Ages (i.e., Age-0+, Age-1, etc.) were assigned to all captured fish according to identifiable size ranges based on analysis of length-frequency histograms for each reach.

During the Pre-flow period, nets were used to block off the full width of the stream in Reach 3; therefore, the sampled areas included the entire channel. This was not possible in Reach 2 during any monitoring year, or in reaches 3 and 4 after the flow release because of the greater depths and velocities associated with higher 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 both trial hydrographs and the two high flow years (2016 and 2017; see September period on Figure 1.3).

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. 1 month after the start of the flow release, was used to estimate the post-flow release distribution for reaches 3 and 4. The location of fish concealed in the substrate could not be determined by the daytime surveys, so the assumption was made that the distribution of fish observed during the microhabitat study would be a reasonable approximation of the location of all fish in the channel (either concealed in the substrate or swimming in the water column).

2.1.6. Adult Escapement

Adult spawner count data for the Lower Bridge River (up to 2016) were provided by Instream Fisheries Research (IFR) whom 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 in 2016 and 2017. Counts by helicopter overflight were conducted in all reaches during the Pre-flow period. Since the flow release began in 2000, visual surveys were conducted in reaches 3 and 4 by streamwalks due to the negative effect of glacially turbid water from Carpenter Reservoir on visibility conditions from the air. These data were extracted from an escapement database maintained by the Fisheries and Oceans Canada (DFO) office in Kamloops, BC.

Visual surveys 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 and recorded species and location. Viewing conditions, cloud cover, and lateral water visibility were also recorded (Burnett et al. 2017).

Visual counts occurred weekly for chinook and coho salmon in Reaches 3 and 4. In 2017, surveys started on August 19 for the salmon species, and continued until November 26 when fish activity ceased based on streamwalk, telemetry and resistivity counter observations. Surveys for steelhead were deemed ineffective in past years due to high turbidity and flows 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 conducted since 2011 (Burnett et al. 2017). 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, escapement prior to the flow release is 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, only a few years of data from the resistivity counter were available (Burnett et al. 2017).

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 (Burnett et al. 2017).

2.2. Additional High Flow Monitoring

Each of the core components described above are a part of the long-term monitoring to assess the effects of releases from Terzaghi Dam on the Lower Bridge River ecosystem across broad time scales and the range of flow treatments. However, increases in the maximum discharges from Terzaghi Dam above the trial flows was expected to have some additional impacts as well. To address some of the identified short-term effects, additional high flow monitoring was incorporated to supplement the BRGMON-1 program. This work was implemented under three new high flow monitoring programs for Trial 3: Surveys of kokanee entrainment from Carpenter Reservoir; High flow ramp down monitoring and stranding risk assessment; and Sediment and erosion monitoring.

High flow monitoring in 2017 also included spot measurement of air temperatures, water temperatures, total dissolved gas (TGP%), and turbidity (NTUs) at three selected locations in the LBR channel spread between the dam and the Yalakom River confluence (i.e., reaches 3 and 4). These locations were: the Terzaghi Dam plunge pool, Russell Springs (river km 30.4), and the Yalakom River confluence (rkm 25.5). Spot water temperature measurements were taken >60 cm depth in the water column, and TGP and turbidity monitoring were conducted according to BC Hydro protocols, as specified in the 2017 Lower Bridge River High Flow Monitoring Scope of Services.

Crews also conducted surveys to identify and assess bank erosion sites associated with the high flows. For each identified location, recorded parameters included: GPS coordinate, type of disturbance (e.g., landslide, gully, terrace, bank, etc.), location of disturbance, estimated canopy cover, origin (road, natural), degree of revegetation of disturbed area, sediment delivery to river, substrate composition (% dominant and subdominant substrate sizes), and approximate size/volume of erosion/deposition area. Photographs were taken at each location for reference (they are not included in this report, but can be provided upon request).

2.2.1. Kokanee Entrainment Surveys

To assess the incidence of kokanee entrainment from Carpenter Reservoir into the Lower Bridge River channel during the period of high flows (>15 $m^3 \cdot s^{-1}$), visual streamwalks were conducted to observe and enumerate kokanee (live and mortalities). The surveys were conducted by two technicians from the Coldstream Ecology Ltd.-Bridge River Band Partnership. Surveys were conducted on the following 7 dates in 2017: 11, 14, 16, 20, 22, 27, and 29 June. The survey area extended from the dam to approximately 1.5 km downstream on the river left side of the channel only (river right was not accessible due to the high flows). Each technician kept a separate tally for their portion of the surveyed area which were summed to generate the total number of kokanee observed for each survey.

All observed kokanee were assessed for fork length (mm), weight (g), condition and sexual maturity at the time of the survey. The data from these surveys provided confirmation that kokanee entrainment occurred during the 2017 high flow event, an index of the number observed on the survey dates, and some observations about the fish that were found. Based on these data, it is not possible to estimate total numbers of entrained fish (since observer efficiency was not assessed), determine the proportion of entrained fish that were live or mortalities, or determine the specific cause of the observed mortalities.

2.2.2. High Flow Ramp Down Monitoring and Stranding Risk Assessment

Stranding site reconnaissance and fish salvage data were collected as part of High Flow Ramp Down Monitoring at LBR discharges >15 $m^3 \cdot s^{-1}$. The methods described in this section come from 2017 weekly monitoring reports and documentation provided by Coldstream Ecology Ltd.-Bridge River Band Partnership, as well as the 2016 and 2017 High Flow Monitoring Reports (McHugh et al. 2017; O'Farrell and McHugh 2017).

Stranding Site Reconnaissance

Field reconnaissance during the ascending limb of the spring hydrograph identified potential new fish stranding sites in reaches 1, 2, 3, and 4 and assessed known strand areas for risk at higher flows. Crews scouted the river each time the flow release was ramped up a major step. Sites with an associated stranding risk that were identified during each particular flow stage, were subsequently revisited during ramp down to document how these locations dewatered and salvage fish, as required.

Crews assessed the potential stranding risk at each site using Broad-based stranding searches and Hot Spot stranding searches. Coldstream Ecology Ltd. described these as follows: Broad-based stranding searches consisted of walking dewatering or dewatered areas, flipping over large boulders and cobble within suspected areas of concern, with particular focus on areas that juvenile salmonids would rear in or potentially be stranded in. Hot spot stranding searches consisted of identifying and examining sensitive or high risk areas (low lying areas that would become isolated pools, large boulders and cobble which are specific habitats for juvenile fish). Crews also determined the vector escapement routes for return to the mainstem channel with close attention to depressions, potholes and mining areas, and assigned a low, medium or high stranding risk to each site, given the available habitat information. Stranding risk was rated High, Medium and Low based on the following criteria:

Stranding Risk Criteria:

Low: No stranding areas, dewaters slowly, multiple exits for fish to escape, constantly watered area.

Medium: Limited areas for stranding, limited exits for fish to escape, small pools become disconnected at flows ≤ 15 $m^3 \cdot s^{-1}$.

High: Multiple areas for stranding, dewaters quickly, mining holes, potholes, isolated pools, large depressions to trap fish from escaping into main channel, pools created by debris jams with no exits for fish to escape.

Ramp Down Monitoring and Fish Salvage

2017 discharge data for Terzaghi Dam and river stage data for Rkm 36.8 (~4 km downstream from the dam; aka 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, or confirm stranding risk at sites identified during the ramp up reconnaissance. Once the flow changes from the dam began, fish salvage crews were dispatched to the areas deemed to have the most immediate risk first, and then moved as the degree of risk shifted from location to location.

Site and habitat information was recorded for each identified stranding location on each ramping day, which included: Date, flow release rate at the dam, approximate river kilometre (upstream of the confluence with the Fraser River), GPS coordinates, bank, area (in m^2 based on length and width measurements), habitat type, substrate composition, and weather.

For fish salvaging, backpack electrofishing (EF) was the sole method employed. Parameters recorded for the fish salvaging included: Sampling effort (EF seconds), number of passes, stranding type (see below), species and age class (i.e., fry or parr), and number salvaged. Forklengths (in mm) were measured for the majority of salvaged fish.

For the first time, in 2017, 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 2017a), 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. As per the BC Hydro Bridge-Seton Fish Stranding Protocol, fish salvage types were defined as follows:

Isolated: fish in wetted areas that are isolated from the main flow of the river (e.g., strand pools);

Stranded: fish that are found in habitats that have completely dewatered, but are still alive when salvaged;

Mortality: fish that are found dead in habitats that are isolated or completely dewatered.

Analyses of the flow ramp down and fish salvage results were based 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 2017 were compared and combined with the 2016 data, the only other year where high flow ramp downs occurred, to better inform the risk of fish stranding at high flows.

Stranding and fish salvage data were also collected under the BRGMON-1 program at LBR discharges ≤15 $m^3 \cdot s^{-1}$. Data collected included the magnitude of the stage change, ramp rate, area of the site, salvage effort, habitat type, and the species, size and age class of fish salvaged. Data were combined with the salvage results from the High Flow Ramp Down Monitoring and Stranding Risk Assessment >15 $m^3 \cdot s^{-1}$, and incorporated into the stranding risk assessment analyses.

2.2.3. Sediment and Erosion Monitoring

The Lower Bridge River sediment and erosion monitoring associated with the high flows in 2016 and 2017 was conducted, analyzed and reported by Kerr Wood Leidal Associates Ltd. (Ellis et al. 2018, in draft). For detailed descriptions of the applicable methods and analyses for their work, please refer to the sections of their report summarized in Table 2.4. The sediment and erosion-related management questions (MQ) that were included in the Scope of Services for this report (see Section 1.4.3, above) are the same as MQ #2 and #5 from their report.

Table 2.4 Summary of directly relevant sections of the sediment and erosion monitoring report completed by Kerr Wood Leidal Associates Ltd. which pertain to the applicable management questions included in this report (Ellis et al. 2018, in draft).

Topic	Sub-category	Kerr Wood Leidal (KWL) Report	
		Section Number	Page Number
Management Questions (MQ)		1.3	1-1
Field Methods	Data Collection Sites	2.1	2-1
	Grain Size Measurements	2.2	2-4
	Topographic Survey	2.3	2-5
	Tracer Stones	2.5	2-7
	Juvenile Fish Shelter Assessment	2.6	2-10
Data Analysis Methods	Topographic Comparison	3.1	3-1
	Surface Grain Size	3.2	3-2
	Surface Sediment Mobility	3.3	3-3
	Tracer Travel Distance	3.4	3-6
	Sediment Transport Capacity Rate	3.5	3-7
	Reach-Scale Substrate Analysis	3.7	3-10
KWL MQ #2	Results	4.1	4-1
	Discussion	4.2	4-8
KWL MQ #5	Results	7.1	7-1
	Discussion	7.2	7-3

2.3. Data Analysis

2.3.1. Periphyton and Habitat Attributes

All statistical analyses were performed in the programming language R (R Core Team, 2016). The test of trial effects on various metrics (e.g. periphyton PB) was done using a one-way analysis of variance model in the R package aov{stats} (Chambers et al 1992). If the effect of trial was significant, we used Tukey HSD in the R package aov{stats} for post-hoc comparisons between trials.

2.3.2. Benthic Invertebrate Abundance and Composition

All statistical analyses were performed in the programming language R (R Core Team, 2016). To test for an effect of Trial and Season on family-level abundance of aquatic invertebrates we used a multivariate permutation of variance analysis (PERMANOVA; vegan::adonis), in the R package vegan (Oksanen et al., 2017). Homogeneity of multivariate dispersion was tested using the Bray-Curtis dissimilarity matrix for the invertebrate assemblage for each Season-Trial combination using vegan::betadispr with 999 permutations. Though PERMANOVA relies on homogeneity of multivariate dispersion, it is the most robust test compared to an analysis of similarities (ANOSIM) and the Mantel test when this condition is not met (Anderson & Walsh, 2013).

Ordination by non-metric multidimensional scaling (NMDS) was used to display dissimilarities among trials using Bray-Curtis dissimilarity matrices and up to 500 starts though the function stopped when it found two similar configurations with minimum Kruskal stress for two dimensions (k). A hierarchical cluster analysis using the R package pvclust (Suzuki & Shimodaira, 2015) was used to examine the groupings of similar invertebrate communities among seasons and trials using Ward's D clustering method on Euclidian distances. This method minimizes the within-group sum of squares (Bocard, Gillet, & Legendre, 2011) and provides p-values for hierarchical clusters based on multiscale bootstrap resampling (Suzuki & Shimodaira, 2015). To determine which families contributed to the dissimilarities among invertebrate assemblages in each Season-Trial combination, a multivariate similarity percentages (SIMPER) procedure with Bray-Curtis dissimilarities was used. This procedure calculates the overall contribution of each family to the Bray-Curtis dissimilarity.

Finally, we used a redundancy analysis to determine the response of the benthic invertebrate community to ecological variables in each Season-Trial combination by reach. By including reach, we could test for a spatial gradient with seasonal variations in flow and flow release increases with each trial. A redundancy analysis is analogous to a multiple linear regression followed by a principal component analysis (PCA) (P. Legendre & Legendre, 2012a). This approach estimates the amount of variation in the standardized total family abundance data matrix **Y** that is explained by the standardized matrix of ecological variables, **X** (P. Legendre & Legendre, 2012a). For this study, **Y** was a $\log_2$ transformed matrix of 60 families for each

Season-Trial combination and grouped by reach and **X** was a matrix of ecological variables. The ecological variables were those that may directly affect assemblages of benthic invertebrates and were not correlated with each other. In an RDA figure, the ecological variables are represented by arrows; longer arrows being more correlated with the redundancy axes and therefore more related to the variation in the community data matrix (Braak, 1987; P. Legendre & Legendre, 2012b). Ellipses were used to identify the relationship between the ecological variables at the reach level and the family assemblages (Braak, 1987; P. Legendre & Legendre, 2012a). We performed a permutation test on the ecological variables ($\alpha = 0.05$, $\beta = 0.01$ and 999 permutations) in each Season-Trial combination to test whether linear relationships between the family abundance data in **Y** and the ecological variables in **X** existed.

2.3.3. Juvenile Fish Production: Size

We evaluated effects of flow on juvenile salmonid growth based on weight samples taken during the annual fall stock assessment. 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 2017; see Table 2.1, 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. Thus, using weight data to make inferences about growth is problematic, especially for chinook. Nevertheless, we computed average weight for each reach and flow treatment, and for the 2016-2017 high flow period. This analysis was done 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.

2.3.4. Juvenile Fish Production: Abundance & Biomass

The abundance and biomass of juvenile salmon in each reach was estimated with a hierarchical Bayesian model (HBM) described in Bradford et al. (2011) and Appendix A. 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 A. 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 (Pre-flow), 3 (Trial 1), and 6 $m^3 \cdot s^{-1}$ (Trial 2), as well as the unplanned high flows which began in 2016.

As described in detail in Appendix A, 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 2.5.

Table 2.5. Range of years used to compute average abundance and biomass for each flow treatment in the Lower Bridge River for chinook, coho, and mykiss fry (Age-0+) and mykiss parr (Age-1).

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

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+ abundance in 2016 and 2017 were used to computed the average abundance and biomass for the High flow regime.

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), and Trial 2 to High flow (i.e., 2016) were also not included in the treatment averages for mykiss parr.

2.3.5. 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.

Escapements 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 (Burnett et al. 2017). Escapement estimates for these species represent abundance in reaches 3 and 4 only as this is where the stream walks were 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 2016 produced fry that were sampled in the fall of 2017). 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 the Pre-flow period (0 $m^3 \cdot s^{-1}$ release), $\lambda_{j=1}$ 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 and 2017), 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. Core Monitoring Components

3.1.1. Discharge, Wetted Area and Water Temperatures

Among the various flow treatments, there has been strong contrast in the physical factors that were expected to be important for all trophic levels (i.e., algae, benthic invertebrates, and fish): flow, wetted area, velocity and water temperature. The high flow releases in 2016 and 2017 resulted in greatly increased discharges in spring and summer relative to the previous trial flows (Figure 1.3 in Section 1.3). Outside of the high flow release period (i.e., in early spring, late summer, fall and winter), discharges were equivalent to the Trial 2 releases. Peak flows in 2016 and 2017 (i.e., 97 and 127 $m^3 \cdot s^{-1}$, respectively) were 6.4- and 8.5-fold higher than Trial 2 peak flows, and mean annual flow was 3.6- and 3.1-fold higher than the Trial 2 average, respectively.

Due to minimal tributary and groundwater inflows in reaches 4 and 3 relative to the magnitude of the release, site-specific discharges were very similar across those reaches (Site A – 39.9 km to Site E – 26.4 km), differing by a maximum of ~3 $m^3 \cdot s^{-1}$ (or 2%) across that distance (Figure 3.1). Due to the contribution of the Yalakom River, site-specific discharge at locations in Reach 2 were up to 20 $m^3 \cdot s^{-1}$ (or 14%) greater than release flows.

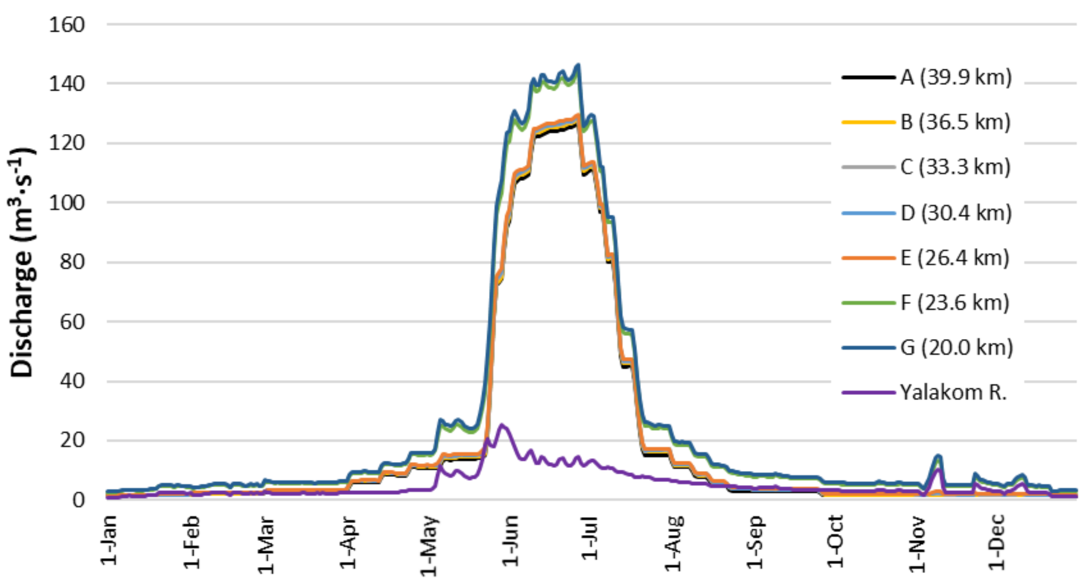


Figure 3.1 Site-specific discharge estimates (shown as mean daily values) in the Lower Bridge River during 2017. Site A is in Reach 4, sites B to E are in Reach 3, and sites F and G are in Reach 2. 2017 Yalakom River discharge is also shown.

Based on past habitat survey data collected across the flow ranges observed during the Trial 0, 1 and 2 periods, increases in wetted area were not proportional to increases in flow due to the generally constrained nature of the channel throughout much of the study area (Figure 3.2). In Reach 4, the most significant gains of wetted area occurred by the rewetting of that reach when the flow release was initiated. On average, just over 2 hectares (ha) per km of wetted area were added between 0 and 1.5 $m^3 \cdot s^{-1}$. In Reach 3, the greatest gains of wetted area were between discharges of 1 $m^3 \cdot s^{-1}$ (pre-release flow in that reach) and ~7 $m^3 \cdot s^{-1}$, where ~1 ha/km was added. Above these discharges, gains of wetted area per increment of flow increase were diminished in each of the study reaches, but particularly Reach 3. Habitat surveys for the Trial 3 high flows (> 15 $m^3 \cdot s^{-1}$ Terzaghi Dam release) have not been completed.

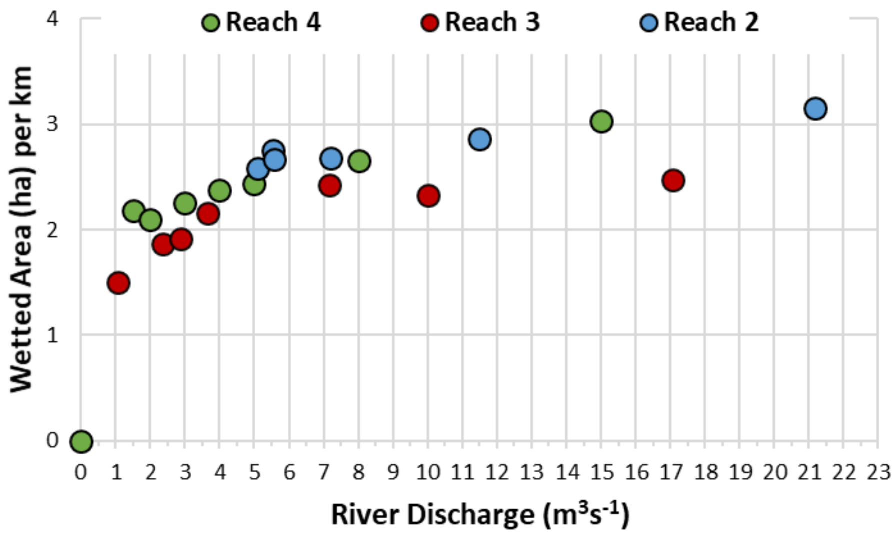


Figure 3.2 Mean wetted area per kilometer of channel length as a function of discharge for reaches 4, 3, and 2 of the Lower Bridge River.

In contrast to the wetted area-discharge relationship, mid-channel velocities followed more of an S-shaped curve with increases in discharge (Figure 3.3). Mean mid-channel velocity increased from 0.23 m/s under pre-flow conditions in Reach 3, to 0.73 m/s at 3.6 $m^3 \cdot s^{-1}$ discharge. Between ~4 and 15 $m^3 \cdot s^{-1}$ increases in mid-channel velocity appeared to be less per increment of flow change; however, there are limited data points for any of the reaches above 6 $m^3 \cdot s^{-1}$ due to challenges and safety concerns related to wading into the river at those flows. The single mean value for Reach 3 at 17.1 $m^3 \cdot s^{-1}$ reflected another marked increase at the high flow end of the range (i.e., >15 $m^3 \cdot s^{-1}$). This value (2.0 m/s) was up to 4-fold greater than the low flow values.

Relative to the Pre-flow period (Trial 0), dam releases have caused water temperatures to be cooler in the spring months, and warmer in the fall (Figure 3.4). These effects were most evident in reaches 4 and 3, with a gradient of effect associated with proximity to the dam. In addition to continuation of these effects, Trial 3 flows in 2016 and 2017 also resulted in warmer temperatures during the period of the year when the high flows were delivered, particularly in June and July. This difference was evident in all three study reaches.

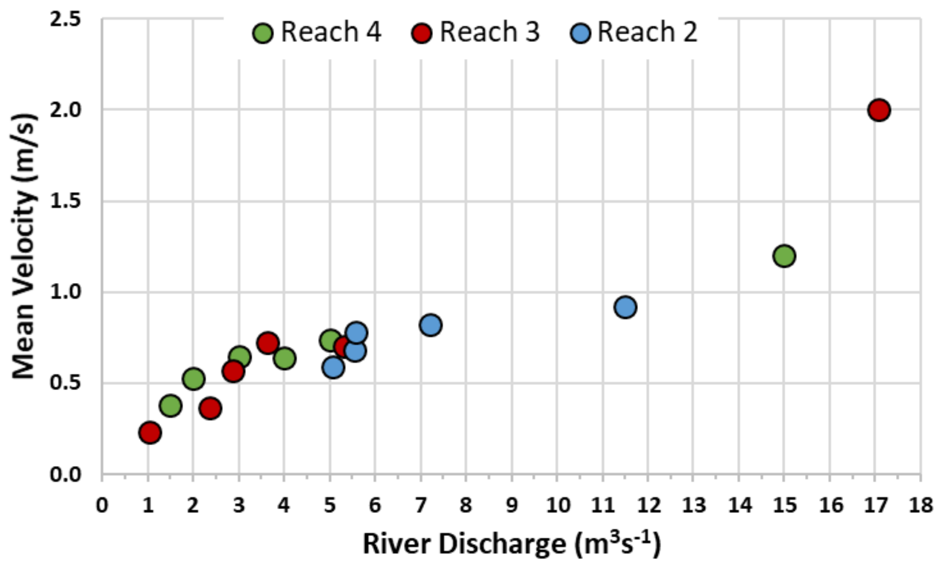


Figure 3.3 Mean mid-channel velocities as a function of discharge for reaches 4, 3, and 2 of the Lower Bridge River.

Given that the warmer temperatures were evident at the Reach 4 monitoring location, which is approx. 800 m downstream of the dam, it seems likely that this effect was directly related to the flow releases in 2016 and 2017 (as opposed to just ambient influence on the river itself). This may have been caused by the interaction of ambient temperatures with particular operational characteristics of Carpenter Reservoir (i.e., upstream of the dam) in those two years; however, the analyses required to determine the cause of the increased temperatures are beyond the scope of this report.

Differences among the Trial 1, 2 and 3 flows during the salmon incubation period in fall were small, though release temperatures were slightly higher in 2016 from 1 November to 9 December of that year. Changes to the thermal regime have caused large differences in the predicted timing of juvenile salmon emergence from the spawning beds. Prior to the flow release the predicted median date of both coho and chinook salmon fry emergence was 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).

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 near the dam. The 0.5 $m^3{\cdot}s^{-1}$ reduction in October-January flows under Trial 2 compared to Trial 1 (Figure 1.2) resulted in a slight delay in predicted emergence timing (i.e., slightly closer to the Pre-flow timing). Due to later spawn timing, the impact of the flow release on coho salmon emergence timing was much smaller with emergence predicted to be advanced by less than 15 days at most locations (Figure 3.5).

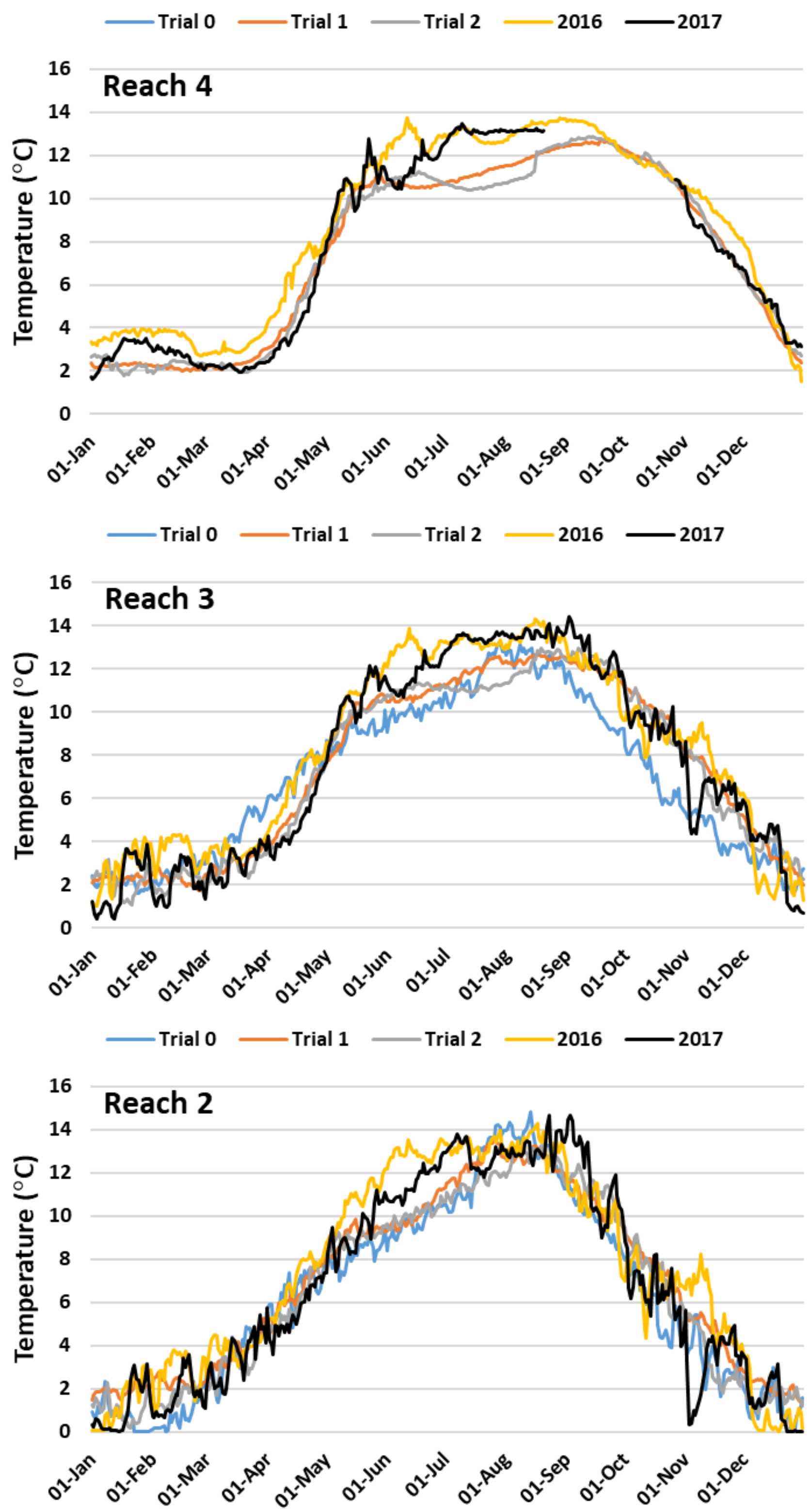


Figure 3.4 Mean daily water temperatures during Trial 0 (pre-flow), Trial 1 (3 $m^3 \cdot s^{-1}$), Trial 2 (6 $m^3 \cdot s^{-1}$), and the high flow years (2016 and 2017) for Reach 4 (top), Reach 3 (middle) and Reach 2 (bottom). Gaps indicate periods when data were missing.

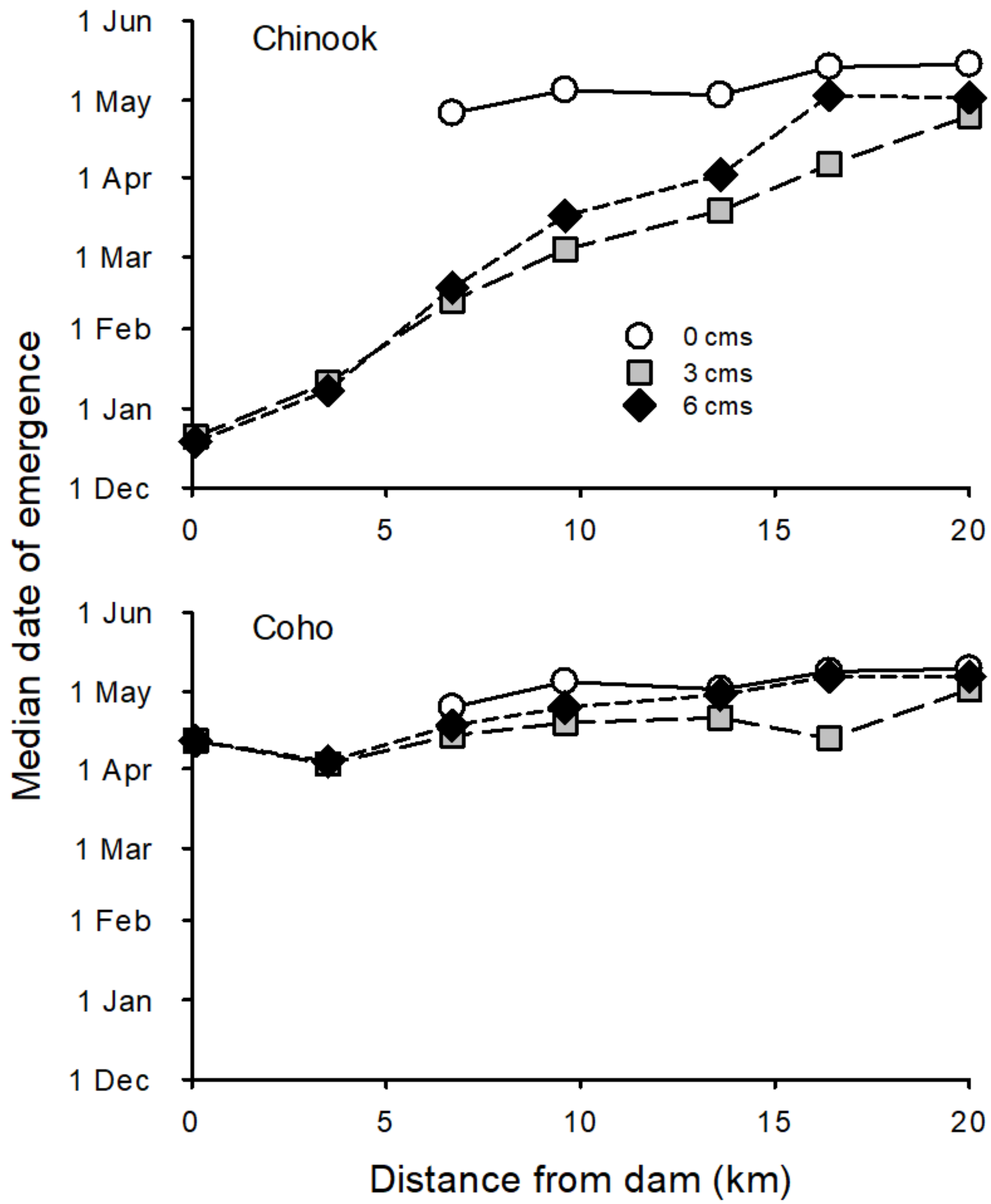


Figure 3.5 Predicted median emergence dates for chinook (top) and coho (bottom) salmon fry at varying distances below Terzaghi Dam based on observed average temperatures for each flow treatment. Since fall flows in 2016 and 2017 were the same as Trial 2, the predicted emergence dates by location were expected to be similar. The breaks between reaches 4-3 and 3-2 are at ~3 and ~15 kms downstream of the dam, respectively.

3.1.2. Periphyton Composition

Periphyton algae was comprised mostly of diatoms with trace numbers of Cyanophyta and rare counts of all other taxa among all trials (FIGURE Figure 3.6). In Trials 0 and 1, the common diatom genera included *Achnanthes, Amphipleura, Cocconeis, Cymbella, Cyclotella, Diatoma, Epithemia, Fragilaria, Gomphonema, Hannaea, Melosira, Meridion, Navicula, Nitzschia, Rhoicosphenia, Rhopalodia, Synedra*, and *Tabellaria.* All of these taxa are commonly encountered in cool mountain streams without pollution. The cyanophytes during Trial 0 included *Merismopedia* sp., *Oscillatoria* sp., and *Anabaena* sp. In Trial 1, these same cyanophytes and *Aphanocapsa*, *Gloeocapsa*, and *Lyngbya* were found. In Trial 2, most of the common diatoms found in Trials 0 and 1 were found as well as *Didymosphenia* sp., *Stauroneis* sp. The cyanophytes at that time included the same genera found earlier and *Arthrospira* sp., *Aphanizomenon* sp. and *Pseudanabaena* sp. In Trial 3, diatom genera were again common and included *Achnanthes, Amphipleura, Asterionella, Cocconeis, Cyclotella, Cymbella, Cyclotella, Diatoma, Didymosphenia, Eunotia, Fragilaria, Gomphonema, Melosira, Navicula, Nitzschia, Rhopalodia, Rossithidium, Stauroneis,* and *Synedra.* In Trial 3 the only cyanophytes present were *Oscillatoria* sp. and *Anabaena* sp.

It is noteworthy that several of the cyanophytes present in the Lower Bridge River are known to produce toxins that are collectively known as microcystins:

> http://healthycanadians.gc.ca/publications/healthy-living-vie-saine/water-cyanobacteria-cyanobacterie-eau/index-eng.php.

Those taxa were *Anabaena*, *Aphanizomenon*, and *Oscillatoria*. Given that these taxa were rare suggests that microcystins from them would be present, at most, in trace concentrations and well below health guidelines. In Trial 2, two Chryso-cryptophytes were found (*Chroomonas* sp. and *Cryptomonas* sp.) (Figure). These taxa are flagellated unicells not common to stream substrata. They likely came from Carpenter Reservoir, where they are known to occur (Perrin et al. 2016) and must have been trapped on the periphyton sampling substrata in the Lower Bridge River.

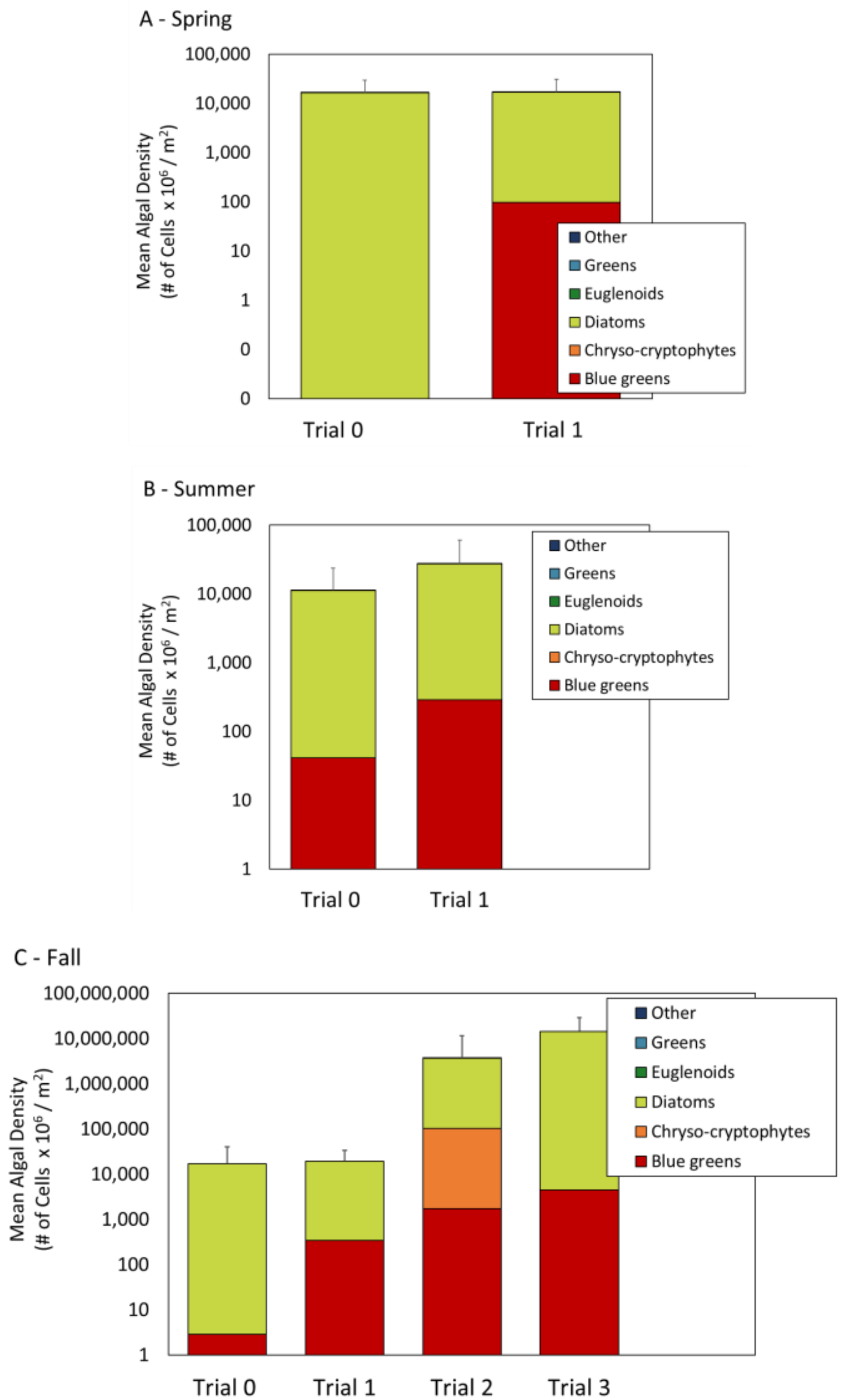


Figure 3.6 Mean algal cell density (±standard deviation) among all reaches by trial in each of spring (top), summer (middle), and fall (bottom).

For the fall sampling that occurred in all trials, Figure shows that algal cell density was the same between Trials 0 and 1 ($p=0.72$), it increased by more than two orders of magnitude in Trial 2 ($p<0.001$), and it increased by another four times in Trial 3 ($p<0.001$) (note the log axis in Figure). Spring and summer periphyton densities that were only measured in Trials 0 and 1 were approximately the same as those found in the fall during the same trials, which implies no season effect during those earlier trials. The large increases in algal cell density between Trials 1 and 2 and between Trials 2 and 3 were statistically significant ($p<0.001$ for each contrast), which showed a trial effect on periphyton cell densities was present. All of these changes between Trials 1 and 2 and between 2 and 3 were mainly due to the diatoms with minor contributions from the cyanophytes and rarer taxa. All reaches contributed to the trial effect in the fall but Reach 4 contributed most and Reach 2 contributed least to the differences in algal cell densities between Trials 2 and 3 (Figure).

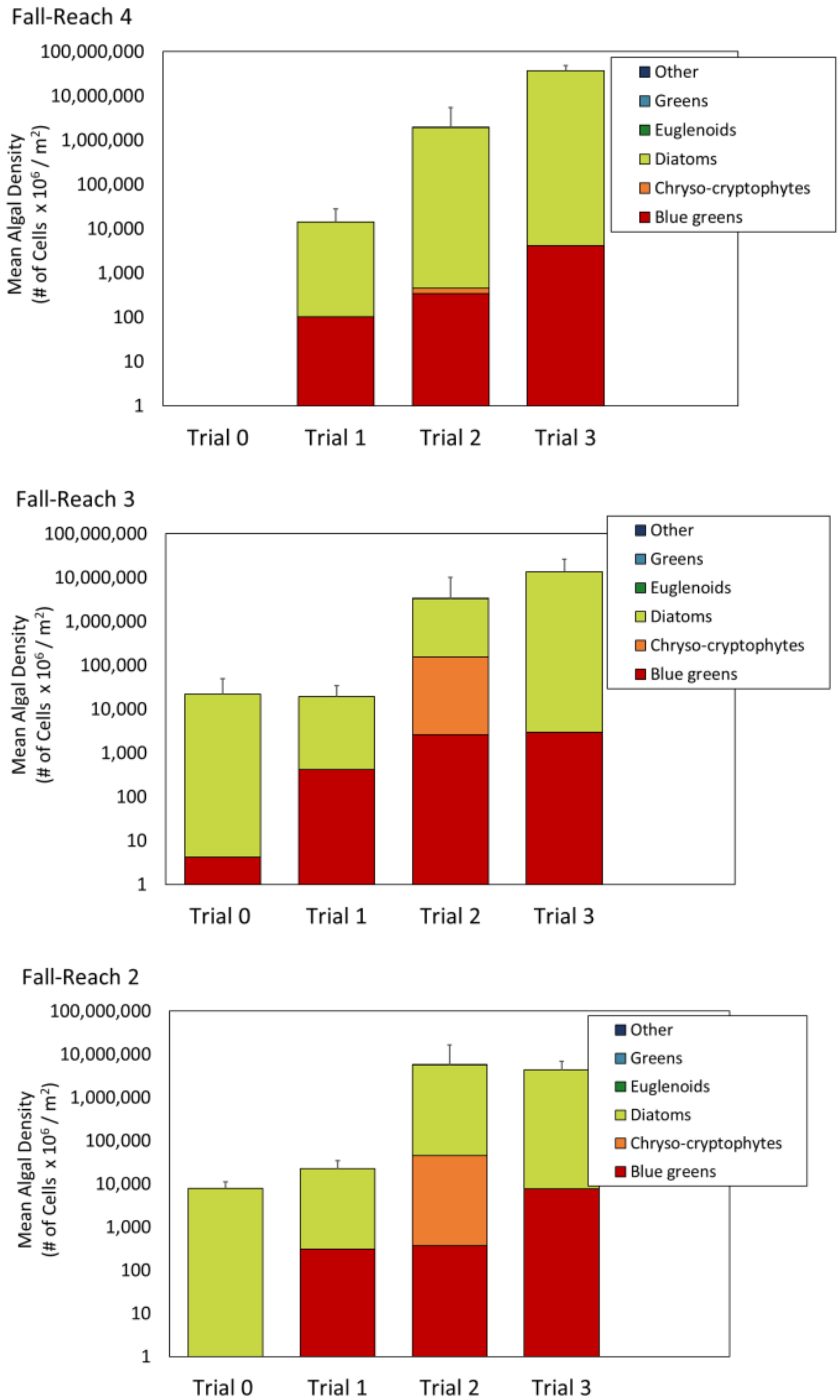


Figure 3.7 Mean algal cell density (±standard deviation) in the fall by trial in each of Reach 4 (top), Reach 3 (middle), and Reach 2 (bottom)

3.1.3. Habitat Attributes

Among all possible habitat attributes that were measured (Section **Error! Reference source not found.**), a subset was selected that could directly affect invertebrate assemblages and were not intercorrelated. This process eliminated conductivity, alkalinity, and nutrient concentrations that may directly affect periphyton but not invertebrates. We selected water depth and velocity at each specific invertebrate basket rather than flow even when calculated on a site-specific basis because benthos is exposed to and may respond to variation in water depth and velocity which are driven by flow. Percent boulder, cobble, gravel, fines, and bedrock were eliminated because use of a standard stone size in the baskets eliminated variation in particle size that may affect benthos assemblages. Distance from origin was eliminated because it was redundant with coding for the influence of the Yalakom River inflow. Following tests for co-linearity no remaining variables were eliminated. Remaining variables used in analyses to examine links between habitat attributes and benthic invertebrate assemblages were algal PB, water depth at the sampler, water velocity at the sampler, water temperature, influence (coded 1) or no influence (coded 0) from the Yalakom River, and pink run on (coded 1) or off (coded 0).

Mean values by season and trial for the continuous habitat variables are listed in

Table . Coding for influence of the Yalakom River and coding for influence of pink salmon runs are not shown because they are switches rather than being continuous. Peak periphyton biomass (PB) was the same between Trial 0 and Trial 1 in the spring ($p = 0.57$) and summer ($p=0.05$) but in the fall PB increased with each successive trial beginning at 4.9 µg chlorophyll $a \cdot cm^{-2}$ in Trial 0 and reaching a mean biomass of 46 µg chlorophyll $a \cdot cm^{-2}$ in Trial 3. These changes were highly significant ($p<0.001$) and they matched the increases in algal cell densities in the fall between trials shown in Figure and Figure . Water depths between trials were similar in the spring ($p=0.9$) but in summer they were greater in Trial 1 than in Trial 0. In the fall, water depths at the samplers were less in Trial 0 than in all of the other trials ($p<0.001$). Water velocity at the samplers was the same between trials in the spring ($p=0.7$) but in summer it was greater in Trial 1 (mean of 0.37 $m \cdot s^{-1}$) than in Trial 0 (mean of 0.29 $m \cdot s^{-1}$). In the fall, water velocity at the samplers was lower in Trial 3 (mean of 0.15 $m \cdot s^{-1}$) than in the other trials (means of 0.27-0.32 $m \cdot s^{-1}$) ($p<0.03$). Water temperature at the samplers in the spring was greater in Trial 1 than in Trial 0 ($p<0.001$) but there was no temperature difference between trials in the summer (approximately 12°C). In the fall, mean temperature was 7.7 to 9.1°C among trials. Only in Trials 1 and 2 were invertebrates exposed to different mean temperatures ($p=0.001$).

Table 3.1 Mean values (±standard deviation) of habitat attributes potentially driving variation in assemblages of benthic invertebrates.

Season	Trial	Mean PB[a] (µg chl-a ·cm^{-2})	Mean water depth at basket sampler (m)	Mean velocity at basket sampler (m·s^{-1})	Mean water temperature at basket sampler (°C)	# of samples upstream and downstream of the Yalakom River	# of samples during low and high pink years	Sample Size
Spring	0	3.8 ± 3.8	0.33 ± 0.11	0.40 ± 0.12	9.3 ± 1.0	20/8	11/17	28
	1	4.1 ± 3.6	0.33 ± 0.08	0.38 ± 0.13	10.4 ± 1.4	25/10	14/21	35
Summer	0	4.0 ± 3.7	0.29 ± 0.07	0.29 ± 0.10	11.8 ± 2.2	16/6	11/11	22
	1	3.2 ± 2.0	0.33 ± 0.09	0.37 ± 0.1	12.3 ± 0.8	30/12	21/21	42
Fall	0	4.9 ± 3.4	0.17 ± 0.04	0.32 ± 0.13	8.1 ± 2.0	12/6	6/12	18
	1	8.5 ± 5.2	0.28 ± 0.08	0.33 ± 0.12	9.1 ± 1.5	45/18	35/28	63
	2	27.2 ± 42.6	0.31 ± 0.08	0.27 ± 0.10	7.7 ± 1.7	25/10	14/21	35
	3	46.0 ± 25.6	0.33 ± 0.08	0.15 ± 0.11	7.7 ± 1.6	10/4	7/7	14

[a] PB is peak biomass of algae accruing on substrata installed in the river.

3.1.4. Benthic Invertebrate Abundance and Composition

Benthic invertebrates from the Lower Bridge River were from the orders Ephemeroptera, Plecoptera, Diptera and "Other" taxa including Oligochaeta, ostracods, Hemiptera and other true bugs (Figure 3.). The most abundant families by order across all seasons and trials were Baetidae (47.4% of ephemeropterans), Heptageniidae (26.2% of ephemeropterans), Nemouridae (45.2% of plecopterans), Perlodidae (23.5% of plecopterans), Chironomidae (74.0% of dipterans) and Simuliidae (24.5% of dipterans). Most of the "Other" taxa were oligochaetes from the family Naididae.

Total mean invertebrate abundance increased by 12% between Trial 0 and 1 in the spring despite a decline in mean Chironomidae abundance per sample (Figure 3., Plot A). The increase in total mean invertebrate abundance coincided with a mean annual flow increase from 0 m^3·s^{-1} to 3 m^3·s^{-1} and was driven by significant (α = 0.05) increases in Ephemeroptera ($p<0.001$) and Other taxa (p = 0.020). There was also a 35% increase in mean invertebrate abundance between Trial 0 and 1 in the summer, despite a significant decline in Plecoptera ($p<0.001$) (Figure 3., Plot B). The greatest increases between these trials were observed in trichopterans (p = 0.009) and Other taxa (p = 0.004). In the fall, there was a significant increase in Ephemeroptera between Trial 0 and Trial 1 (p = 0.001) and an increase in Other taxa between Trial 1 and Trial 2 ($p<0.001$). Declines in abundance of Chironomidae and other Diptera offset those increases, resulting in no overall change in mean invertebrate abundance between trials 1 and 2 in the fall (Figure 3., Plot C). Total invertebrate abundance declined by 64% between Trials 2 and 3 in the fall (p=0.001: Figure 3., Plot C). The greatest decline in abundance was found among the Chironomidae (p=0.02), Ephemeroptera ($p<0.001$), Tricoptera ($p<0.001$), and

Plecoptera (p=0.03). The only order that showed no significant change was the Diptera other than Chironomidae.

The change in abundances in the fall occurred in different ways among reaches (Figure). Reach 3 supported relatively high invertebrate densities during Trial 0 (about 5 times that in Reach 2) but with the onset of flow release in Trial 1, the densities in Reach 3 declined while all invertebrate orders became established in relatively high densities in Reach 4 and densities in Reach 2 increased by approximately 5 times. With the greater flow release in Trial 2, densities further increased in Reach 4, and stayed about the same in Reach 3 and 2. With the spill flows during Trial 3, densities of all taxa declined in all reaches with greatest effects in Reach 4 and Reach 2.

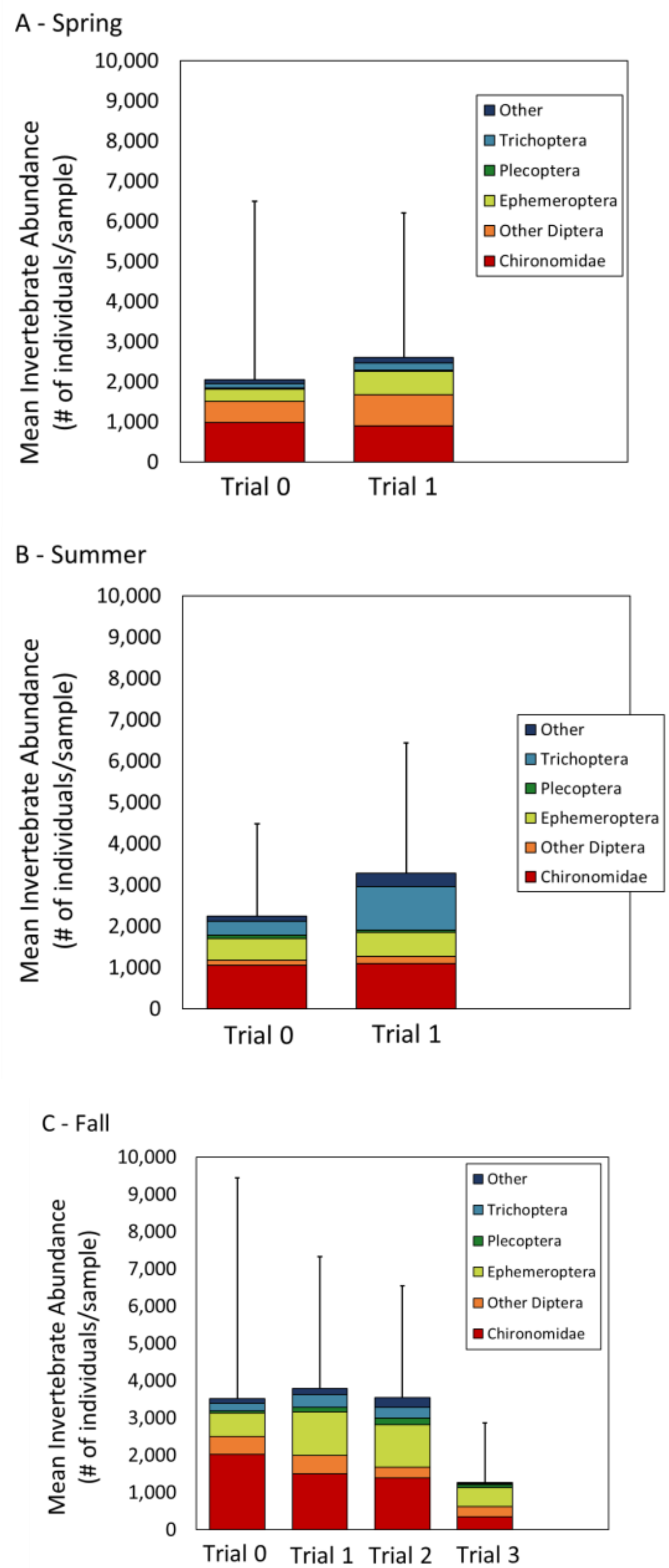


Figure 3.8 Mean abundance (±standard deviation) of invertebrate orders among all reaches by trial in each of spring (top), summer (middle), and fall (bottom).

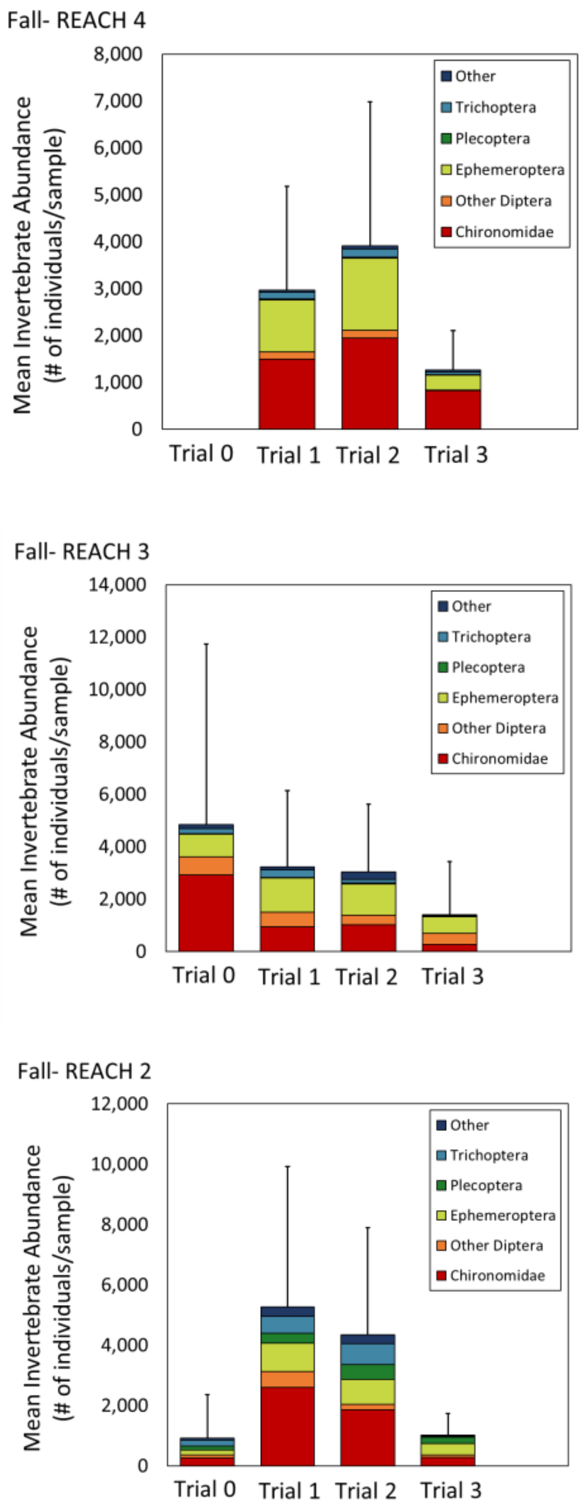


Figure 3.9 Mean abundance (±standard deviation) of invertebrate orders in the fall in each of Reach 4 (top), Reach 3 (middle), and Reach 2 (bottom).

The permutational multivariate analysis of variance (PERMANOVA) revealed significant differences between invertebrate family assemblages among trials within each season (p=0.03 for spring, p=0.01 for summer, p=0.001 for fall). These differences are apparent in Figure where polygons outlining family assemblage data for each site within each trial did not overlap except for one site in the spring (Site F, Trial 1; Figure , Plot A) and four sites in the fall (Sites C and D, Trial 0 and Sites B and G, Trial 1; Figure , Plot C) . Figure also highlights the dissimilarity between invertebrate communities among sites within trials; sites that were further apart were less similar than sites that were closer together. For example, the invertebrate assemblages at sites downstream of the Yalakom River (F and G, 23.6 km and 20.0 km, respectively, **Error! Reference source not found.**.4) were typically more similar to each other than sites upstream of the Yalakom (A through E, 39.9 km to 26.4 km, **Error! Reference source not found.**.4). As mean annual flow increased between trials, there was also a contraction of the polygons, implying that invertebrate communities at sites within trials became more similar as flow increased (Figure). This contraction in ordination space was particularly evident in the fall, where the invertebrate assemblage between sites was more similar in Trial 2 than between sites in Trial 0 or Trial 1. Some divergence occurred in Trial 3 when dissimilarity of assemblages between sites increased compared to that in Trial 2. There was a gradient of change in invertebrate communities in the fall corresponding to a change in flow between Trials. This gradient increased between Trial 2 and 3 (polygons far apart in ordination space) compared to Trials 0, 1, and 2 (polygons closer together in ordination space) (Figure , Plot C).

Hierarchical cluster analysis confirmed that the invertebrate communities grouped by trials and season, with spring communities being more similar to summer communities than fall communities (Figure). One exception were community assemblages sampled in fall Trial 3, which were more similar to spring assemblages than the other fall assemblages (Figure).

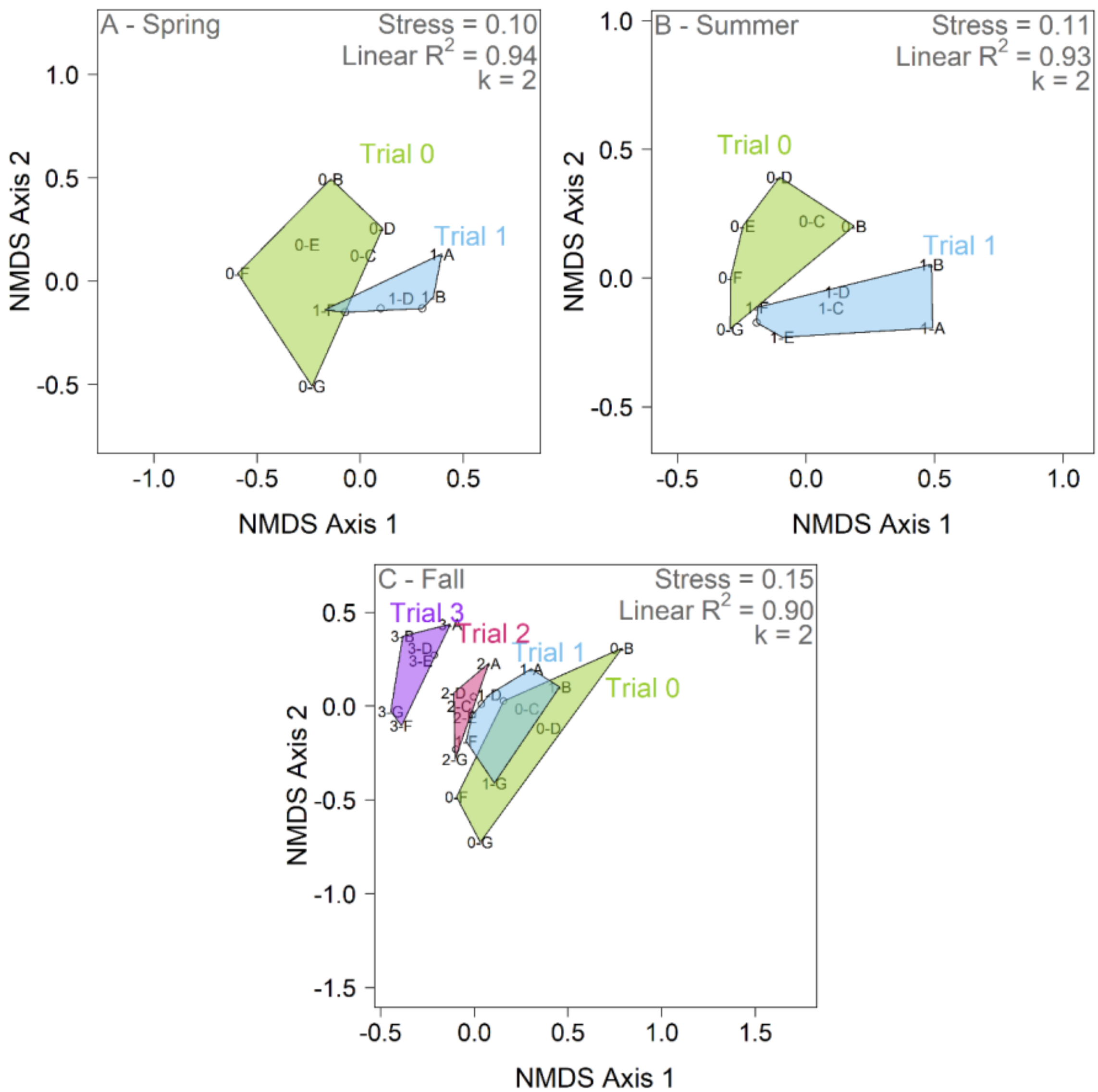


Figure 3.10 Non-metric multidimensional scaling (NMDS) plots contrasting family abundance using Bray-Curtis dissimilarity reduced to two dimensions (k) for (A) Spring, (B) Summer and (C) Fall. Stress is a measure of goodness of fit and the linear R^2 is the squared correlation between fitted and observed ordination values.

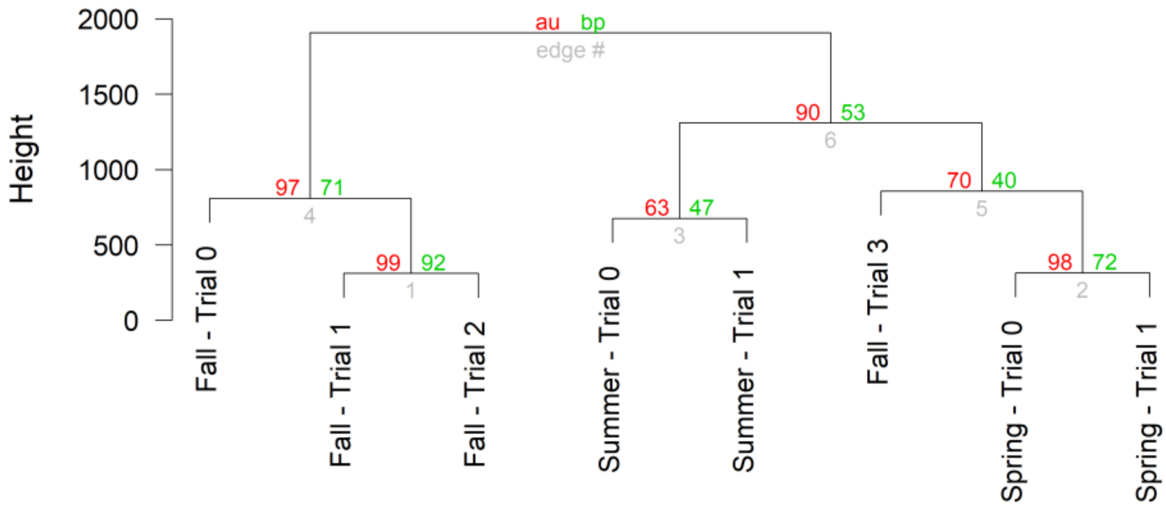


Figure 3.11 Dendrogram of the cluster analysis for invertebrate family abundance showing the differences between seasons and trials. Approximately unbiased (au; red text) p-values and bootstrap probability (bp; green text) are identified for each branch in the dendrogram.

Similarity percentages (SIMPER) revealed seven taxonomic groups accounted for 86% or more of the dissimilarity among the invertebrate assemblages between trials within each season
(

Table 1.2 and Table). Chironomidae was the most abundant family in each Season-Trial combination and accounted for the highest percent contribution to the dissimilarity between trials (28.7% to 36.3%,

Table 1.2 and Table). Simuliidae, another dipteran family, contributed to 24.3% of the dissimilarity between trials in the spring while the remaining five taxa that contributed to 10.2% or less of the dissimilarity included Ephemerillidae, Heptageniidae and Baetidae, (Order Ephemeroptera), Hydropsychidae, (Order Trichoptera) and the assemblage of rarer taxa called “others”. In the summer, Hydropsychidae contributed to nearly as much of the dissimilarity at Chironomidae at 27% of the dissimilarity between trials and the remaining five taxa contributed to 11.4% or less (

Table 1.2). In the fall, Baetidae, Ephemerillidae, Heptageniidae, Simuliidae, Hydropsychidae and "other" invertebrates contributed to approximately 50% of the cumulative dissimilarity between trials (Table). The 64% decline in overall invertebrate density in the fall between Trial 2 and 3 (Figure 3.) was driven by lower abundance of all the key families shown in Table with most important contributors being Chironomidae, Simuliidae, Baetidae, and Heptageniidae.

Table 1.2 Spring and summer SIMPER results for comparisons between invertebrate family-level abundances between trials

Season	Family	Average Abundance per sample ± SD (# of invertebrates/sample)		% Contribution to Dissimilarity Between Trials
		Trial 0	Trial 1	
Spring	Chironomidae	958 ± 1809	902 ± 1444	32.2
	Simuliidae	524 ± 1226	751 ± 1495	24.2
	Ephemerellidae	153 ± 229	266 ± 250	10.2
	Other	94 ± 254	132 ± 110	6.2
	Heptageniidae	88 ± 93	118 ± 106	6.1
	Hydropsychidae	45 ± 59	118 ± 106	5.4
	Baetidae	67 ± 60	169 ± 198	5.2
	Cumulative % Contribution			**89.4**
Summer	Chironomidae	1024 ± 1039	1092 ± 1306	29.2
	Hydropsychidae	335 ± 628	930 ± 1251	27.0
	Baetidae	300 ± 352	330 ± 428	11.4
	Other	74 ± 76	333 ± 909	6.6
	Simuliidae	93 ± 148	142 ± 269	5.0
	Ephemerellidae	74 ± 146	124 ± 108	4.4
	Heptageniidae	110 ± 161	128 ± 158	4.2
	Glossosomatidae	48 ± 91	112 ± 208	2.3
	Cumulative % Contribution			**89.9**

Table 3.3 Fall SIMPER results for comparisons between invertebrate family-level abundances between trials.

Family	Average abundance per sample ± SD (# of invertebrates/sample)				% Contribution to dissimilarity between trials					
	Trial 0	**Trial 1**	**Trail 2**	**Trial 3**	**0 & 1**	**0 & 2**	**0 & 3**	**1 & 2**	**1 & 3**	**2 & 3**
Chironomidae	1982 ± 1494	1494 ± 1750	1380 ± 1177	349 ± 311	36.6	36	30.4	33.5	31.9	32.6
Baetidae	428 ± 671	635 ± 516	435 ± 386	185 ± 161	15.7	11.6	13.6	13.3	17.1	11
Simuliidae	477 ± 964	467 ± 952	360 ± 746	318 ± 529	14.7	12.8	19.1	13.2	15.3	13.3
Heptageniidae	101 ± 102	338 ± 232	406 ± 263	220 ± 156	8.1	10	8.9	8	8.5	10.9
Hydropsychidae	112 ± 154	210 ± 236	191 ± 399	23 ± 24	5.3	5.3	6.4	6.2	5.8	4.6
Ephemerellidae	96 ± 208	192 ± 201	294 ± 229	101 ± 83	4.3	6.9	5.0	6.4	3.6	6.5
Other	96 ± 177	166 ± 326	305 ± 408	0 ± 0	3.4	5.2	2.7	6.4	3.6	6.5
Cumulative % Contribution					**88.3**	**87.8**	**86.1**	**87**	**87.1**	**86.5**

3.1.5. Attributes Driving Benthos Assemblages

Six ecological variables constrained the ordination in the redundancy analysis for each Season-Trial combination. The variables were water temperature (Temp), water depth at the sampler (SDepth), water velocity at the sampler (SVelocity), peak periphyton biomass (PB), pink salmon on/off (Pink) and influence of the Yalakom River (on/off) to the sampling site (Yalakom). In the spring, during Trial 0 and Trial 1 all of the variables combined explained 40% of variance in assemblage pattern in Trial 0 and 42% of variance in assemblage pattern in Trial 1. In the summer, all of the variables explained 48% and 42% of assemblage patterns during Trials 0 and 1, respectively. In the fall, the variables explained 54%, 38% and 67% of variance in assemblage patterns during Trials 0, 1 and 3 respectively. Note that sample depth and sample velocity were not consistently recorded during Trial 2, so these variables were excluded from the analysis of Trial 2. As a result, water temperature, peak periphyton biomass, pink abundance and sample location relative to the Yalakom River explained 44% of the total variation in the invertebrate community patterns.

In the spring, the first redundancy axis (RDA1) in each trial was significant ($p \leq 0.01$) and explained 45.5% and 55.4% of the fitted variation in the family-level invertebrate assemblages in Trials 0 and 1 (Figure). The RDA2 axes during Trial 0 and 1 were also significant ($p<0.05$) and explained 26.5% and 17.8% of the fitted variation, respectively.

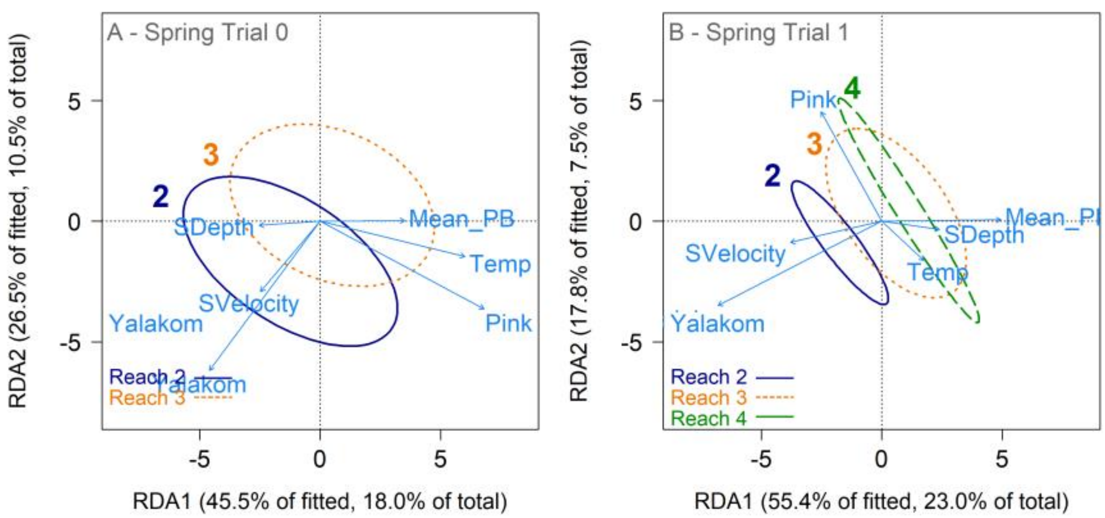


Figure 3.12 Redundancy analysis (RDA) plot of the spring invertebrate family structure as it relates to environmental attributes in each trial. Percent of variation explained by each axis is separated as a percent of explained variation by the constraining ecological variables (fitted) and the total variation in the multivariate regression model (total).

During Trial 0 in the spring, the invertebrate communities separated along a continuum in peak periphyton biomass, sample depth, sample velocity and proximity to the Yalakom River (Figure , Plot A). Communities in Reach 2 (solid blue ellipse) were associated with deeper and faster areas of the river and were strongly influenced by discharge from the Yalakom River while communities sampled in Reach 3 (short-dashed orange ellipse) were associated with greater peak periphyton biomass, higher water temperature, and slower and more shallow areas of the river. Communities in both reaches were almost equally influenced by pink salmon abundance. In Trial 1 (Figure , Plot B), the relationship between the ecological variables and the invertebrates sampled in Reach 2 and 3 did not substantially change from Trial 0 but the spatial gradient along the river continuum became more apparent with the inclusion of samples from Reach 4 (long-dashed green ellipse; Figure , Plot B). Samples collected downstream of the Yalakom River in Reach 2 were more influenced by the Yalakom inflow and were found in faster water with less periphyton biomass than found upstream. Assemblages from Reaches 3 and 4, upstream of the Yalakom River, were more correlated to each other and associated with more periphyton biomass, deeper water and spanned a larger gradient in water temperature. Samples in all three reaches were almost equally influenced by pink salmon abundance (Figure , Plot B).

In the summer, RDA1 was significant during both trials ($p = 0.001$) and explained 38.0% and 63.7% of the fitted variation in the invertebrate assemblages (Figure). RDA2 was also highly significant ($p \leq 0.006$) during both trials and explained 27.1% and 16.9% of the fitted variation in assemblage patterns. As in the spring, the invertebrates sampled during summer spanned a gradient in sample velocity, sample depth, peak periphyton biomass, water temperature and differed in their proximity to the Yalakom River (Figure). The largest influence on invertebrate assemblages during Trial 0 was from the Yalakom River inflow and presence/absence of pink salmon (longest arrows in Figure , Plot A). In Trial 1 the greatest influence was again from the Yalakom inflow, temperature, and hydrologic variables. During Trial 0, invertebrates sampled downstream of the Yalakom River in Reach 2 were not only influenced by the Yalakom River but also by ranges of temperature and velocity while invertebrates sampled upstream of the Yalakom River in Reach 3, were more correlated with slower stream velocities and cooler water temperature (Figure , Plot A). This trend carried into Trial 1, where samples in Reach 2 were still highly correlated with the Yalakom River, shallower water depths with less periphyton biomass. Assemblages in Reach 4 were most correlated with the invertebrate communities sampled in Reach 3, in deeper, cooler water with higher periphyton biomass (Figure , Plot B). Invertebrates from all three reaches were equally influenced by pink salmon and those collected in Reach 4 spanned a wider range in stream velocity than those sampled in Reaches 2 and 3 (Figure , Plot B).

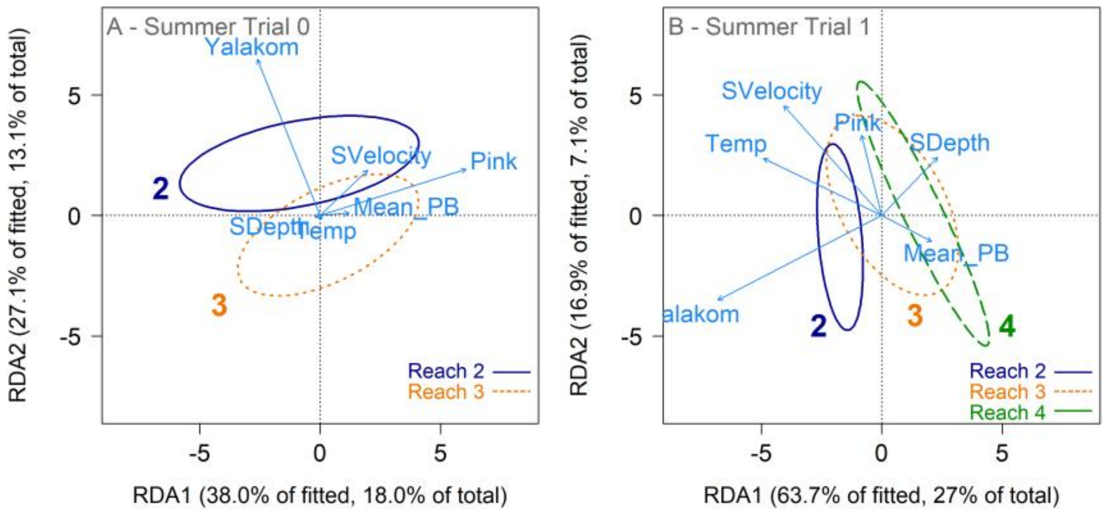


Figure 3.13 Redundancy analysis (RDA) plot of the summer invertebrate family structure as it relates to environmental attributes in each trial. Percent of variation explained by each axis is separated as a percent of explained variation by the constraining ecological variables (fitted) and the total variation in the multivariate regression model (total).

Driver variables in the fall explained more of the variation in the invertebrate assemblages within each trial compared to the other seasons (**Error! Reference source not found.**). The first redundancy axis was significant ($p = 0.001$) in all four trials and explained up to 62.5% of the fitted variation in assemblage patterns. The second redundancy axis in the fall was also highly significant ($p \leq 0.012$) and explained up to 39.3% of the variation.

Distinct environmental drivers showed up in the fall. In each trial, invertebrates in Reach 2 were the most influenced by inflow from the Yalakom River (**Error! Reference source not found.**). During Trials 0 and 1 the community in Reach 2 was associated with a velocity gradient (**Error! Reference source not found.**, Plots A and B). The communities in Reaches 3 and 4 were more correlated with water temperature. The invertebrate communities in Reaches 3 and 4 also spanned a wider gradient in periphyton biomass compared to invertebrates sampled in Reach 2 during Trials 0 and 1. Invertebrates sampled during Trial 0 in Reaches 2 and 3 were almost equally influenced by pink abundance as was the case during Trial 1. The invertebrate community in Reach 4 became less like the community in Reach 3 in the change from Trial 1 to 2. In Trial 3 there was greater separation of the invertebrate assemblage patterns (greater spread of the ellipses) than in the earlier trials. Reach 2 invertebrates were again strongly influenced by the Yalakom River inflow with relatively small influence from the other attributes. In contrast the assemblages in Reach 3 laid on a gradient of water depth and velocity and a strong influence from the presence of pink salmon. The assemblage in Reach 4 was narrowly confined in ordination space, which means abundance and diversity was lower than in the other reaches. This degree of confinement was not observed in the other seasons and trials, indicating a major change in the assemblage in Reach 4 in the fall during Trial 3. The assemblage was strongly related to temperature and periphyton biomass.

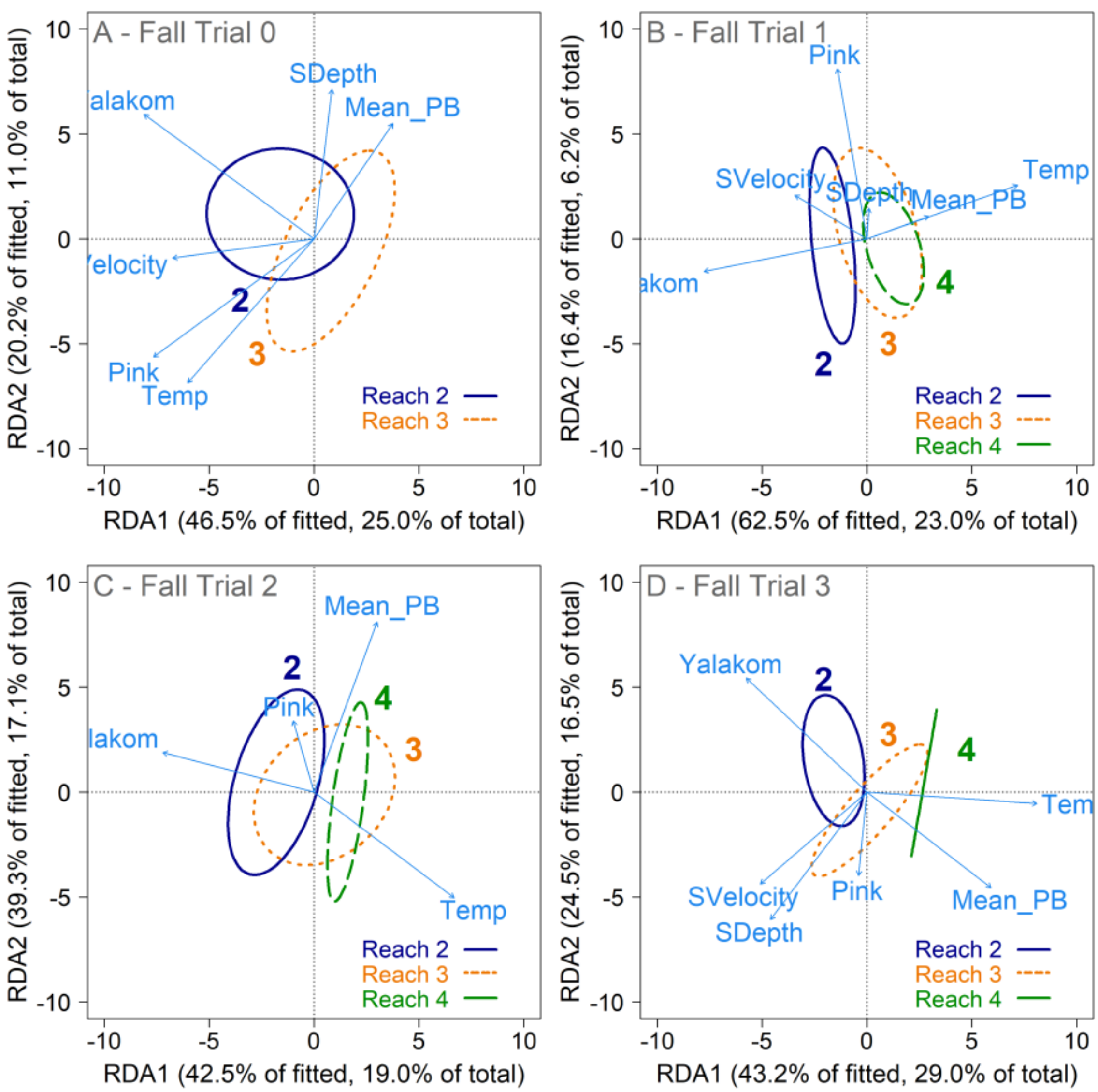


Figure 3.14 Redundancy analysis (RDA) plot of the fall invertebrate family structure as it relates to environmental attributes in each trial. Percent of variation explained by each axis is separated as a percent of explained variation by the constraining ecological variables (fitted) and the total variation in the multivariate regression model (total).

3.1.6. Juvenile Fish Production: Size

Mean weight of Age-0+ mykiss in all reaches was almost always higher during the high flow period (2016-2017) compared to other treatment periods (Figure 3.15). This likely occurred because of reduced density (see Figures 3.16 and 3.17 in Section 3.1.7). 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 3.9, above) combined with the quality rearing conditions in this reach prior to the flow release. Mean size was greater during the high flow period for Age-1 mykiss, however there was considerable overlap in standard deviation error bars. Average weight of Age-0+ coho in Reach 2 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. Density increased across these treatments (see Figure 3.16 in Section 3.1.7), so the potentially larger size under Trial 2 could be due to improved rearing conditions in Reach 2. Average Age-0+ coho weight in Reach 2 was highest during the Trial 3 high flow years, likely due to much lower densities. Patterns in mean weight for Age-0+ coho across flow treatments in reaches 3 and 4 closely matched the patterns seen for Age-0+ mykiss and were likely caused by higher growth in Reach 3 during Trial 0 due to better food availability (benthic invertebrate abundance), and better growth during the high flow period related to lower densities.

In Reach 2, mean weight of Age-0+ chinook was higher under Trial 2 (6 $m^3 \cdot s^{-1}$) and the Trial 3 high flow period 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 growth was slightly higher during the Trial 3 high flow period but again there was considerable overlap in error bars. These changes could be due to reduced density and earlier emergence. Mean growth was higher during the high flow period in Reach 4, again likely due to lower densities.

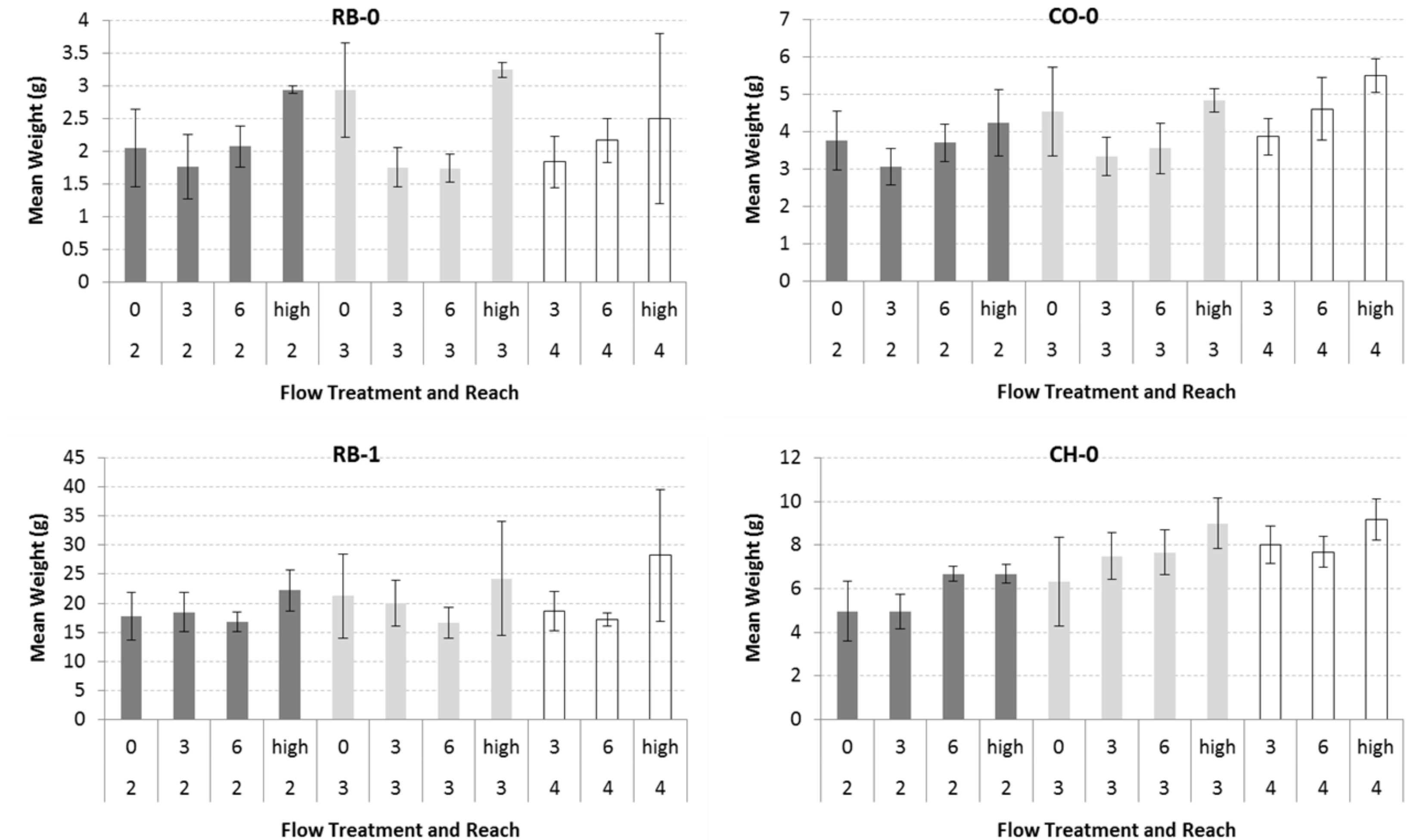


Figure 3.15 Mean juvenile salmonid weight during fall standing stock assessments across flow treatments (0, 3, and 6 $m^3 \cdot s^{-1}$ treatments and the high flow period) and reaches (2, 3, and 4). RB-0, RB-1, CO-0, and CH-0 denote Age-0+ mykiss, Age-1 mykiss, Age-0+ coho, and Age-0+ chinook, respectively. 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).

3.1.7. Juvenile Fish Production: 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 Age-0+ mykiss in reaches 2 and 3 and there was substantial new production in Reach 4 (Table 3.4 and 3.5, Figures 3.16 and 3.17). Age-0+ mykiss abundance increased by an average of 1.8- and 1.9-fold under the Trial 1 and Trial 2 (6 $m^3 \cdot s^{-1}$) treatments compared to Trial 0, respectively. In contrast, Age-0+ mykiss abundance under recent high flows (2016 and 2017) was 0.45-fold of the mean abundance under Trial 0 (i.e., abundance was 55% lower). Relative to Trials 1 and 2, the high flow mykiss abundance was 0.23- and 0.25-fold, or a decline of 77% and 75%, respectively

Table 3.4 Average total abundance (a, '000s) and biomass (b, kg) of juvenile salmonids in the Lower Bridge River across all reaches by flow treatment. 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

Flow	RB-0	RB-1	CO-0	CH-0
0 $m^3 \cdot s^{-1}$	92	36	25	39
3 $m^3 \cdot s^{-1}$	175	36	82	22
6 $m^3 \cdot s^{-1}$	163	34	77	13
High	41	11	9	12

b) Biomass

Flow	RB-0	RB-1	CO-0	CH-0
0 $m^3 \cdot s^{-1}$	249	690	108	228
3 $m^3 \cdot s^{-1}$	305	653	281	134
6 $m^3 \cdot s^{-1}$	311	554	286	92
High	122	315	40	101

Age-1 mykiss abundance increased a small amount in Reach 2 from Trial 0 to Trial 1 while the opposite occurred in Reach 3. Trial 1 produced about 10,000 additional parr in Reach 4. Across reaches there have been negligible changes in mykiss parr abundance across the Trial 0, Trial 1, and Trial 2 treatments. Age-1+ mykiss abundance under high flows was 1/3rd the averages from previous years (i.e., Trials 0, 1 and 2), representing a decrease of approx. 70% relative to each previous trial average. Note that this average high flow abundance is only based on one year (2017) and is within the range of low annual abundance estimates observed during the pre-flow period (Trial 0).

Table 3.5 Relative number of fish produced (by species and age class) under each flow treatment. Each value reflects production by the flow treatment in the column label relative to the flow treatment in the row label (1.0 = equivalent production).

Species-Age Class		Flow Treatment (Mean Annual Release) Trial 1 ($3\ m^3/s$)	Trial 2 ($6\ m^3/s$)	High Flows (>15 m^3/s)
RB Age-0+	Pre-Flow ($0\ m^3/s$)	1.9	1.8	0.4
	Trial 1 ($3\ m^3/s$)		0.9	0.2
	Trial 2 ($6\ m^3/s$)			0.3
RB Age-1	Pre-Flow ($0\ m^3/s$)	1.0	0.9	0.3
	Trial 1 ($3\ m^3/s$)		0.9	0.3
	Trial 2 ($6\ m^3/s$)			0.3
CH Age-0+	Pre-Flow ($0\ m^3/s$)	0.6	0.3	0.3
	Trial 1 ($3\ m^3/s$)		0.6	0.6
	Trial 2 ($6\ m^3/s$)			1.0
CO Age-0+	Pre-Flow ($0\ m^3/s$)	3.3	3.1	0.3
	Trial 1 ($3\ m^3/s$)		0.9	0.1
	Trial 2 ($6\ m^3/s$)			0.1
All Salmonids	Pre-Flow ($0\ m^3/s$)	1.7	1.5	0.4
	Trial 1 ($3\ m^3/s$)		0.9	0.2
	Trial 2 ($6\ m^3/s$)			0.3

Age-0+ coho abundance trends followed those for Age-0+ mykiss 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 3.4a, Figures 3.16 and 3.17). On average, Age-0+ coho abundance increased by 3.3- and 3.1-fold under the Trial 1 and Trial 2 treatments compared to under the Trial 0 pre-flow condition, respectively. Similar to Age-0+ mykiss, Age-0+ coho abundance under recent high flows (2016 and 2017) was only 1/3rd of the abundance under Trial 0.

Age-0+ chinook abundance increased slightly in Reach 2 under the Trial 1 treatment relative to Trial 0, but declined in Reach 3 owing to higher incubation temperatures resulting in premature emergence (Table 3.4a, Figures 3.16 and 3.17). Chinook make little use of Reach 4. As a result of these factors, Age-0+ chinook abundance under the Trial 1 and 2 treatments and high flows (Trial 3) have been 0.6-, 0.3- and 0.3-fold of the abundance under Trial 0, respectively. Unlike the case for Age-0+ mykiss and coho, high flows in 2016 and 2017 have not resulted in a further decline in Age-0+ chinook abundance, perhaps because their abundance is already severely depressed.

Differences in biomass among flow treatments for all species and age classes closely matched those based on abundance because there have been only minor changes in average weights (Table 3.4b, Figures 3.18 and 3.19).

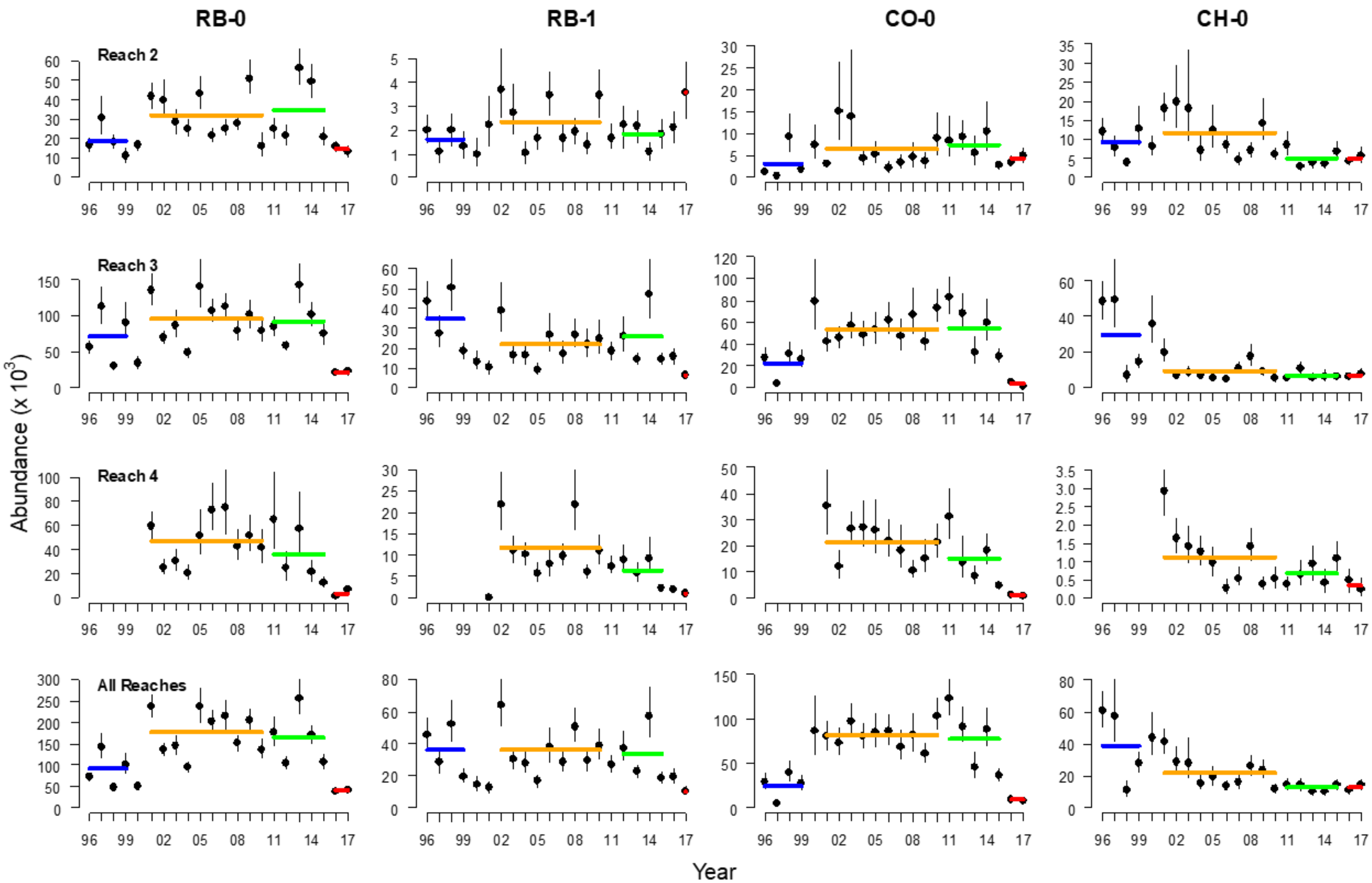


Figure 3.16 Abundance (in thousands) of juvenile salmonids in the lower Bridge River by reach (row) and species-age class (column). Points and vertical lines show mean values and 90% credible intervals from posterior distributions of abundance for each year from the hierarchical Bayesian model, respectively. 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 mkiss, age-1+ mykiss, age-0 coho, and age-0 chinook, respectively.

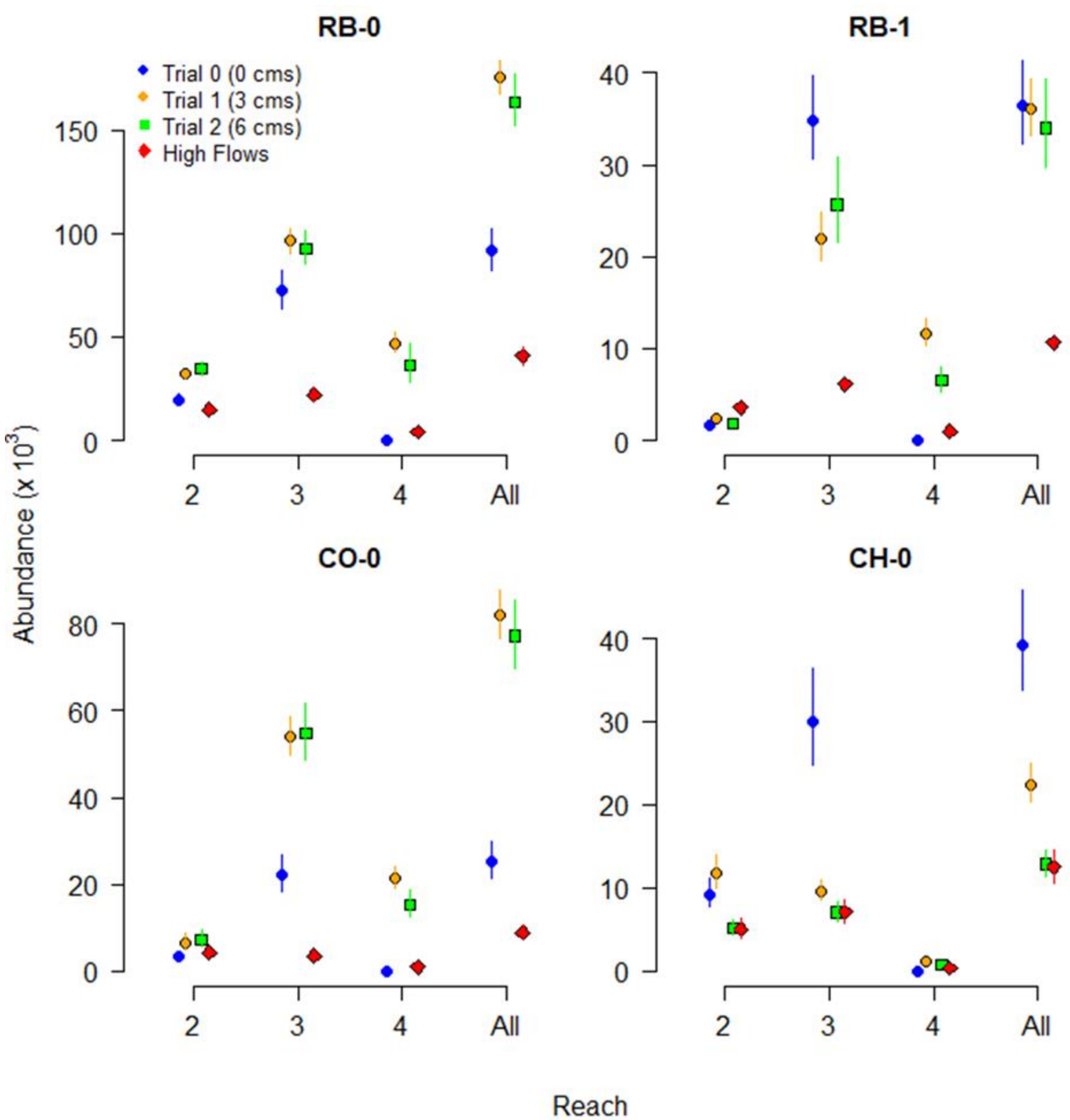


Figure 3.17 Average (points) and 90% credible intervals (vertical lines) of juvenile salmonid abundance by reach for each flow treatment. RB-0, RB-1, CO-0, and CH-0 denote Age-0+ mykiss, Age-1 mykiss, Age-0+ coho, and Age-0+ chinook, respectively.

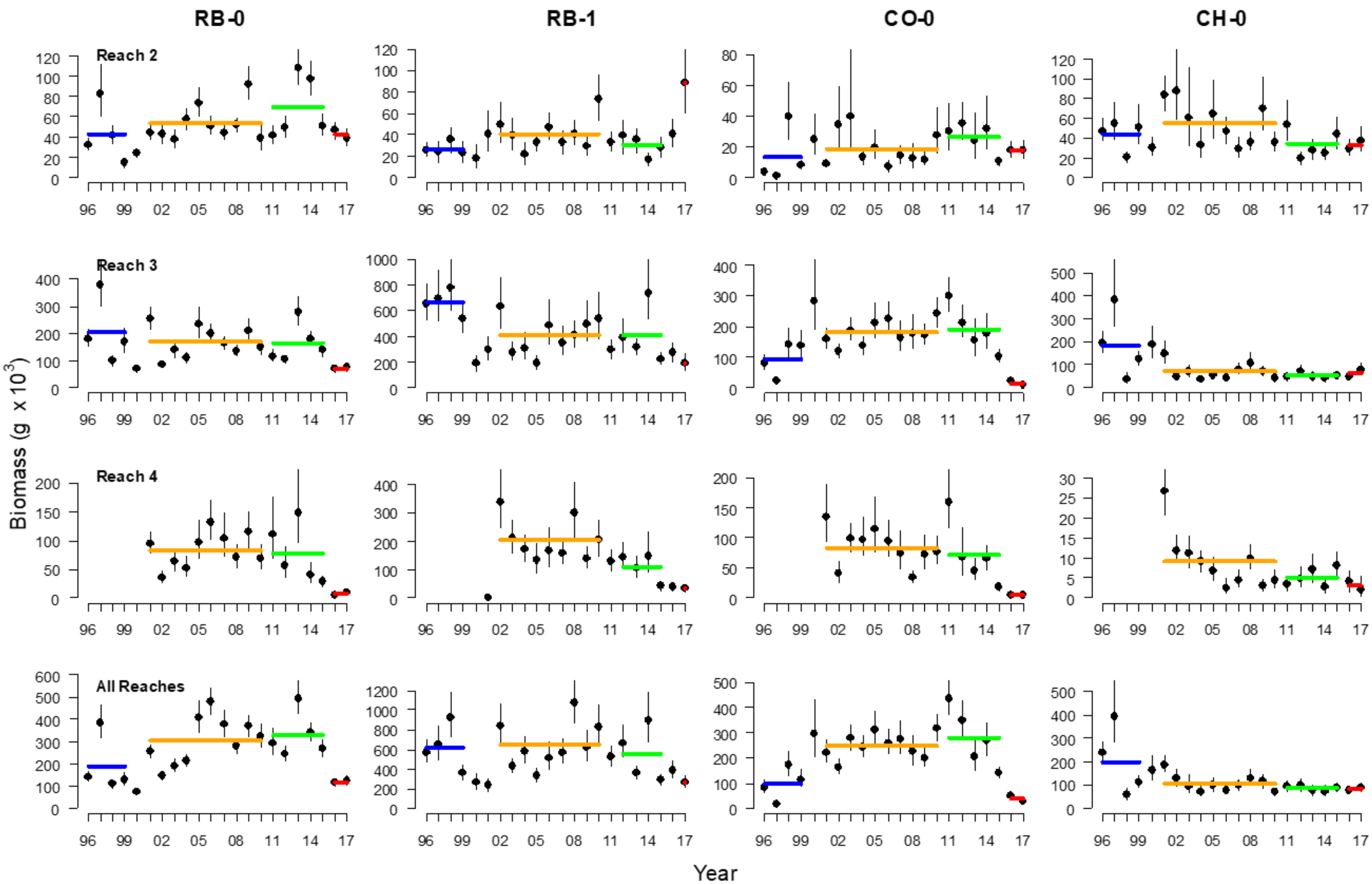


Figure 3.18 Biomass (in thousands of grams or kilograms) of juvenile salmonids in the Lower Bridge River by reach (row) and species-age class (column). See caption for Figure 3.16 for details.

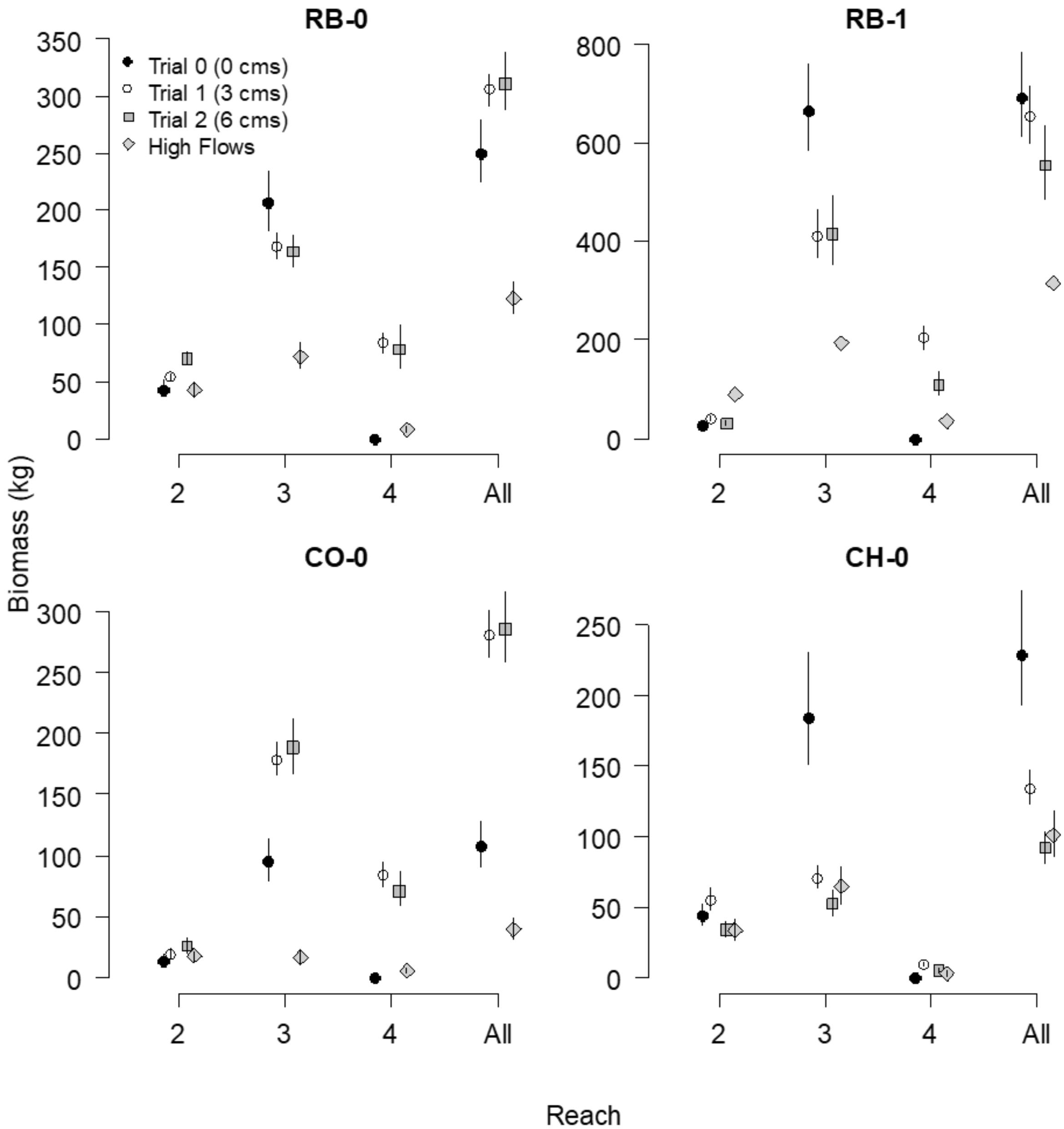


Figure 3.19 Average (points) and 90% credible intervals (vertical lines) of juvenile salmonid biomass by reach for each flow treatment.

3.1.8. Juvenile Fish Production: Stock-Recruitment

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. Age-0+ coho abundance increased under the Trial 1 and Trial 2 treatments relative to the Trial 0 pre-flow period (Figure 3.20). The magnitude in the shift in the stock-recruitment curve for Age-0+ coho (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 a 2- to 3-fold increase under trials 1 and 2 relative to pre-flow conditions, respectively, and a reduction by about 1/3rd under the recent high flows.

There is no indication in the shape of the stock-recruitment curve that coho fry production is limited by escapement (i.e. under-seeded) as almost all data points are near or on the asymptote of the stock-recruitment curve. Escapements in 2005 and 2009 were low and close to the origin but fry production in the following years 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 curve to a much greater extent (Figure 3.21). In this case there are many data points that have escapements less than required to maximize fry production. This more constrained curve provides a near equivalent fit to the data. The difference in log-likelihood measuring the fit of the curves in Figures 3.20 and 3.21 is less than 2 units and therefore these curves are not significantly different. The stock-recruitment curve in Figure 3.21 implies that the population has been under-seeded. In this case poor fry production under recent high flow years can be partially attributed to low escapement. More data are required to better define the initial slope of the stock-recruitment relationship to strengthen inferences about spawning stock limitation 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 3.22). 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 0.7-fold, 0.46-fold, and 0.44-fold lower than under the pre-flow conditions. Owing to the steep initial slope there is no indication that escapement has been limiting fry abundance. 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 Trial 1 flow treatment relative to the pre-flow conditions, and no effects of the Trial 2 flow treatment and higher flows in 2016 and 2017 (Figure 3.23). Again, this more constrained curve provides a near equivalent fit to the data (the likelihood difference between fits is less than 2 units). Thus, the data are not sufficient to allow us to separate flow effects from stock size effects (escapement).

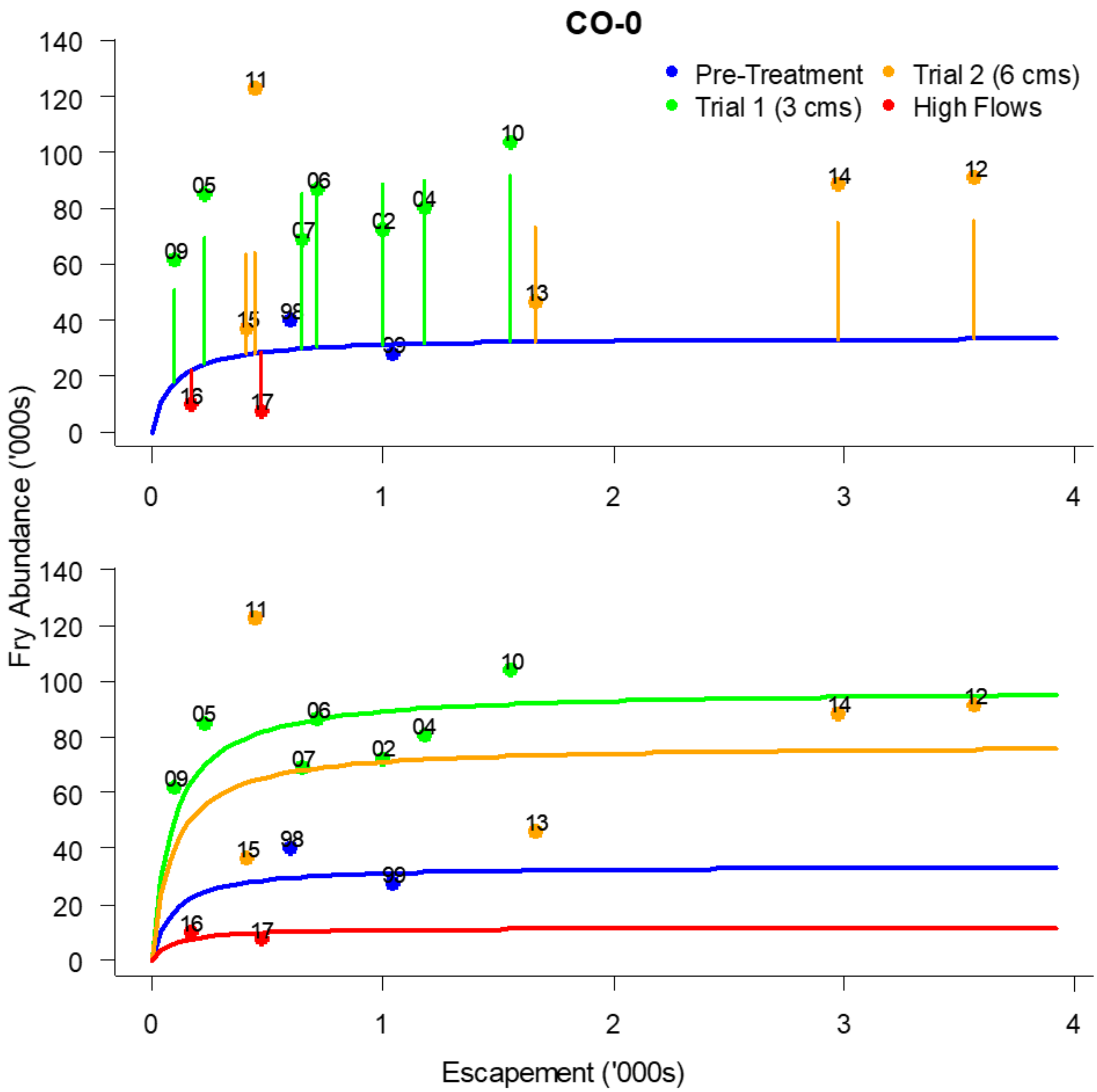


Figure 3.20 Spawner-fry coho Beverton-Holt stock-recruitment curves fit assuming a maximum initial slope of 375 fry/spawner (50% egg-fry survival rate). Points show annual estimates of escapement and Age-0+ abundance with the label beside each point showing the brood year (year of escapement). The blue 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).

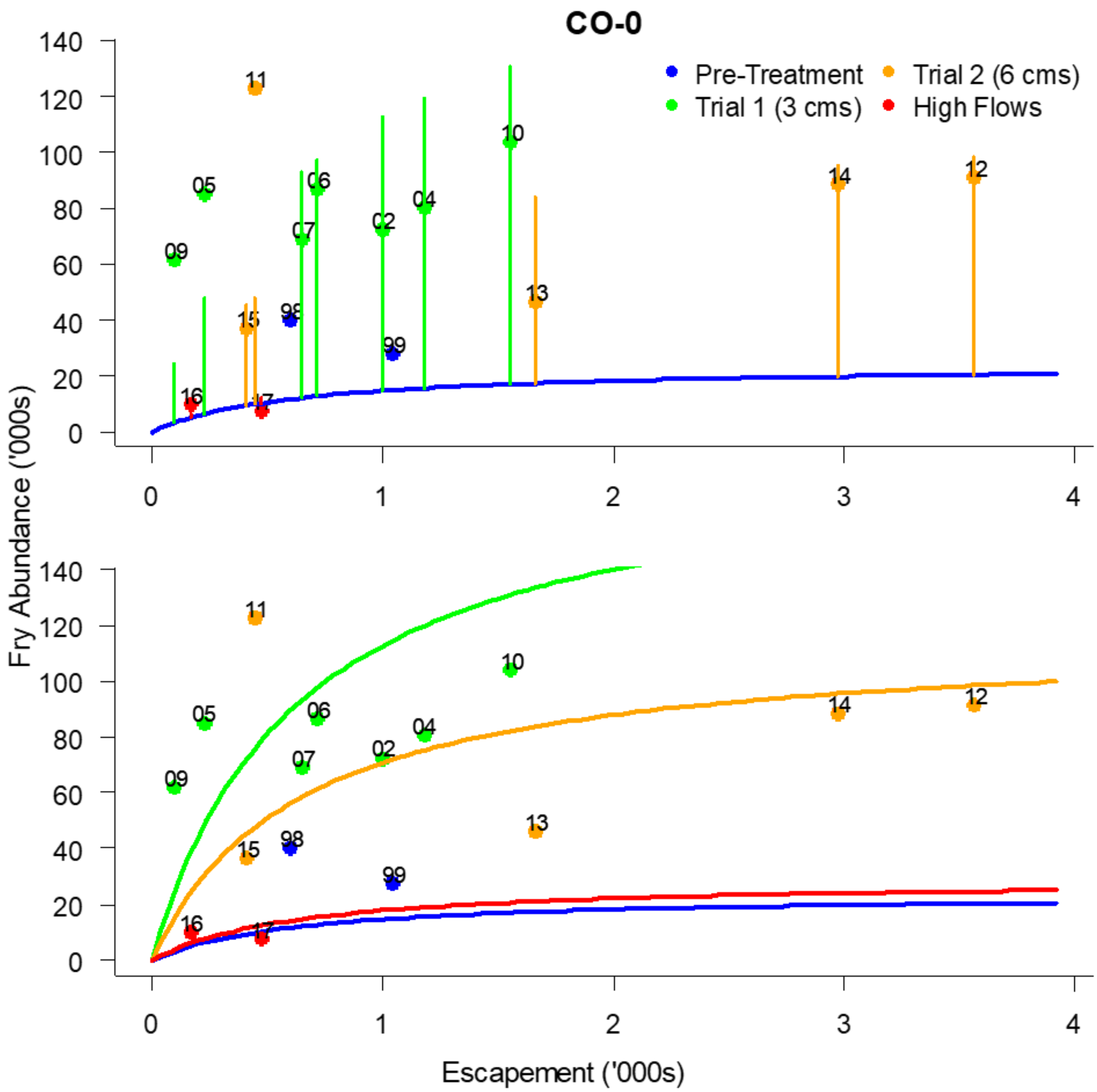


Figure 3.21 Spawner-fry coho Beverton-Holt stock-recruitment curves fit assuming a maximum initial slope of 37.5 fry/spawner (5% egg-fry survival rate). See caption for Figure 3.20 for additional details.

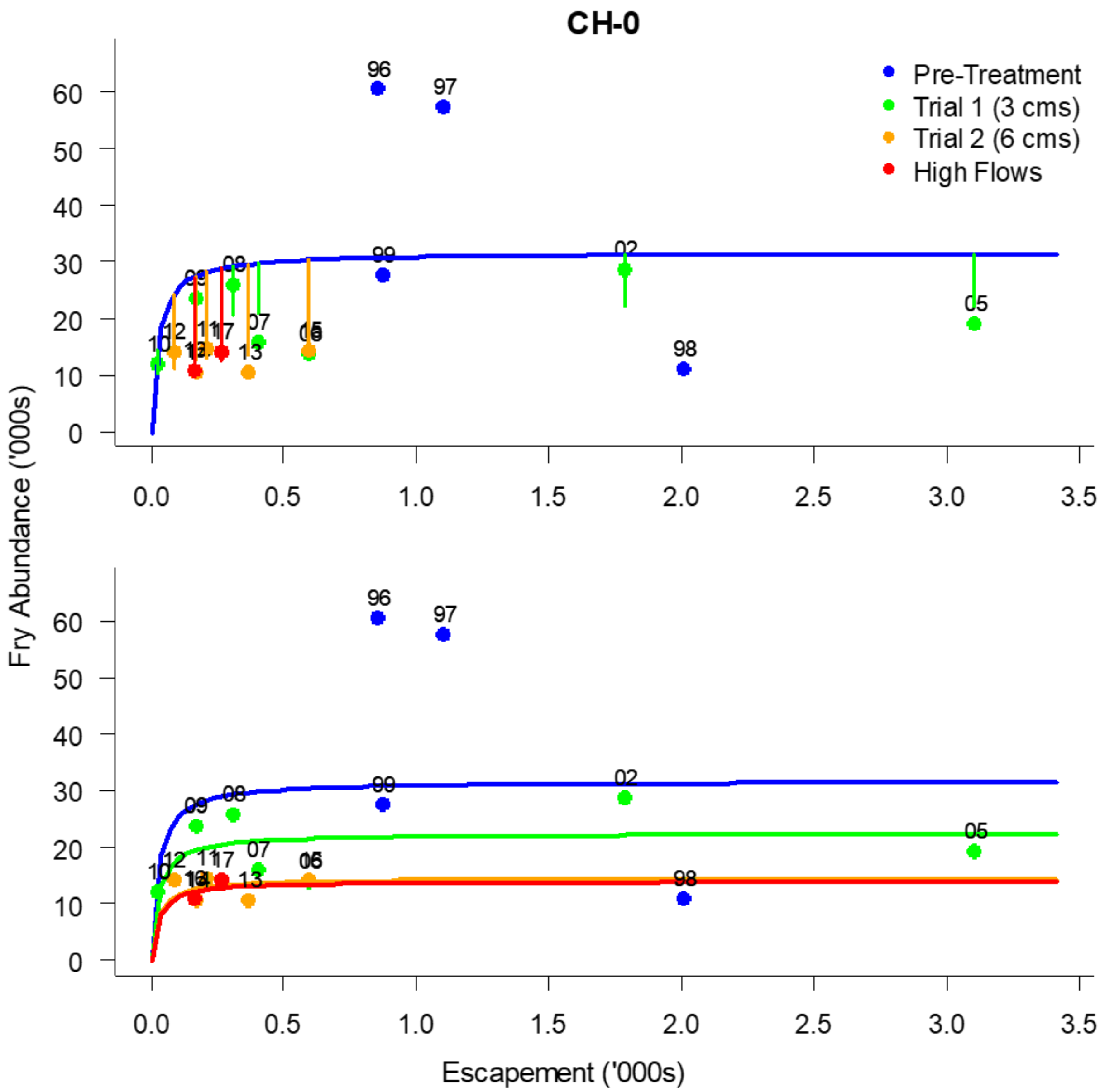


Figure 3.22 Spawner-fry chinook Beverton-Holt stock-recruitment curves fit with a constraint that assumes a maximum egg-fry survival rate of 50% (maximum initial slope of 1250 fry/spawner). See caption for Figure 3.20 for details.

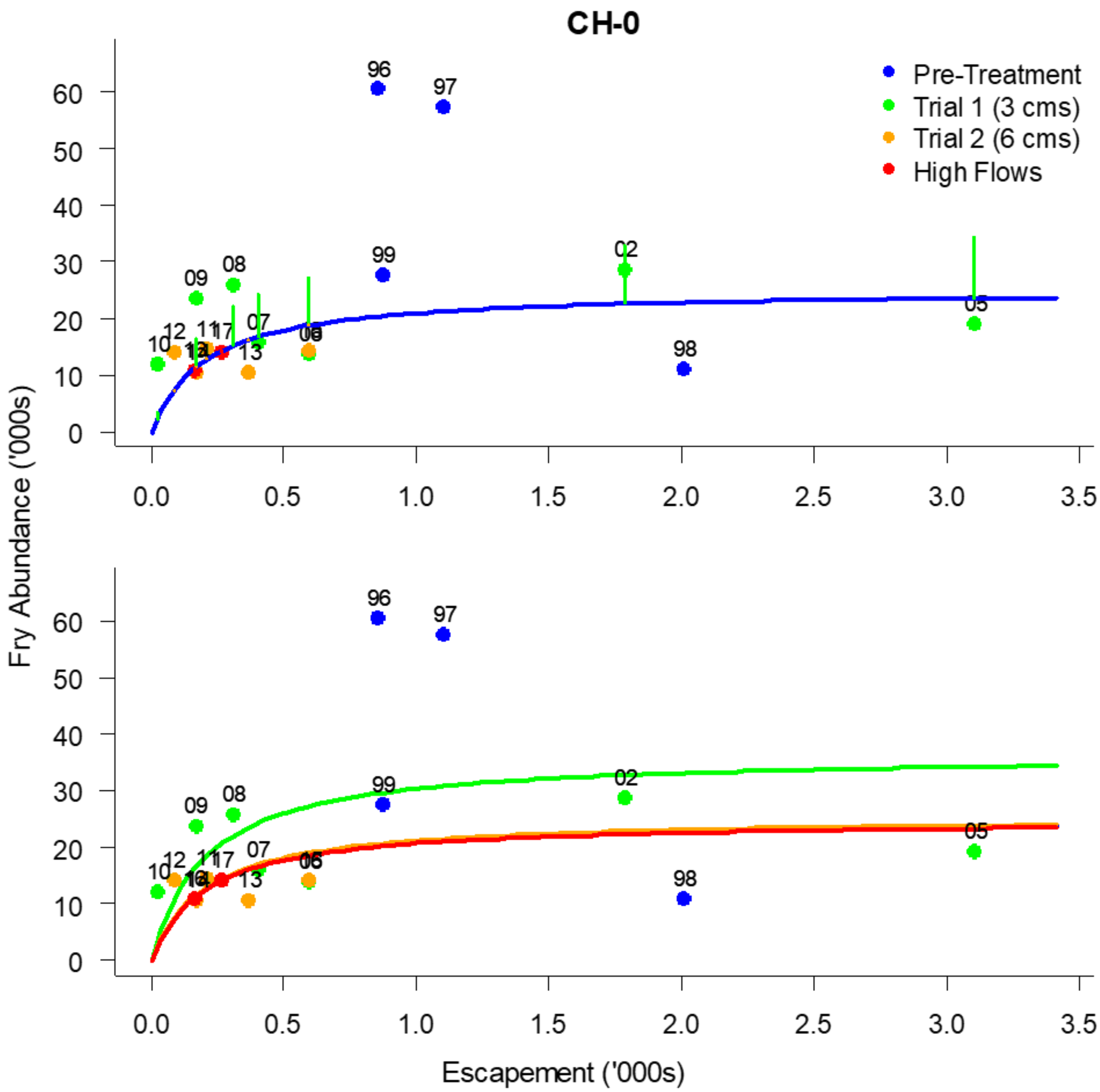


Figure 3.23 Spawner-fry chinook Beverton-Holt stock-recruitment curves fit with a constraint that assumes a maximum egg-fry survival rate of 5% (maximum initial slope of 125 fry/spawner). See caption for Figure 3.20 for details.

3.2. High Flow Monitoring

3.2.1. Kokanee Entrainment Surveys and Spot Water Quality Monitoring

As in 2016, kokanee entrainment monitoring confirmed that entrainment of kokanee from Carpenter Reservoir into the Lower Bridge River occurred under high flow releases from Terzaghi Dam in 2017 (O'Farrell and McHugh 2017). Surveys were conducted between 11 June and 25 July 2017, which bracketed the peak flow release period (see Figure 1.3 in Section 1.3). Flows were 122 $m^3 \cdot s^{-1}$ on 11 June, increased incrementally to the peak of 127 $m^3 \cdot s^{-1}$ on 27 June, and then were ramped down to 109 $m^3 \cdot s^{-1}$ by the last survey on 29 June. Carpenter Reservoir was filling across this period from 631.78 m to 634.65 m, an increase of 2.87 m in surface elevation across the month of June 2017. According to the terms described in the WUP, the licenced minimum and maximum levels for Carpenter Reservoir are 606.55 m and 651.08 m under normal operations, respectively; however, the reservoir is operated to a target maximum of 648.00 m for the purpose of minimizing spills at Terzaghi Dam (BC Hydro 2011).

In total, 48 kokanee were enumerated by observers on river left within the 1.5 km survey area below the dam, and 98% of the observed fish were mortalities. As noted in the 2016 report, effective enumeration of live kokanee was not feasible due to the high flows and high turbidity, which made for poor visibility conditions into the water (O'Farrell and McHugh 2017). A total of 83 kokanee were observed across 13 survey dates in 2016. As in 2017, the majority (~98%) were observed at the highest discharges that year (i.e., 81 to 96 $m^3 \cdot s^{-1}$), and at Carpenter Reservoir elevations between 634.42 m to 635.65 m. Refer to the 2016 kokanee entrainment monitoring report for more detailed results from that year (McHugh et al. 2017). More years of monitoring data would be required to sort out the potential interaction between Carpenter Reservoir elevations and Terzaghi Dam discharges on the incidence of kokanee entrainment observed in the LBR.

Mean fork length of the entrained kokanee in 2017 was 235 mm (range = 172 to 311 mm; n= 48). Observers reported that all of the fish were considered mature (based on size) and 81% were in good physical condition (i.e., cause of mortality was not obvious). Wounds such as abrasions, punctures, tissue damage, and damaged or missing fins were noted for 8 fish (17% of the total). The one live individual was noted to have lost swimming equilibrium when observed. Following a peak number of kokanee observed on 11 June, the numbers declined on each subsequent survey date (Figure 3.24). Entrained kokanee were not observed after the 22 June survey.

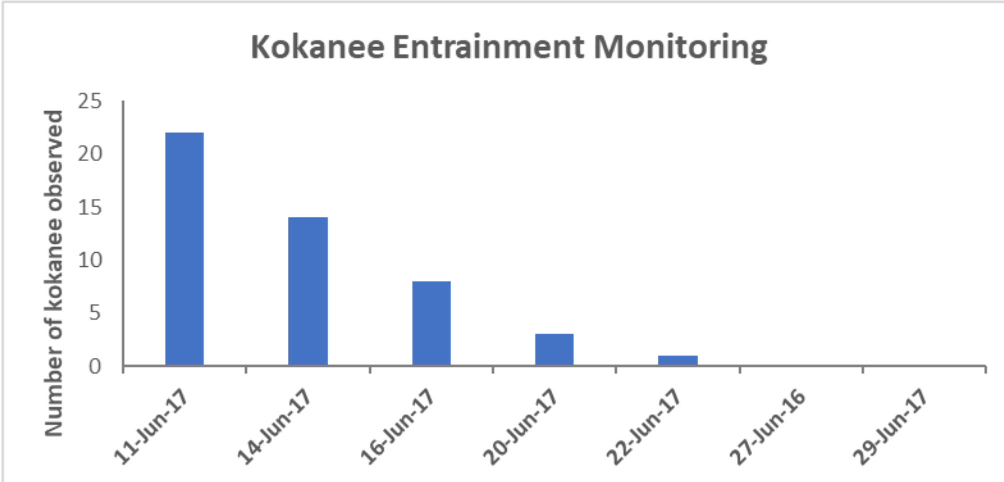


Figure 3.24 Numbers of kokanee observed below the Terzaghi Dam plunge pool on each survey date in 2017 (taken from Coldstream Ecology, Ltd.–Bridge River Band Partnership Weekly Monitoring Report) (McHugh et al. 2017).

Due to the non-quantitative nature of the sampling method, it is not possible to know what proportion of total entrained fish the observed number represents. It is also not possible to determine the specific causes of mortality, or whether the survey dates bracketed the entire entrainment period.

Air and water temperatures generally increased across the monitoring dates (based on the spot measurements), turbidity levels generally declined (i.e., highest levels were on the first survey date), and total dissolved gas (TDG) levels generally followed the trajectory of the flow release at each monitored location (Table 3.6). Water temperatures were within the range 10.3 to 15.3°C and were relatively consistent among locations. Highest turbidity values were between 47.9 and 57.9 NTU on the first survey date (9 June 2017), and diminished across the survey dates to the lowest values between 13.9 and 15.7 NTU on the last survey date (29 June 2017), according to location. The lowest TDG level was 96% saturation at the Russell Springs site (km 30.4) on the first survey date, but otherwise the levels were between 100% and 107% saturation at each site on every survey date. The cause of the single lower value at the Russell Springs site is unknown, but is not likely related to an effect of the high flows. Potential effects of the highest TDG levels (i.e., 107%) on fish, in terms of incidence of gas bubble trauma for instance, were not assessed.

The effects of elevated TDG on fish result from an interaction of several physical factors (e.g., maximum and sustained versus background TDG saturation levels; depth and distance of effect downstream of the source; duration; temperature, accessible depths available to fish in the river, etc.) coupled with potential compensatory mechanisms available to fish (e.g., physiological, behavioural, species- and life history-specific characteristics, etc.). Assessment of the possible interaction of each of these factors for determining potential detrimental effects on fish were not feasible with the scope of information available from the 2016 and 2017 high flow monitoring. It is also unknown how these TDG values compare to levels at the same time of year under the previous flow trials, for context. In future, it is recommended that monitoring follow the BC Hydro Total Dissolved Gas Management Strategy: Implementation Plan when the measured TDG values reach or exceed the 110% to 115% range (BC Hydro 2014).

The bank erosion and deposition surveys resulted in the identification of 11 sites: 3 were observed at 126.6 $m^3 \cdot s^{-1}$ release from the dam, and 1 was observed at 109.5 $m^3 \cdot s^{-1}$ release (Table 3.7). Photos of these locations are not included in this report, but are available upon request.

Table 3.6 Summary of water quality measurements taken at 3 monitoring locations in the Lower Bridge River during the peak flow release period in 2017 (O'Farrell and McHugh 2017).

Site	Survey Date	Flow Release ($m^3 \cdot s^{-1}$)	Water Temperature (°C)	Air Temperature (°C)	Total Dissolved Gas (TGP%)	Mean Turbidity (NTU)
Plunge Pool (km 40.9)	9 June	116.7	10.3	-	-	47.9
	11 June	122.2	11.0	-	-	36.8
	14 June	123.3	10.9	20.5	101	27.5
	16 June	123.9	12.1	16.8	105	19.9
	20 June	124.3	13.1	21.2	105	15.4
	22 June	125.0	12.0	17.6	107	18.3
	27 June	126.6	11.7	18.9	106	15.4
	29 June	109.5	12.2	22.5	104	14.1
Russell Springs (km 30.4)	9 June	116.7	10.8	-[a]	-	49.7
	11 June	122.2	12.5	-	-	29.4
	14 June	123.3	11.3	16.4	96	27.9
	16 June	123.9	12.4	18.6	101	21.6
	20 June	124.3	15.3	23.3	100	15.3
	22 June	125.0	12.5	20.1	102	18.0
	27 June	126.6	12.2	19.6	101	17.2
	29 June	109.5	12.8	24.2	101	13.9
Yalakom R. Confluence (km 25.5)	9 June	116.7	10.9	-	-	57.9
	11 June	122.2	-	-	-	-
	14 June	123.3	11.7	11.7	101	28.4
	16 June	123.9	12.9	21.7	101	23.0
	20 June	124.3	13.9	27.8	101	18.1
	22 June	125.0	12.9	24.9	102	19.3
	27 June	126.6	12.8	27.8	102	18.7
	29 June	109.5	13.4	32.1	101	15.7

[a] McHugh et al. 2017 note that "-" indicates measurement not collected due to changes in scope: on June 11 data were measured at Fraser Lake (km 33.3) instead of Russell Springs (km 30.4) and on June 9 data were measured at km 26.4 instead of the Yalakom confluence.

Table 3.7 Summary of erosion and deposition sites observed during surveys at high flows in 2017 (O'Farrell and McHugh 2017).

Location Names	Lat-Long Coordinates	Rkm	River Bank	Reach	Approx. Length (m)	Approx. Width (m)	Approx. Area (m^2)	Initial Observed Discharge (m^3/s)	Sediment Composition	Picture #
Plunge Pool	50.78799, -122.21834	40.5	RR	4	40	10	400	126.6	Cobble 45%, gravel 50%, boulder 5%	IMG_3205
35.8	50.79627, -122.17605	35.8	RR	3	15	5	75	126.6	60% Fine, 15% Boulders, 12.5%m Cobbble, 12.5% gravel	IMG_3206
Below Fraser Lake	50.80041, -122.17984	33.3	RR	3	70	10	700	126.6	Boulders 5%, cobble 20%, gravel 5% and fines 70%	IMG_3207
Top of new riprap at Fraser Lake	50.81660, -122.17490	33.3	RL	3	4	2	8	126.6	gravel 50% , fines 50% plus rip rap that was added last in 2016	IMG_3208
32.7	50.81924, -122.17475	32.7	RR	3	30	3	90	109	30% fine, 40 % cobble, 40% boulder, 10% gravel	IMG_3209
Between Russel and Fraser	50.81658, -122.17483	32.4	RR	3	25	3	75	126.6	Fines 40%, Cobble 30%, Boulder 15%, Gravel 15%	IMG_3210
Between Fraser/Russel (2)	50.82086, -122.17648	32.2	RR	3	25	2	50	126.6	60% Fine, 25% gravel, 15% boulder	IMG_3211
Below Russel	50.83334, -122.19795	29.9	RR	3	25	3	75	126.6	Fines 70%, Cobble 15%, Gravel 15%	IMG_3212
Below Russel (2)	50.83377, -122.19858	29.8	RR	3	25	5	125	126.6	25% gravel, 25% fines, 25% boulder, 25% cobble	IMG_3213
Below muddy hell	50.84494, -122.20638	28.2	RR	3	100	5	500	126.6	Fines 70%, Gravel 5%, Cobble 5%, Boulder 20%	IMG_3214
Below Horseshoe	50.85574, -122.14984	22.5	RR	2	90	10	900	126.6	Clay 90%, Fines 4%, gravel 3%, cobble 3%	IMG_3215

3.2.2. High Flow Ramp Down Monitoring and Stranding Risk Assessment

A summary of the high flow fish stranding site reconnaissance survey results is provided in Appendix C. A total of 26 potential stranding locations were characterized between the Terzaghi Dam plunge pool and the Applesprings off-channel habitat in Reach 1. Ten sites were in Reach 4, ten sites were in Reach 3, two sites were in Reach 2, and four sites were in Reach 1. Eleven of the sites were on river left side, and the other fifteen were on river right. Potential stranding risk was qualitatively rated as Moderate to High for 25 of the 26 locations, and Low for 1 location. Potential total stranding area as assessed for these locations was 64,045 m^2 (25,550 m^2 (40%) in Reach 4; 17,745 m^2 (28%) in Reach 3; 3,550 m^2 (6%) in Reach 2; and 17,200 m^2 (27%) in Reach 1).

In the tables and figures throughout this section, comparable ramping information from the 2016 high flows as well as ramping results within the "normal" Trial 2 range (15 to 1.5 $m^3 \cdot s^{-1}$) have been included along with the 2017 results, for reference.

Ramp downs from high flows (i.e., between 127 and 15 $m^3 \cdot s^{-1}$) occurred across 9 dates between 28 June and 21 July 2017, representing a total flow reduction of 96.5 $m^3 \cdot s^{-1}$ across that period (Figure 3.25 and Table 3.8). For more detailed information on flow and stage changes for each rampdown event, refer to Tables D1 and D2 in Appendix D. Total stage change at the 36.8 km compliance location was 143 cm, and maximum daily stage change rate implemented was 4.1 cm/hr. The implementation of some higher ramp rates in 2017 (compared to past years) meant that the reduction of flows from a higher magnitude could be completed over a shorter timeframe. Flow ramping within the Trial 2 flow range (≤15 $m^3 \cdot s^{-1}$) was conducted over an additional 9 dates in August, which was comparable to the usual timing from the previous Trial 2 years (2011 to 2015). Flow change, stage change, and ramp rates below 15 $m^3 \cdot s^{-1}$ were also the same as previously reported (Sneep 2016).

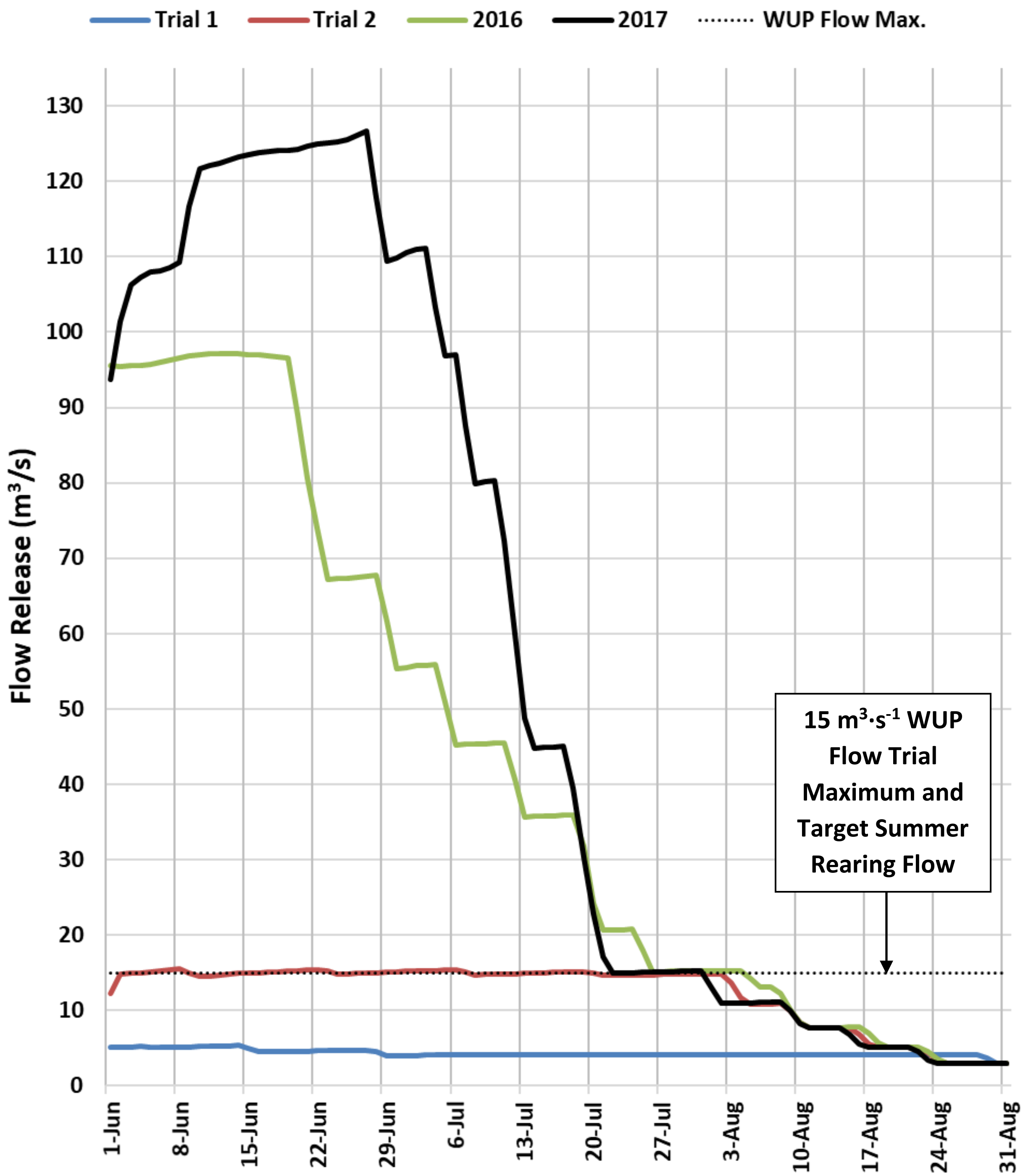


Figure 3.25 **Schedule of flow releases and ramp downs from the peak period to the start of the fall low flow period in 2016 and 2017. For reference, Trial 1 and 2 flows are shown for the same period.**

Table 3.8 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 2016 and 2017. 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 $m^3 \cdot s^{-1}$)	2016	Jun – Jul	8	-81.4	-108	-2.3
	2017	Jun – Jul	9	-96.5	-143	-4.1
"Normal" Ramp Events (≤15 $m^3 \cdot s^{-1}$)	2016	Aug – Sep	10	-13.8	-67	-3.0
	2017	Aug – Sep	9	-13.7	-67	-2.6

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 3.26). The curve drawn through the points has a good fit ($R^2 = 0.9967$), such that the associated equation ($y = 0.6903x^{0.2868}$) may be useful for predicting stage changes for particular flow changes within this range. If higher flows are observed in future years, observed stage and discharge values can be included in this plot to update the relationship. 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, such that the discharge-stage relationship becomes close to linear across the higher flows.

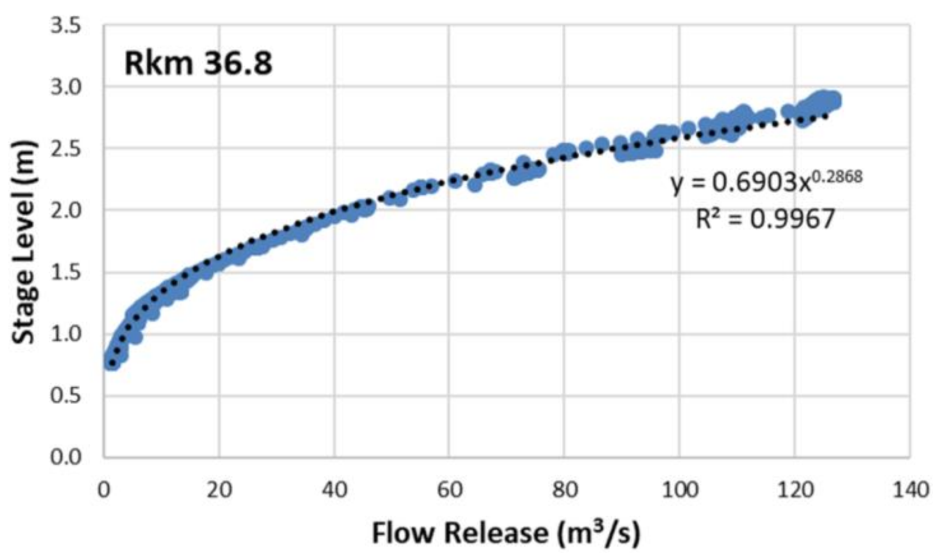
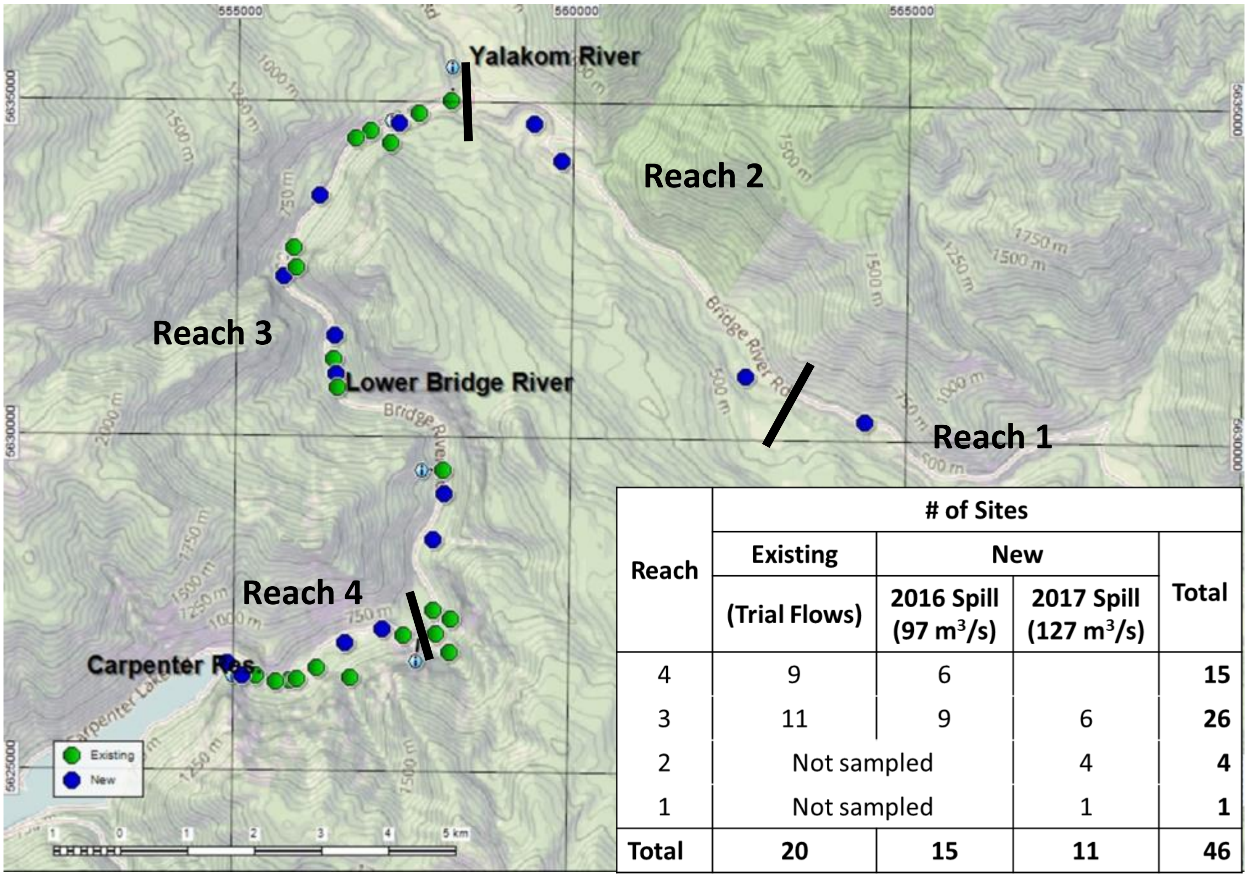


Figure 3.26 Discharge-stage relationship at 36.8 km (the compliance location) across the range of flows observed across all flow treatments.

As a result of the fish stranding site reconnaissance and flow ramp down surveys conducted during the past two high flow years, the incidence of fish stranding was documented at 26 new locations (*n*= 15 in 2016; *n*= 11 in 2017) for flows >15 $m^3 \cdot s^{-1}$ across all four reaches of the Lower Bridge River (Figure 3.27). These were in addition to the 20 sites that had been previously identified for ramp downs below 15 $m^3 \cdot s^{-1}$ during the Trial 1 and 2 years (in reaches 3 and 4 only). The majority (81%) of the new sites at flows >15 $m^3 \cdot s^{-1}$ was in reaches 3 and 4. There were 4 new sites in Reach 2 and 1 new site in Reach 1.



Reach	# of Sites			
	Existing	New		Total
	(Trial Flows)	2016 Spill (97 m^3/s)	2017 Spill (127 m^3/s)	
4	9	6		**15**
3	11	9	6	**26**
2	Not sampled		4	**4**
1	Not sampled		1	**1**
Total	**20**	**15**	**11**	**46**

Figure 3.27 Survey area map for ramp monitoring and fish salvage on the Lower Bridge River showing existing fish salvage locations (green dots) from Trial 2 flows, and newly identified locations (blue dots) under high flow conditions in 2016 and 2017. Stage monitoring 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 high flow range were consistently low relative to the results for the Trial 2 range (≤15 $m^3 \cdot s^{-1}$; Figure 3.28). In previous years, 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 2017 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 (≤99 fish per 1 $m^3 \cdot s^{-1}$ flow change), as defined in the Fish Stranding Protocol for the Lower Bridge River (Sneep 2016). This threshold flow appears to be at ~13 $m^3 \cdot s^{-1}$. 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 both high flow years to-date (see Section 3.1.8). Relative to the Trial 2 averages, abundance of coho and steelhead fry was down by 70% and 90%, 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.

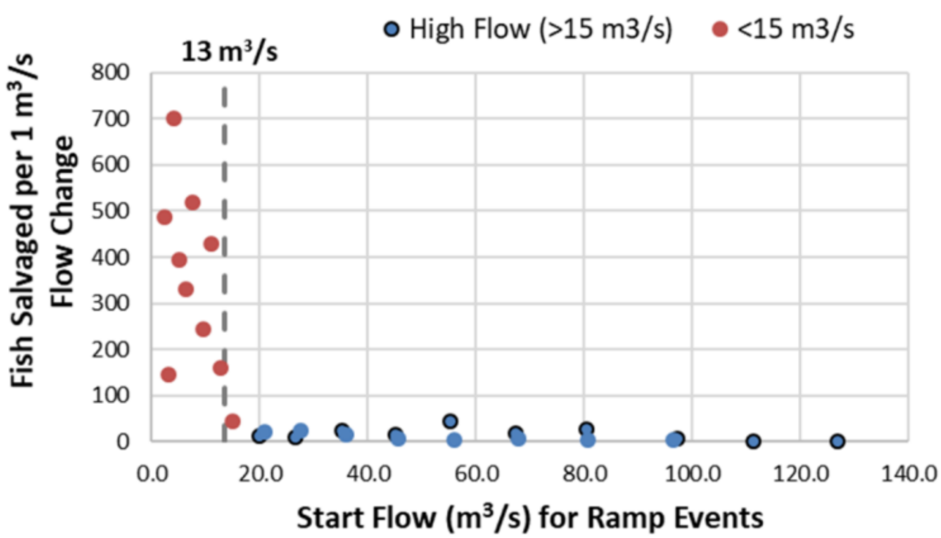


Figure 3.28 Relative differences in number of fish salvaged per increment of flow change for ramp downs from high flows (>15 $m^3 \cdot s^{-1}$) versus Trial 1 and 2 flows (≤15 $m^3 \cdot s^{-1}$). The vertical dashed line represents the approximate flow threshold (~13 $m^3 \cdot s^{-1}$) where the apparent break between high stranding risk and moderate or low stranding risk occurs. Note: values do not include incidental catches. Plain blue circles represent 2016 high flow data, and blue circles with black border are 2017 high flow data.

Compared to survey results from the previous flow trials, relatively large areas of fish stranding habitat were documented in 2016 and 2017 (41,290 m^2), as the high flows expanded the wetted area of the river and flooded additional side channels and benches throughout the study area (Table 3.9). The proportions of stranding area by reach were 12%, 20%, 26%, and 17% for reaches 4, 3, 2, and 1, respectively. Under the trial flows (≤15 $m^3 \cdot s^{-1}$), the total stranding area was 9,405 m^2, which was fairly equally distributed between reaches 4 and 3 (52% and 48%, respectively).

Across the high flow range (>15 $m^3 \cdot s^{-1}$), the highest proportions of salvaged fish per stranding habitat area were in reaches 2 and 3, although the differences among reaches were relatively small (<10 fish per 100 m^2; Figure 3.29). Within the Trial 2 flow range (≤15 $m^3 \cdot s^{-1}$), fish stranding densities were much greater and the highest proportion was in Reach 4 followed by Reach 3 (63 and 25 fish per 100 m^2, respectively). Reaches 1 and 2 were not surveyed before 2017, so trial flow results were not available for those reaches.

Table 3.9 Summary of fish stranding area and numbers of fish salvaged by reach for 2016 and 2017 high flow (>15 $m^3 \cdot s^{-1}$) and 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
High Flows (>15 $m^3 \cdot s^{-1}$)	4	6	5,082 (12%)	118	2
	3	15	8,090 (20%)	668	8
	2	4	10,700 (26%)	745	7
	1	1	7,000 (17%)	121	2
High Flow Totals		**26**	**41,290**	**1,652**	**4**
Trial Flows (≤15 $m^3 \cdot s^{-1}$)	4	9	4,865 (52%)	3,080	63
	3	11	4,540 (48%)	1,123	25
	2	------ No data ------			
	1	------ No data ------			
Trial Flow Totals		**20**	**9,405**	**4,203**	**45**

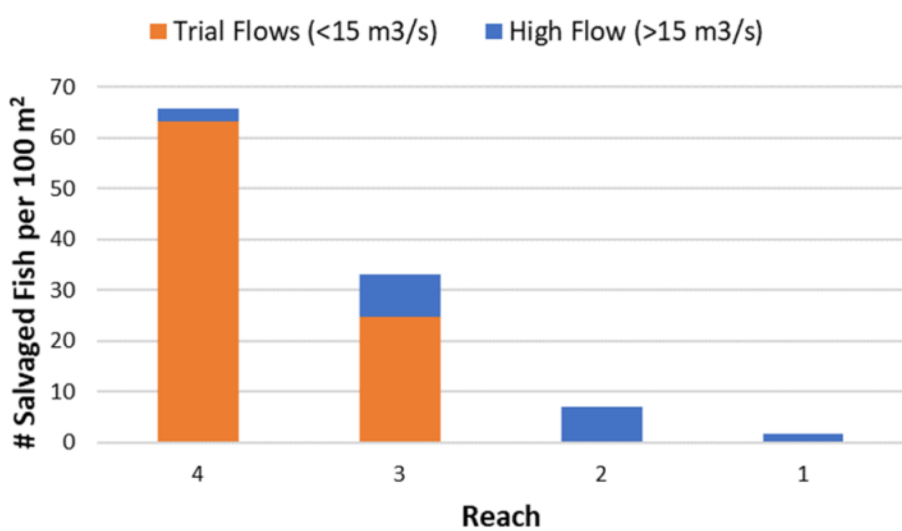


Figure 3.29 Summary of mean numbers of fish salvaged per dewatered habitat area by reach for 2016 and 2017 high flow (>15 $m^3 \cdot s^{-1}$) and trial flow (≤15 $m^3 \cdot s^{-1}$) ranges. Note: there was no data for fish stranding or salvage in reaches 1 and 2 under the trial flows.

With the benefit of fish salvage crews on the ground, some higher ramp rates (up to 4.1 cm/hr) were implemented in 2017. 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 fish salvage results from 2017, 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. 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. This could introduce some flexibility for ramping high flows down more quickly than would be possible using the WUP rate (i.e., to reach more optimal summer rearing flows, for instance). However, the sample size for stranding results at rates >2.5 cm/hr is still very small and needs to be expanded before conclusions can be more firm. Also, it is not possible to rule out the confounding effect of the high flows on these results to-date due to substantially reduced abundance of the most strand-risky fish (coho and steelhead fry) in 2016 and 2017, as mentioned above.

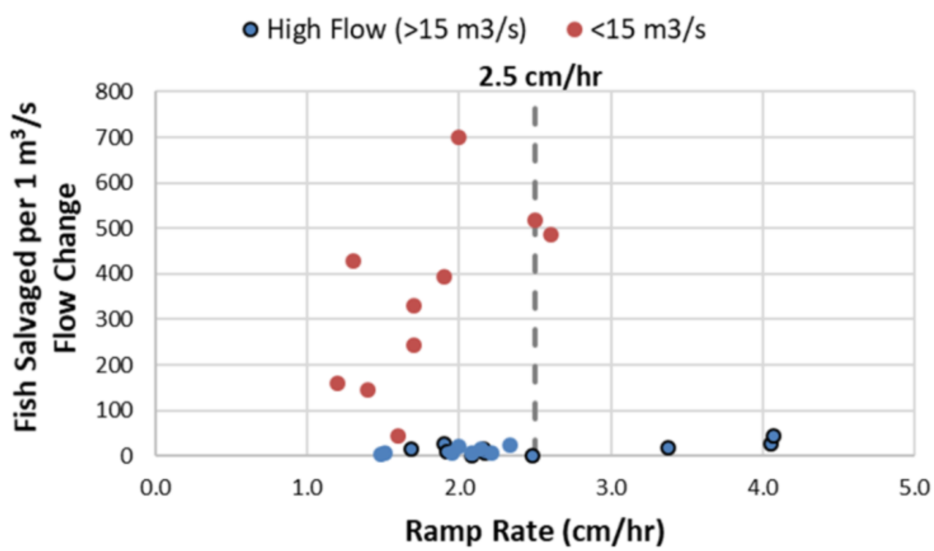


Figure 3.30 Relative incidence of fish stranding per increment of flow change according to different ramping rates under high flow (>15 $m^3{\cdot}s^{-1}$) and trial flow (≤15 $m^3{\cdot}s^{-1}$) ranges. The vertical dashed line depicts the ramp rate (2.5 cm/hr) specified in the WUP when fish salvage crews are not present. Plain blue circles represent 2016 high flow data, and blue circles with black border are 2017 high flow data.

The proportions of identified stranding sites on river left (80%) versus river right (20%) were not equal under the Trial 1 and 2 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 3.10). 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 more equal with 40% on river left and 60% on river right based on the new site reconnaissance conducted by staff from the Coldstream Ecology, Ltd-Bridge River Band partnership.

As identified for past fish salvage surveys under the trial flows (≤15 $m^3 \cdot s^{-1}$), coho and steelhead fry were the most frequently encountered species and age class under high flows in 2017 (contributing 57% and 18% to the total catch, respectively). They tended to be most prevalent at sites in Reach 3, followed by Reach 2. This despite the fact that abundance of these fish was substantially reduced in 2017 overall, likely caused by the high flows. As noted in the Fish Stranding Protocol, coho and steelhead 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 3.10 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).

Flow Range	Left Bank	Right Bank
Trial Flows (≤15 $m^3 \cdot s^{-1}$) *Reaches 3 & 4 only	80%	20%
High Flows (>15 $m^3 \cdot s^{-1}$) *New Site Reconn.	40%	60%
All	**58%**	**42%**

The next most abundant were chinook fry, which were most prevalent at sites in Reach 2. 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 are much less abundant overall than coho and steelhead fry since the flow trials began, particularly in reaches 3 and 4 (see Section 3.1.8). Steelhead parr were the least sampled species and age class, as is typically the case for ramp down survey results. They are highly mobile and able to utilize a broader range of habitats, including offshore areas that are deeper and with higher velocities.

Table 3.11 Summary of numbers of fish salvaged by species-age class and reach under high flow ramp downs (>15 $m^3 \cdot s^{-1}$).

Species	Age Class	Reach 4	Reach 3	Reach 2	Reach 1	Total
CH	Fry	2	42	193	17	254 (15%)
CO	Fry	52	419	396	81	948 (57%)
RB	Fry	24	177	82	14	297 (18%)
	Parr	43	15	83	13	154 (9%)
All		**121**	**653**	**754**	**125**	**1,653**

3.2.3. Sediment and Erosion Monitoring

This section provides the pertinent results from the KWL Lower Bridge River Sediment and Erosion Monitoring, 2017 report (Ellis et al. 2018) as directed by the BRGMON-1 Analysis and Reporting Scope of Services (BC Hydro 2017b).

Bed mobility flow thresholds based on the 2017 post-freshet surveys at 11 KWL sites (3 monitoring sites and 8 other surface sediment sampling sites) are provided in Table 3.12.

Table 3.12 2017 post-freshet mobility flow thresholds at KWL sites.

Location Name River Chainage (m, from Fraser River confluence)	2017 Post-Freshet Median Surface Grain Size (D_{50}, mm)	2017 Post-Freshet Flow Threshold for Mobility: Best Estimate [a] [*Range*] [b] (m^3/s)
Monitoring Sites		
Monitoring Site 5 (26+180 m)	60	**45** [*28 - 62*]
Monitoring Site 7 (28+510 m)	31	**23** [*11 - 34*]
Monitoring Site 6 (33+590 m)	65	**47** [*25 - 83*]
Other KWL Surface Sediment Sampling Sites		
Site 5 Alt ("Russell Springs") (30+790 m)	76	**57** [*37 - 84*]
Site 6 DS Bar (33+500 m)	34	**15** [*3 - 27*]
WP 302 (27+860 m)	74	**44** [*18 - 59*]
WP 405 (28+900 m)	133	**140** [*6 - 179*]
WP 406 (27+180 m)	59	**19** [*6 - 41*]
WP 407 (26+590 m)	18	**< 3** [*<3*]
WP 408 (25+270 m)	67	**42** [*30 - 53*]
WP 410 (31+440 m)	82	**311** [*152 - > 460*]

[a] 'Best Estimate' is for a dimensionless shear stress (τ^*_c) value of 0.045.

[b] 'Range' spans dimensionless shear stress (τ^*_c) values from 0.03 to 0.06.

Sediment transport capacity rates were calculated for the 2017 hydrograph for Monitoring Site 6 and Monitoring Site 7. The slope of the sediment transport capacity vs. flow relationship is much steeper at higher flows suggesting that the mobility rate increases at higher flows (Figure 3.31). According to Ellis et al. (2018): "It is important to note that the estimated sediment transport capacity rates are only indicative of the potential for sediment transport, and are not predictions of actual volumes of transport experienced in 2017. Actual sediment transport rates depend on the availability of sediment to be transported, which is not assessed in the underlying sediment transport rate theory."

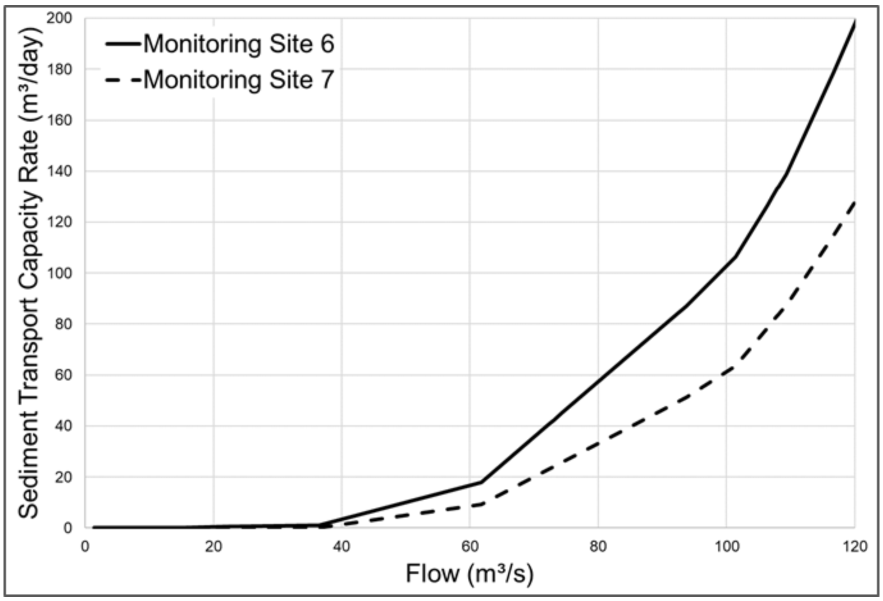


Figure 3.31 2017 Freshet sediment transport capacity rate vs. flow (from Ellis et al. 2018).

Measurements of interstitial space size and availability at 2 sites in Reach 4 before and after the 2017 high flows suggested a potential change in embeddedness measures may have occurred at those locations (Figures 3.32 and 3.33). The results showed an increase in pore depth (by ~1.8x and 1.7x for each site, respectively), but a reduction in pore density (by 0.9x and 0.6x, respectively). In other words, the spaces that were available got larger after the high flows, but there were fewer of them.

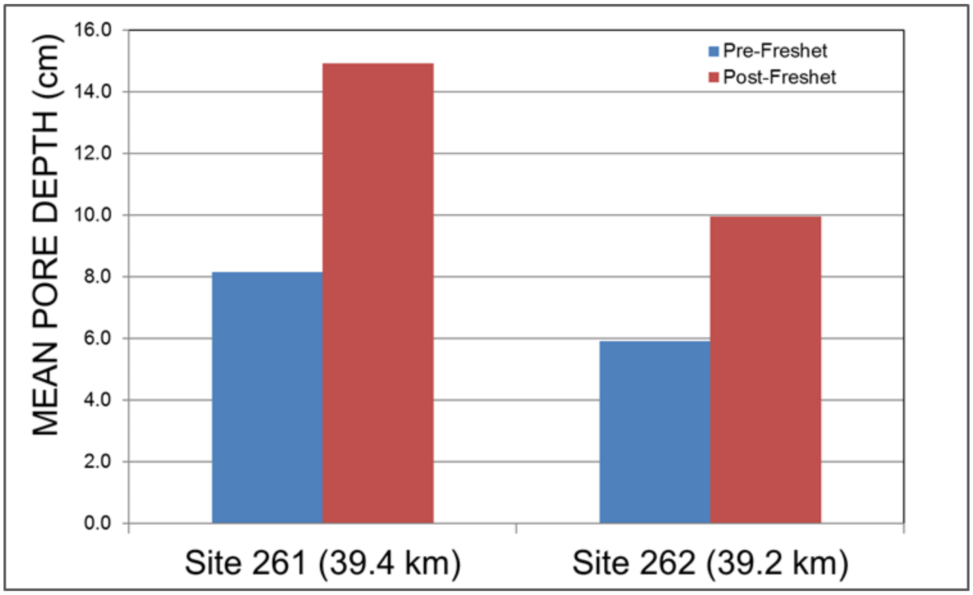


Figure 3.32 Mean pore depth before and after the 2017 freshet (from Ellis et al. 2018)

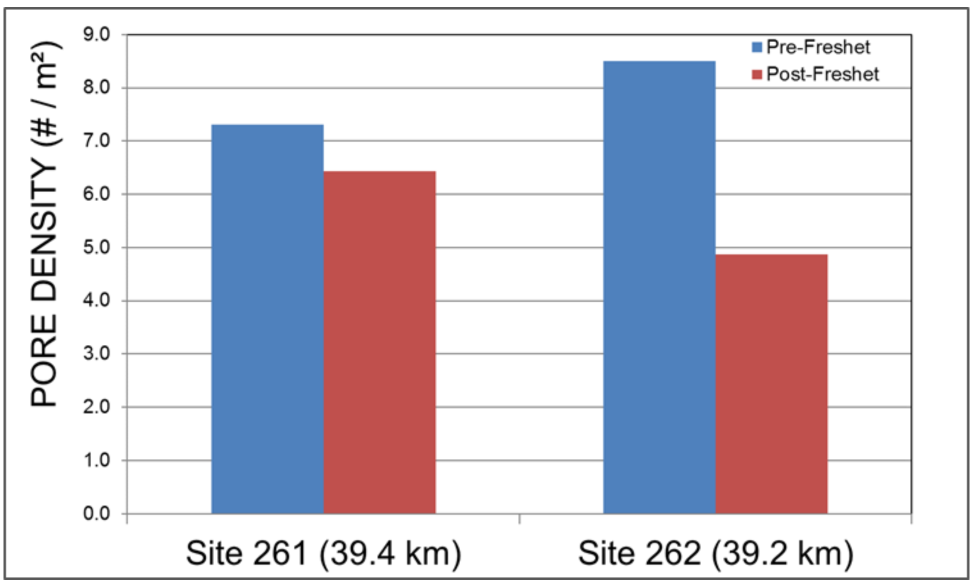


Figure 3.33 Pore density before and after the 2017 freshet (from Ellis et al. 2018).

To understand the degree of potential protection afforded to juvenile fish by the mean depth of interstitial spaces measured, KWL also estimated the average depth of erosion at 3 sites, albeit from different locations than the pore measurements were taken. Based on this analysis, the average depth of erosion at sites 5, 6, and 7 was between 10 and 24 cm depth (Table 3.13), which was greater than the pre-freshet mean pore depth at sites 261 and 262.

Table 3.13 Average depth of erosion from 2017 monitoring site topographic change assessment (from Ellis et al. 2018).

Change Metric	Site 5	Site 7	Site 6
Average Depth of Erosion (m), With Associated Uncertainty	**0.16** ± 0.07	**0.24** ± 0.02	**0.10** ± 0.04

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?

Flow releases in 2016 and 2017 were substantially higher than any other since the start of monitoring for the Lower Bridge River flow experiment. During the peak period in 2017, Terzaghi discharges completely dominated flow volumes across all of reaches 4 and 3 (8.5-fold higher than the Trial 2 peak), and were more than 5-fold greater than peak Yalakom inflows at the top of Reach 2. These high flows had impacts on physical conditions within the study area that included changes to wetted area, depths, velocities, water temperature, turbidity, channel erosion and deposition, and substrate characteristics. 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).

Prior to the onset of high flows into the Lower Bridge River channel in 2016, the most substantive effect of the continuous flow release on physical conditions in the Lower Bridge River was the continuous rewetting of Reach 4. Prior to the flow release, the total wetted area of mainstem habitat between the dam and the Yalakom confluence was approx. 17.6 hectares (ha). The inundation of Reach 4, which had been dry since the completion of Terzaghi Dam (save for periodic spill events), added 7 ha (an increase of 40% relative to pre-flow) of wetted habitat to the river at the lowest observed flow (1.5 $m^3{\cdot}s^{-1}$), and 9.7 ha (an addition of 55% relative to pre-flow) at 15 $m^3{\cdot}s^{-1}$.

Due to the constrained nature of channel, increases in wetted area were not proportional to increases in flow. The greatest gains per increment of flow in Reach 4 were between 0 and 1.5 $m^3{\cdot}s^{-1}$. Greatest gains in Reach 3 were between 1 (the pre-flow discharge) and ~7 $m^3{\cdot}s^{-1}$. The total gains in Reach 2 were less substantial because the channel was already receiving the attenuated tributary inflows, including the Yalakom River discharge prior to the flow release.

At the high flows in 2016 and 2017, the added discharge contributed additional wetted area and increased river stage by 1.43 m above the Trial 2 peak (at the top of Reach 3), but also reduced the proportional area of rearing habitat by increasing velocities beyond levels that juvenile fish can withstand throughout more of the channel. However, it was not possible to measure depths and velocities in mid-channel at the high flows. 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 when available.

During the peak period, the high flows elevated water temperatures in relative to the previous flow treatments. Particularly during June and July, water temperatures up to 3 °C higher than during trials 0, 1, or 2, and the effect was apparent in all three study reaches due to the dominance of the release flows during that period. However, despite these differences, temperatures in spring and summer were still within optimal ranges reported in the literature for steelhead spawning and incubation, and rearing for each species (Brett 1952, Bjornn and Reiser 1991, Oliver and Fidler 2001). Outside of the peak period, the thermal regime matched what has been reported previously for Trials 1 and 2: 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.

Based on spot monitoring during the high flow period in 2017, turbidities increased with the rising limb of the hydrograph, peaked at between 48 and 58 NTUs (according to location) in early June, and then steadily declined to between 14 and 16 NTUs by the end of June. Some of this effect was caused by the turbidity of the water drawn out of Carpenter Reservoir; however, much of it was also due to the effects of the high flows on channel disturbance and the flooding of edge areas that had not been wetted since the early 1990s.

The high flow magnitude in 2017 also adjusted the flow thresholds for substrate mobility in reaches 3 and 4 going forward, although the specific thresholds vary according to location due to spatial variability in both sediment grain size distributions and applied shear stress (Ellis et al. 2018). The KWL findings suggested that in order to minimize mobility (and potential loss) of spawning-size sediments, the upper flow limit would need to be between 20 and 50 $m^3 \cdot s^{-1}$ from the dam. If high flows continue to persist above these levels going forward, the river bed will likely tend towards a stable state that may not retain as much spawning-size sediments, as was likely the case under pre-dam flow conditions (~100 $m^3 \cdot s^{-1}$ MAD). See more discussion of the sediment and erosion results in Section 4.5.3.

Examination of pre- and post-freshet interstitial space depth and density (#/m^2) at a couple of sites that have supported high densities of rearing juveniles during the annual stock assessment in September suggested a potential increase in pore depth but a decrease in density at those locations. However, it is unclear how much these results reflect the broader conditions in reaches 3 and 4 due to small sample size (n=2), or to what degree these changes affect fish retention or survival.

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?

Benthic invertebrate communities in the lower Bridge River were diverse and abundant, particularly during Trials 0, 1 and 2. All of the orders commonly found in clean water mountain streams were found including caddisflies (Tricoptera), Plecoptera (stoneflies), mayflies (Ephemeroptera), the ubiquitous chironomids (midges), other true flies (Diptera), and a range of rarer taxa including naidid worms. All of the insects in these orders are highly desirable fish food organisms (Hynes 1970, Scott and Crossman 1973, Wipfli and Baxter 2010). The mean abundances of 1000 to 4000 animals per basket sample (Figure 3.) translates to areal densities of 25,000 to 100,000 animals·m^{-2}. That range is within those found in the Cheakamus River, British Columbia (13,350 – 200,325 animals·m^{-2}; Perrin 2010), the lower Capilano River, British Columbia (24,250 – 390,325 animals·m^{-2}; Perrin 2004), and the lower Coquitlam River (2,575 – 255,275 animals·m^{-2}; Perrin and Bennett 2011), but higher than in interior rivers like the Bull River in the east Kootenays of British Columbia that has average invertebrate densities of 5,148 – 14,538 animals·m^{-2} (Perrin and Bennett, 2013). Deegan et al. (1997) reported aquatic insect densities of 5,000 – 15,000 animals·m^{-2} in the Kuparuk River, Alaska. Wipfli et al. (1998) reported densities of 1,000 – 11,000 animals·m^{-2} in another Alaskan stream. These densities increased to 40,000 animals·m^{-2} in the presence of decomposing salmon carcasses. Rosario and Resh (2000) reported densities up to 18,000 animals·m^{-2} and an average of 13,761 animals·m^{-2} in perennial streams of northern California. Densities of 1,500 – 40,000 animals·m^{-2} were reported by Dewson et al. (2007a) among several undisturbed streams in New Zealand. Densities of 6,600 – 14,000 animals·m^{-2} were reported by Rader and Belish (1999) among undisturbed streams of the Rocky Mountains in Colorado. All of these comparisons show that invertebrate densities in the Lower Bridge River are within or higher than those found in other mountain rivers.

Most of the variation in invertebrate assemblages among trials was due to change in abundances of seven taxonomic groups: chironomids, simuliids (blackfly larvae), larvae of three mayfly families (Ephemerelidae, Heptageniidae, Baetidae), larvae from one caddisfly family (Hydropsychidae), and a group of “Other” taxa including Oligochaeta (mainly naidid worms), ostracods, Hemiptera and other true bugs. Given that these taxa were most sensitive to Trial effects and were numerically abundant, they can be considered Trial indicators. All of the insect groups (chironomids, mayflies, caddisflies) are fish food organisms, which means that change in abundance of these taxa due to the flow trials potentially affected availability of food for fish. This finding does not mean that fish abundance and growth was actually changed by a change in abundance of these indicator invertebrates. The decline in invertebrate density, for example, from Trial 2 to Trial 3, only means that availability of the insects that were produced in the river and are known fish food organisms declined in the change in flows from Trial 2 to Trial 3. Density dependent feeding strategies by the fish, ingestion of food organisms from riparian sources, movement of fish to other reaches may all play a role in determining actual change in condition and abundance of fish in the Lower Bridge River.

Redundancy analysis showed that several habitat attributes influenced the invertebrate assemblages. Reach 2 invertebrates were linked to the inflow of the Yalakom River. This finding shows that recruitment from the Yalakom introduced invertebrates and modified the assemblage patterns in Reach 2 of the lower Bridge River. This effect was present regardless of trial. Assemblages were modified by the presence of spawning pink salmon. This effect may be due to the release of nutrients that would stimulate periphyton growth and the release of organic matter that can be direct food for the scavenging and collector/gatherer insects.

Indeed, the invertebrate assemblages was influenced by periphyton biomass, but this effect differed by trial and reach. It was strongest in Trial 0, particularly in Reach 3 but relatively weak in Trial 3, except in Reach 4 where assemblages were associated with relatively high periphyton biomass. The assemblages laid along wide temperature ranges, indicating that the assemblage patterns shifted with temperature but were not restricted by temperature. The same was true for response to gradients of water velocity and depth.

The marked increase in densities of invertebrates with the onset of flow release to attain a mean annual flow of 3 $m^3 \cdot s^{-1}$ (Trial 1) and further increase to 6 $m^3 \cdot s^{-1}$ (Trial 2) was a potential benefit for fish in the Lower Bridge River. Taxa responding most to the effect of Trial 1 were the mayflies (in spring and fall) and caddisflies (in summer), which are important fish food organisms. Greatest response was in Reach 4 that was essentially dry during Trial 0. This finding shows that an onset of flow to produce wetted habitat can be an important area for new production. With a further increase in flow (Trial 1 to Trial 2), complexity of habitat would be expected to increase as well as added habitat area, thereby adding more suitable conditions for production of benthic invertebrates. While our data analyses dealt mainly with animal density, the coupling of an increase in density due to increased flow from Trial 0 to Trial 1 and again from Trial 1 to Trial 2 with increased wetted area would have compounded the gain in availability of food for fish. This rationale shows that the flow added to the river in Trial 1 and further in Trial 2 would have added to food potentially available for fish that use benthic invertebrates as a major food source in the Lower Bridge River.

All this changed in Trial 3. Overall invertebrate abundance dropped by 64% and the assemblages became significantly different (abundance and composition) compared to those in Trial 2. This response is common to what happens during flood events in streams mostly due to scour and physical movement of particles including invertebrates due to increased shear forces during high flow (Robinson et al. 2004). All major fish food taxa contributed to the overall decline in density. Taxa contributing most were the chironomids and mayflies followed by black flies and caddisflies. Given that the increase in spring and summer flow was the only known factor affecting the benthic community between Trials 2 and 3, we conclude that the high flows in Trial 3 caused the change in density and patterns of assemblages between those two trials. The contrast of this response to that of earlier trials showed that an optimum flow for sustaining densities of benthic invertebrates that are fish food organisms was exceeded during Trial 3.

This finding is compelling because samples of benthic invertebrates for the assessment of Trial 3 were collected at low base flow in the fall, three months after the large flow release had occurred in spring and summer. It means that the effect of the flow on invertebrates was sustained for a lengthy period without sign of recovery that is typical after stormflow events in streams and rivers. Colonization after disturbance resulting in large flows that cause bedload movement is typically rapid, usually occurring in days to a month (Mackay 1992, Figueroa et al. 2006). This ability to rapidly colonize is an adaptation to highly variable physical conditions in rivers. It is why invertebrates don't disappear from rivers that have frequent and high magnitude fluctuations in flow. The colonization process is mediated by animal movement from refuges like the hyporrheic zone that may be occupied during disturbance, recruitment via drift from upstream, adult flight and oviposition from other streams or downstream reaches of the same stream, with modifications by changes in substrate texture and particle size, food (e.g. periphyton), and competition and predation altered by change in habitat spaces between substrata particles (Mackay 1992, Gore 1982, Tronstad et al. 2007).

The lack of recovery of invertebrate densities to those found in earlier trials means that one or more of these processes limited the colonization process. The amount of refuge in the hyporheic zone of the Lower Bridge River is unknown but it is not likely to have changed among the different flows because it is not influenced by surface particle movement. If invertebrates did use it as refuge, there is no reason that hyporrheic space would not continue to be used during the Trial 3 flows. Recruitment from upstream may be a plausible factor. The Terzaghi Dam and Carpenter Reservoir are essentially a physical reset to the continuum from upper reaches of the Bridge River by preventing drift of benthos from upstream fluvial river segments to reaches downstream of the dam. It means that Reach 4, in particular, was much like a headwater stream with respect to invertebrate recruitment rather than a higher order river. This interference with the biological continuum in rivers by dams is well known in other places (Standford et al. 1996, Marchant and Hehir 2002). Following high shear forces in the river associated with high water release from the dam in Trial 3, recruitment via drift from upstream is cut off by the dam and reservoir. The absence of this source of colonizing invertebrates may limit the rate of recovery of the benthic invertebrate community in the Lower Bridge River. Adult flight and oviposition would not be expected to change between years of the Trial 3 flows and earlier trials. Particle size distribution in all reaches may have changed in association with the Trial 3 flows, given that bank erosion and bedload movement occurred during the highest flows of Trial 3. While it is unknown what is the resulting particle size distribution, it is unlikely to have resulted in substantial embeddedness (filling spaces with fines). This hypothesis means that wide ranging sizes of interstitial spaces in the river substrata remains, regardless of bed movement during the high flow release during Trial 3. These conditions would be expected to support an abundant and diverse assemblage of benthic invertebrates. If this hypothesis is correct, little change in competition and predation driven by interstitial space in the river substrata would be expected between Trial 3 and the earlier trials. The last factor is food for benthic invertebrates. It can come from two sources: allochthonous organic matter largely from the riparian vegetation and autochthonous production of algal periphyton. Allochthonous inputs to the river would not be expected to change between trials. In contrast the fall biomass of the diatom-dominated periphyton increased with rising flows through the successive trials. Periphyton are an extremely sensitive indicator of change in nutrient availability (Bothwell 1989). Even a sub-part per billion change in phosphorus concentration can be quickly sequestered by periphyton assemblages resulting in no apparent change in measured phosphorus concentration but substantial change in periphyton growth and biomass. The increase in PB between each of the successive trials shows that some increase in nutrient availability was present through the trials. That increase in biomass would contribute food for benthic invertebrates, hence ruling out food supply as a factor in the decline of invertebrates between Trials 2 and 3. Again it is important that during the actual high flows of spring and summer in Trial 3, scour of the river substratum would have occurred and potentially removed some of the attached periphyton. Sampling of periphyton at the relatively low flow in the following fall would have allowed ample time for the community to recover and produce biomass that was observed. If the benthic invertebrates were limited by food availability, the invertebrate densities would be expected to increase coincident with the observed change in periphyton biomass. This did not happen. These arguments suggest that the single factor explaining the decline in benthos abundance between Trials 2 and 3 was loss of invertebrates during the high Trial 3 flows followed by lack of recruitment from upstream to recolonize substrata.

The redundancy analysis for fall in Trial 3 shows how this process influenced links between habitat attributes and the invertebrate assemblages (**Error! Reference source not found.**). Reach 2 assemblages were strongly influenced by the Yalakom inflow and were at the low end of algal biomass. Reach 4 assemblages were constrained to relatively high algal biomass and Reach 3 assemblages were found at intermediate algal biomass as measured in the fall during Trial 3. Similarly, Reach 2 assemblages occurred at relatively low temperature compared to those in Reach 4 and Reach 3. Reach 4 assemblages were exposed to relatively high temperatures. This pattern shows that assemblages varied with cooling of water over the downstream gradient following release from Carpenter Reservoir. Assemblages in Reach 3 were most sensitive to variation in water depth and velocity because they almost directly laid along those arrows in **Error! Reference source not found.**, Plot D. All of these conditions were established after the three months following the large flow release in spring and summer during Trial 3.

In conclusion, the diverse and abundant assemblage of benthic invertebrates in the Lower Bridge River was modified by the flow trials. The release of water from Carpenter Reservoir in Trials 1 and 2 favoured invertebrate assemblages, particularly those that are fish food organisms. The compounding of a positive density response and increased wetted area that occurred with the onset of the flow release in Trial 1 and a further increase in wetted area in Trial 2 supplied abundant food for fish. Habitat conditions in Trial 3 exceeded those suitable to sustain invertebrate densities found in the earlier trials, resulting in a 64% decline in invertebrate density compared to that in Trial 2. All fish food organisms were affected. This decline in density was found from measurements in the fall of each year, which was three months after the Trial 3 flow release, which typically is ample time for a benthic community to fully recover from disturbance. Finding relatively low density in Trial 3 compared to that in the fall during earlier trials shows that colonization was restricted. Among several processes that contribute to colonization, the one that stands out is disconnection of the biological continuum from upstream caused by the Carpenter Reservoir acting as a barrier to drift from upstream. This inhibitory effect of impoundment on recruitment of fish food organisms following the flows in Trial 3 does not mean that growth of fish in the Lower Bridge River was limited by food supply. Compensatory mechanisms affecting density dependent feeding strategies by the fish, supply of food organisms from riparian sources, movement of fish to other reaches may all play a role along with change in supply of food that is produced in the river in determining actual change in condition and abundance of fish in the Lower Bridge River.

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?

Overall, juvenile salmonid abundance and biomass have been substantially reduced under the two years of high flows, compared to the two flow trials and pre-flow baseline period. 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 ~72,000 fish, respectively). Overall, the recruitment of juvenile salmonids was reduced by 70% to 80% under High flows (in 2016 and 2017) relative to trials 1 and 2, when production was greatest overall in each reach.

While all species and age classes were reduced, the degree of effect varied among some of them. Under the high flows, the average production of steelhead fry was 20% to 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 didn't further decrease under the high flows (relative to the Trial 2 mean) since their abundance was already severely depressed due to early emergence effects caused by the flow release. Coho fry abundance was 10% of the Trial 1 and 2 numbers. Differences 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 relatively minor (see more on this in the final paragraph of this section) compared to the changes in abundance.

While the duration and magnitude of the high flows were different in 2016 and 2017, the resulting abundance and biomass estimates among those two years were fairly equivalent, particularly for the fry stage of chinook, coho and steelhead. This suggests that the differences in how the high flows were delivered (i.e., magnitude and duration) between 2016 and 2017 did not result in substantial differences in the recruitment of fry between those years – both were equivalently low. This suggests the possibility that exceedance of a particular flow threshold in the channel below Terzaghi Dam could be more important than the specific magnitude or duration of flows above that threshold, though there are few high flow replicates to confirm this at this stage. Further, due to the dramatic difference between the magnitude of the high flows and the next highest monitored releases (i.e., Trial 2 flows), it's not possible to define what this specific threshold may be with the data currently available.

Adult salmon escapement estimates were provided by the BRGMON-3 program in order to evaluate stock-recruitment relationships according to flow release treatments. 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-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). Thus, more data are required to better define the initial slope of the stock-recruitment relationships to strengthen inferences about whether spawning stock size has limited chinook and coho recruitment during any of the monitoring years.

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 food availability. Mean weights of each species and age class was almost always highest (or among the highest) in each reach during the high flow period (2016-2017) compared to the other treatment periods. 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 many cases. The reason the mean sizes tended to be highest during the high flow years despite significantly reduced abundance of benthic invertebrates (see discussion of benthic invertebrate results in Section 4.2, above), is likely due to the significantly reduced density of juvenile fish in 2016 and 2017. Significantly reduced fish numbers means significantly lower competition for the food resources that are available.

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 2017 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 2016). Ramping across this range in 2017 generally conformed to the timing and shape implemented under the previous trial flows. However, the results did affirm that 13 $m^3 \cdot s^{-1}$ is the approximate flow threshold below which fish stranding risk tends to increase. As such, implementing the WUP rates (≤2.5 cm/hr) 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) to reduce flows more quickly without increasing fish stranding risk significantly (based on results for 2016 and 2017).

4.5. Additional (High Flow) Management Questions

4.5.1. High Flow Ramp Down Monitoring and Stranding Risk Assessment

Is the stranding risk during ramp downs at flows >15 $m^3 \cdot s^{-1}$ different than the stranding risk during ramp downs ≤15 $m^3 \cdot s^{-1}$?

According to the fish salvage results for ramp downs from high flows (>15 $m^3 \cdot s^{-1}$) in 2016 and 2017, the answer to this MQ is: a qualified Yes. Per 1 $m^3 \cdot s^{-1}$ increment of flow change, the fish stranding risk was consistently low (or occasionally moderate) above a threshold of ~13 $m^3 \cdot s^{-1}$ (see Figure 3.28) based on the criteria defined in the fish stranding protocol (Sneep 2016). Conversely, below the 13 $m^3 \cdot s^{-1}$ threshold, the fish stranding risk was more consistently moderate or high. This difference likely provides the opportunity to continue to implement faster ramp rates above this threshold such that flows can be reduced from peak levels to more optimal levels for summer rearing (i.e., the Trial 2 peak or lower) in less time, or over fewer days.

An important caveat that must be noted for the 2016 and 2017 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). 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 and 2017 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.

Is the stranding risk equal across the reaches of the Lower Bridge River?

According to the results of all fish salvage monitoring in the Lower Bridge River to-date, the answer to this MQ is: No. Under the previous trial flows (≤15 $m^3 \cdot s^{-1}$), only reaches 3 and 4 were surveyed, but differences in the number of fish salvaged per 100 m^2 of stranding area were significant: On average, Reach 4 densities were more than double the Reach 3 densities (see Figure 3.29) and amount of identified stranding area was nearly equivalent among them (4,865 and 4,540 m^2, respectively; Table 3.9). Differences in stranding risk among reaches were also apparent at high flows (>15 $m^3 \cdot s^{-1}$), although they were smaller (<10 fish per 100 m^2). Fish stranding densities were slightly higher in reaches 2 and 3, than in reaches 1 and 4 (Figure 3.29). At high flows, total amount of identified stranding area varied among the reaches: 5,082, 8,090, 10,700, 7,000 m^2 in reaches 4, 3, 2, and 1, respectively (Table 3.9).

Despite differences in sample size (i.e., # of years) for ramping and fish salvage data between high flow years and the previous trial 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 and 2017 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), and to characterize stranding risk in reaches 1 and 2 at flows ≤15 $m^3 \cdot s^{-1}$.

Does the stranding risk change when the maximum hourly ramping rate is greater than 2.5 cm/hr?

According to the fish salvage results for ramp downs from high flows (>15 $m^3 \cdot s^{-1}$) in 2017, the answer to this MQ is: a qualified No. Ramping rates implemented in 2017 were between 1.7 and 4.1 cm/hr (mean stage reduction per hour at the 36.8 km compliance location). This represented the first time 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 avoided proactively moving fish out of strand-risky habitats in advance of isolation or dewatering (i.e., "incidental" catches) such that catch data reflected actual numbers of stranded fish. Despite the variation in ramping rates across the high flow range, the incidence of fish stranding did not change substantively relative to the identified risk for flows ≤15 $m^3 \cdot s^{-1}$. The observed stranding risk remained low (<10 per 1 $m^3 \cdot s^{-1}$) to moderate (10 to 99 fish per 1 $m^3 \cdot s^{-1}$), as defined in the fish stranding protocol (Sneep 2016), across each of the implemented rates at high flows (Figure 3.30).

Currently the sample size for stranding monitoring at ramping rates >2.5 cm/hr is small. As was noted for the MQ above, juvenile fish abundance in 2017 was low overall, which could have confounded the incidence of stranding despite the higher rates in that year. 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.

Is the stranding risk equal on the left and right banks of the Lower Bridge River?

According to the fish salvage results available to-date, the answer to this MQ is: a qualified Yes based on site reconnaissance and salvage surveys at high flows (>15 $m^3 \cdot s^{-1}$) in 2017; and a qualified No for flows <15 $m^3 \cdot s^{-1}$. At high flows, the distribution of sites was close to equal at 40% on river left and 60% on river right; whereas, within the previous trial flow range, the distribution was 80% and 20%, respectively. Upon initial purview, differences in distribution of sites according to side of the river may seem unexpected (Table 3.10), 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), 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.

What are the potential incremental impacts of 2016-2017 flows on fish stranding in the Lower Bridge River during summer ramp downs from high flows?

As noted in the answers to several of the MQs above, the high flows in 2016 and 2017 resulted in much lower fish abundance across the study area than under any of the previous flow treatments (i.e., trials 0, 1, or 2). Compared to these earlier trials at lower flows, mean high flow abundance of juvenile salmonids was 60%, 80%, and 70% lower than Trial 0, Trial 1, or Trial 2 mean abundances, respectively. These substantially reduced numbers would most certainly have had an influence on the number of fish stranded in 2016 and 2017, particularly because coho and steelhead fry were most strongly affected by the high flows (reduced by 90% and 70% compared to Trial 2 average, respectively), and these are the species and age class that are usually at greatest risk of stranding in the Lower Bridge River (Section 3.2.2; Table 3.11; Sneep 2016).

What are the changes to the Adaptive Stranding Protocol recommended to manage fish stranding risks in the future?

The ramp down and fish salvage data from the high flows in 2016 and 2017 provide an important supplement to the data that were compiled for the Lower Bridge River Fish Stranding Protocol up to the 20 $m^3 \cdot s^{-1}$ flows that had been monitored at the time it was written (Sneep 2016). While the recommendations for ramping at flows within the Trial 1 and 2 ranges described in the protocol still stand, there has been some important learning about stranding risk according to flow rate, reach, side of the river, and ramp rates based on the work under the high flows that had not previously been assessed. The 2016 and 2017 data were also useful for bolstering support that ~13 $m^3 \cdot s^{-1}$ is an apparent threshold above which the observed stranding risk is generally low or moderate, and below which it transitions to high (according to the definitions included in the protocol).

A potentially significant uncertainty included: the degree to which the low fish abundance confounded some of the high flow ramp down results compared to the previous trial flow results (although, given the various parameters described in this report, fish abundance would likely always be lower at the flow magnitudes experienced in 2016 and 2017). Limited sample size at high flows, or for faster ramp rates, also precluded full certainty for answering some of the MQs at this stage (as described in the sub-sections above). Continued monitoring of high flow ramp events (and testing the effects of ramp rates >2.5 cm/hr) would be useful for reducing these uncertainties and further defining optimal ramping strategies for the Lower Bridge River across a broad range of operations at Terzaghi Dam.

Attempts were made in the analysis and description of the high flow ramping and fish salvage data to correspond with the existing information as presented in the protocol and facilitate potential incorporation of the high flow data in the future, if there is interest in doing so.

4.5.2. Sediment and Erosion Monitoring

What is the post-2016 flow threshold that is likely to mobilize the remaining spawning size sediment?

The following excerpts, which address this MQ were taken directly from the Lower Bridge River Sediment and Erosion Monitoring report prepared by Kerr Wood Leidal (Ellis et al. 2018).

"[This MQ] is substantially addressed by the mobility analysis … that identified mobility threshold flows at a variety of locations along the Lower Bridge River.

The results show that a single flow threshold cannot be identified for river bed mobility for the entire river because of the spatial variability of both sediment grain size distributions and applied shear stress. The 2017 post-freshet mobility analysis results … highlight the heterogeneity of river-bed sediments and hydraulics in the lower Bridge River.

This variability exists even when site selection is constrained to sediment sizes that are suitable for spawning habitat. This is because of the spatial variability of applied shear stress, which is influenced by various factors, including:

- *river channel dimensions and geometry,*
- *geomorphology of the representative reach (e.g. step-pool, bars, side channels, etc.), and*
- *the slope of the representative reach."*

"[The MQ] findings suggest that setting an upper flow limit for future hydrographs to minimize mobility of most spawning-sized sediment mixtures would result in a relatively low value (e.g. 20 m^3/s to 50 m^3/s). This would essentially require a return to pre-2016 regulated conditions, which is consistent with the maintenance of spawning habitats during pre-2016 flows. A low peak flow threshold may be impractical to implement in future years depending on runoff conditions and infrastructure constraints, and as such may not be a useful approach to reducing potential impacts from mobility.

However, the findings also indicate some spawning habitat sediment sites may be more resilient than others, in that they can withstand higher flows while maintaining suitable sediment grain sizes. For example, Monitoring Site 5, Monitoring Site 6, and WP 406 all have similar surface grain size distribution median sizes (59 mm to 65 mm). However, Site 5 and Site 6 are estimated to withstand mobility potentially up to 45 m^3/s to 47 m^3/s, whereas mobility at WP 406 is estimated to begin at a flow that is less than half those flows. Based on the above, it may be possible to locate and maintain future spawning habitat in areas that are more resilient against high flows. These more resilient areas may have also been the location of spawning habitat prior to the construction of Terzaghi Dam. Additional mobility analysis at a finer resolution could potentially identify the most resilient locations and/or define the characteristics of a more resilient site through channel geometry, morphology, and slope."

"The following can be summarized from comparison of the 2016 and 2017 hydrographs … in the context of the mobility analysis:

- *The duration of high flows (above the pre-2016 flow regime, e.g. above 25 m^3/s), was shorter in 2017, but the peak flow was higher;*
- *Pre-freshet mobility flow thresholds at the monitoring sites were exceeded for approximately 145 days in 2016, and 50 days in 2017;*
- *The freshet peak flow exceeded the pre-freshet mobility flow thresholds at the monitoring sites by approximately 900% during the 2016 freshet, and by approximately 200% during the 2017 freshet; and*

Mobility flow thresholds at the monitoring sites increased by approximately 400% during the 2016 freshet, and decreased by approximately 10% during the 2017 freshet.

The comparison of 2016 and 2017 hydrographs in the context of mobility suggests that both flow duration over the mobility threshold, and the peak flow, likely have an influence on the flow threshold following the freshet. It is assumed that less time above the mobility threshold, and a lower peak flow, would result in less change to the river bed and therefore the flow threshold would be less likely to change.

The trade-off between the peak flow magnitude and the duration over the mobility threshold remains unclear. The 2017 freshet resulted in a much smaller change to mobility threshold (reduction by approximately 10%), than the 2016 freshet (increase of approximately 400%). This may suggest that duration over the mobility threshold is more important, in determining the change in flow threshold, than the magnitude of the peak flow. Alternatively, and more simply, it may suggest that the mobility flow threshold may only be sensitive to the change in the peak flows from the previous year to the current year. The 2017 flow mobility thresholds did not change as much in 2017 as they did in

2016 because the difference in peak flows between pre-2016 and 2016 is much larger than between 2016 and 2017."

"The sediment transport capacity rate vs. flow relationships estimated for Monitoring Site 6 and Monitoring Site 7 … can be used to explore the sensitivity of mobility and sediment transport to peak flow duration, and the maximum peak flow magnitude. This was conducted by constructing a theoretical alternative 2017 hydrograph with a 'table-top' shape to be compared against the 2017 hydrograph, which generally had a high peak, and shorter-duration shape. The 2017 theoretical 'tabletop' hydrograph was shaped to convey the same total volume of water as the 2017 hydrograph. The average sediment transport capacity rate vs. flow relationship based on Monitoring Site 6 and Monitoring Site 7 was applied to both the 2017 hydrograph and the 2017 theoretical 'table-top' hydrograph, and sediment transport capacity volumes were calculated."

"This sensitivity investigation highlights the importance of the slope of the sediment transport relationship with flow: depending on the total volume of water to be conveyed, a lower peak flow sustained for a longer period could reduce the total sediment transport capacity significantly. This statement will be highly dependent on where the 'table-top' peak flow sits with respect to the associated sediment transport rate."

What are the impacts of high flows on the level of sediment embeddedness and concentration of fines in the mesohabitats used by juveniles in the Lower Bridge River?

For stream-dwelling salmonids, predation risk is often managed through the use of cover, in the form of physical refugia from predators. Juvenile salmonids that live in streams are sometimes nocturnal and may spend the day concealed in the stream substrate. Changes to substrate embeddedness following disturbance, such as high flow events, may impact the availability of interstitial cover for fry and parr.

According to the results of an earlier study on activity patterns among juvenile chinook and steelhead in the Lower Bridge River during Trial 0, fish activity was observed to follow diel concealment cycles that varied according to flow condition, habitat, season, and fish size (Bradford and Higgins 2001). In Reach 2, which had higher flows, most fish were nocturnal year-round, and they emerged from the substrate only at dusk to forage. In Reach 3, which had lower flows, some fish were active in the water column in the day in summer, but others remained concealed in the substrate until dusk. Parr and older fish were more nocturnal in summer than fry. All fish were nocturnal in winter.

As reported in the Kerr Wood Leidal (KWL) sediment and erosion monitoring report for the Lower Bridge River in 2017 (Ellis et al. 2018), the mean depth of pore spaces in the river bed increased, whereas the density (i.e., #/m2) diminished at their monitoring locations between pre- and post-high flow surveys in 2017.

Following are the pertinent portions of the discussion taken directly from the KWL report (Ellis et al. 2018):

> *"It is not known if juvenile fish are seeking shelter within the substrate during freshet in the Lower Bridge River. However, there is a limited availability of off-channel habitats for juvenile fish in Lower Bridge River (Jeff Sneep, personal communication) which may suggest that sheltering in the substrate during high flows could be more important in Lower Bridge River than in systems where off-channel habitat is abundant."*
>
> *"The availability of shelter habitat within the substrate is also dependant on the depth to which sediment becomes mobile during high flow events. It is suspected that the mortality of juvenile salmonids would be high should the sediments fish are sheltering within begin to move. Although sediment scour information was not collected at the two embeddedness sites, mean depth of erosion was collected within the topography change analysis at the three monitoring sites …*
>
> *The mean erosion depth ranged from 10 cm to 24 cm for all three monitoring sites … Erosion, or scour depth, can vary considerably from one location to the next, due to local hydraulic conditions, surface sediment size, etc. However, assuming a similar depth of erosion occurred at the embeddedness sites, where the mean pore depth ranged from 6 to 8 cm, it appears that sheltering in sediment may not provide adequate refuge for juvenile salmonids during high flows."*

There are a few limitations to interpreting the results of this assessment: 1) understanding the mechanism or causes for the changes observed (i.e., were they due to net increase in fines, or net loss of gravels, or other forms of compositional shift at these locations?); 2) understanding how the changes in pore depth and density translates to the quality of habitat that fish use across the range of flow conditions (i.e., to what degree do the observed changes affect fish retention or survival); and 3) understanding how representative the results from the two monitored sites are relative to the reach-wide or study area-wide conditions.

Sites were selected based on two locations in Reach 4 that were known to consistently support high densities of rearing fish based on standing stock assessment results from flow trials 1 and 2. While the results suggest a possible degradation of interstitial space availability at the two selected locations, it remains unknown if this potential reduction in habitat quality may have been offset by improved conditions elsewhere in the reach. As Ellis et al. (2018) phrased it: "Since embeddedness was only assessed at two locations, it is not known whether these results are indicative of embeddedness change between pre- and post 2017 freshet more generally, and whether these results are representative of shelter habitat availability on the Lower Bridge River."

5. Recommendations

Six recommendations result from the assessment of 1997 through 2017 data for lower trophic levels and fish response to Terzaghi Dam release flows:

1. Monitoring of more replicate years of Trial 3 (i.e., flow variance from Trial 2) are needed to increase sample size and establish trend and extent of response to the high flows (and potential range of high flows).

   For the fall sampling (juvenile fish stock assessment, periphyton and benthos sampling) that is most consistent across trials there were three-to-four replicate Trial 0 years (1996-1999), nine or ten Trial 1 years (2000-2010), five Trial 2 years, and to-date there are two high flow (Trial 3) years. Sample size is weak in the Trial 3 years and needs to be increased to gain confidence in the extent of the changes observed (as reported in this document).

   The periphyton and benthos data are also an important line of evidence used to interpret time course change in size of fish populations and use of the different habitats in the Lower Bridge River. To supply that evidence, the sampling of habitat attributes, periphyton accrual, and benthic invertebrate assemblages needs to continue in association with the fish sampling. In addition, benthic assemblages are universally recognized as key indicators of ecosystem health. Data describing their condition is part of the suite of information needed to assess condition of the river as evidence builds to determine optimum flows in future water use planning.

2. Maintain close consistency in fall sample-timing for juvenile fish abundance and biomass (stock assessment), periphyton, benthic invertebrates, and habitat attributes. A single, standardized time of year is needed to reliably compare effects on the lower trophic levels and fish, and time course change in those responses, among trials.

   The first three weeks of September has been the most consistent timing for juvenile stock assessment. Efforts should be made to ensure data collection is completed within this timeframe to ensure consistency in comparisons among years. Juvenile fish tend to grow rapidly at this time of the year, and differences in sample timing (even by weeks) can result in differences in fish size that affects the biomass results.

   Fall is a standard time for bioassessment in general, which makes comparisons of the benthos data from the Lower Bridge River to other rivers possible. Fall sampling also allows for community recovery (if it exists) following disturbance from high flows in spring and summer. In this way the assessment of Trial effects includes acute disturbance from high flow and recovery, which is necessary for a holistic view of flow effects on benthic assemblages. The timing of fall sampling should be early October through mid-November. Any later and benthos responses may be diminished by low temperature. Sampling at water temperatures above 7°C should be a target as has typically been done to-date.

3. Ensure that all habitat sampling is done without fail. Water depth and velocity measurements were missing for 2011 and 2012, and in other years were only collected when the periphyton and benthic samplers were installed. Since those data were lacking, those variables could not be included in analyses of habitat attributes contributing to assemblage patterns in Trial 2. Habitat variables must include all of those used for the analyses in this report (mean daily flow release, distance from origin, pink salmon on/off years, depth and velocity at sampling substrata, water temperature at sampling substrata, concentration of all forms of N and P, basic water chemistry (pH, alkalinity, conductivity, total dissolved solids), and presence/absence of influence from Yalakom River). These variables are not just used for descriptions but are used for statistical procedures that link flow and other habitat conditions to biological assemblages and ultimately fish.

4. For the juvenile stock assessment sampling, ensure that a minimum of 3 passes (max. = 4) of depletion sampling are conducted at each site every year and maintain site dimensions (m^2) from previous years to the extent possible. Maintaining consistency of effort (which we can control), even in years when fish abundances are low, is essential for reducing uncertainty in the results, particularly because there can be a degree of variability among years (even within treatments) for reasons we cannot control for.

5. Consider eliminating the monthly fish growth sampling and re-allocate that portion of the budget to other activities (e.g., further investigating potential shift in timing of emergence and effects on survival and life history of the chinook salmon population; or investigating effects of high flows on dispersion of juvenile fish within and out of the study area). Monthly growth sampling has been conducted during each of the flow trials to-date, but it has generally not been possible during periods when flows are >15 $m^3 \cdot s^{-1}$ due to crew safety concerns, poor catchability, etc. Further, sample timing has varied across years and there have been changes in emergence timing for fry due to the effects of the flow release. Each of these factors affects fish size at a given time of year and are confounding to an understanding of differences in growth. For these reasons, these data have not been included in any of the analyses to-date.

6. Develop a robust index of benthic community condition that can be calculated annually. Detailed statistical procedures that are in this report are not necessary each year but an index that summarizes community condition and food available for fish can be useful to look for trends and major change over time. Such an index may be possible with the available data. Annual calculation of an index followed by detailed analyses as in this report once every three years will lower costs but maintain insight into river condition.

Calculation of an index does not mean that measurements in the field may change. They need to be maintained as described in items 1-3 above.

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

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

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

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.

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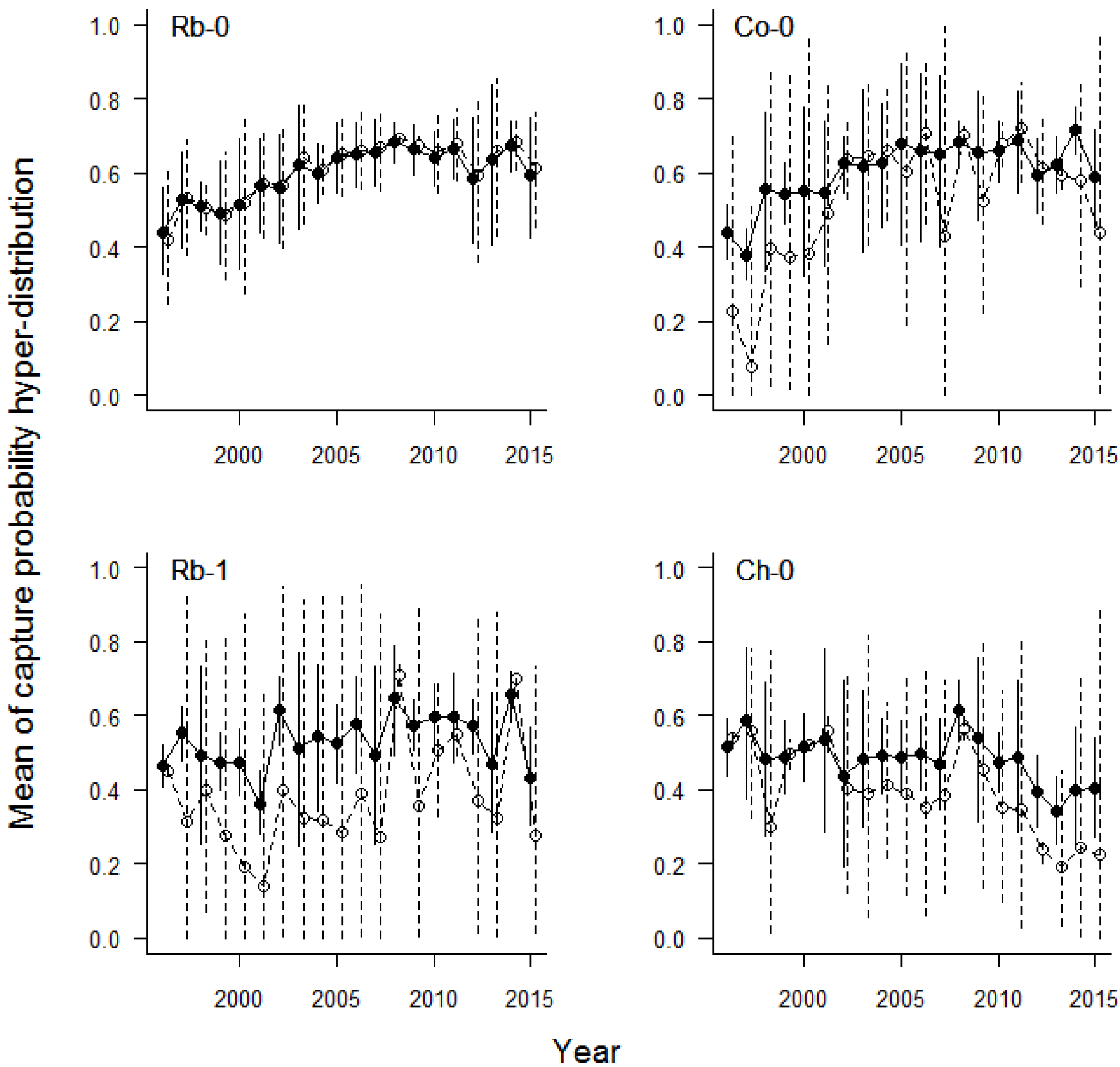


Figure A1. Annual estimates of the mean (with 90% credible interval) of the capture probability hyper-distribution (distribution of capture probability across sites) based on the HBM with more restrictive priors for the capture probability hyper-distribution (solid symbols). Also shown are estimates based on uninformative capture probability priors used in Bradford et al. (2011, open symbols).

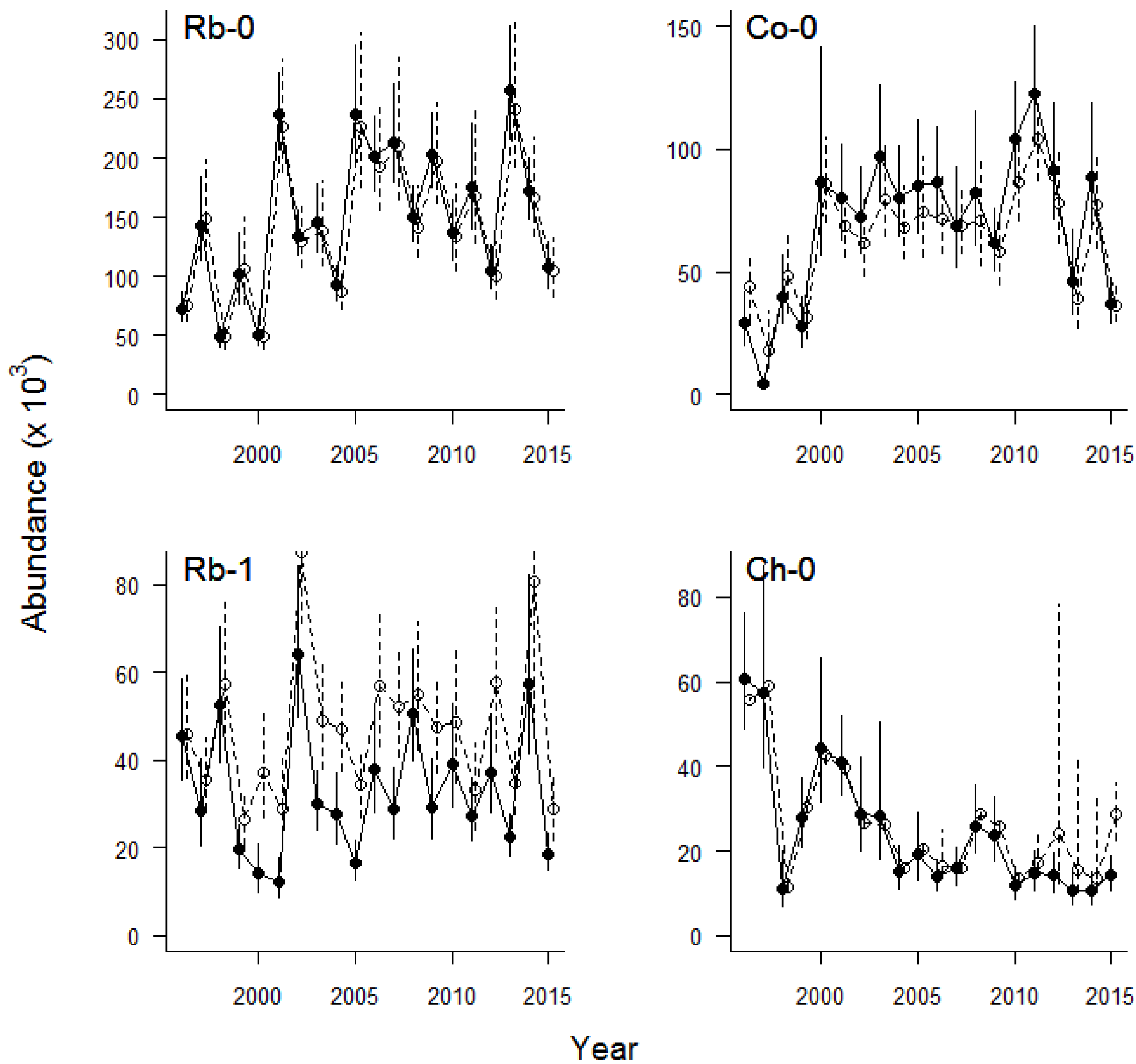
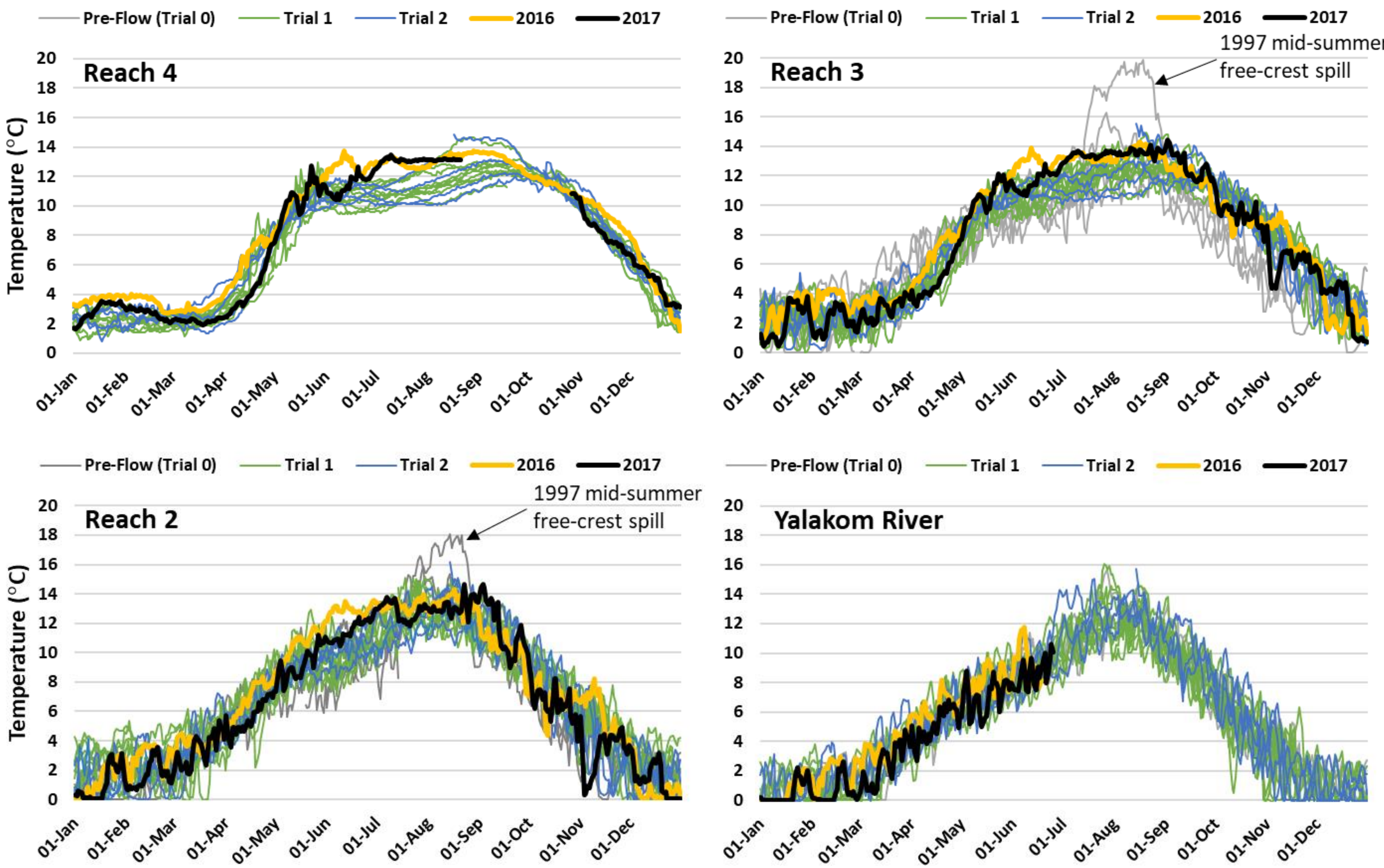


Figure A2. Annual estimates of abundance (all reaches combined) based on the HBM with a more restrictive prior (solid symbols). Also shown are estimates based on the uninformative priors used in Bradford et al. (2011, open symbols).

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



Appendix C – Summary of Potential Fish Stranding Site Reconnaissance Surveys

Name	GPS Location	Reach	Rkm	Riverside (L or R)	Habitat (pool, riffle, run, etc)	Potential Stranding Site Description	Approx: Flooded area (length x width in meters)	Stranding Risk Rating (Low, Medium, High)	Potential flow @ dewatering	Fish Observed	Initial Dewatering (m3/s)	Observations	Accessability	Picture #
Plunge Pool	50.78836, -122.22078	4	41	L	Pool	Large pothole pool, will be cut off at potential flows of 120-100.	100X30	M-H	110-100	N	110	Only a concern between rampdown flows of 110-96 cms around edges of pothole, as there is an inflow from under ground pipe maintaining water levels.	Yes, road access	IMG_3123
Plunge Pool	50.78836, -122.22078	4	41	L	Pool	Large pool below pothole pool, will be cut off below pool	50X30	M-H	45-30	N	45	Will dewater between 45-30	Yes, road access	IMG_3124
Plunge Pool	50.78764, -122.21967	4	41	R	Riffle, run pool	Large side channel, potholes, vegetation and boulders	70X30	M-H	55	N	55	Gone dewater at 55-45	No (river right) Boat?	IMG_3125
Below Plunge Pool	50.78681, -122.21604	4	40	R	Run, riffle	potholes, side channel, boulders and vegetation	150X20	M-H	55	N	55	Gone dewater at 55-46	No (river right) Boat?	IMG_3179, IMG_3180, IMG_3181
Eagle	50.78773, -122.20149	4	40	L	Run, riffle	potholes, side channel, boulders and vegetation	100X50	M-H	45-20	N	55	Potholes throughout vegetation	Road access	P7140405
Eagle	50.78773, -122.20150	4	40	R	Run, riffle	potholes, side channel, boulders and vegetation	100X20	M-H	110-45	N	110	Potholes throughout vegetation	No (river right) Boat?	IMG_3182, IMG_3183, IMG_3184
Bluenose	50.79115, -122.19592	4	39	L	Pool	large pool, eddy, large potholes, side channels, boulders and vegetation	10x10	M-H	55-45	N	45	Large pool, grassy water edge, getting shallower, losing depth, concern at next rampdown	Yes, road access	IMG_3132, IMG_3133, IMG_3134, IMG_3135
37.5	50.79261, -122.18939	4	38	R	run, riffle, pools	side channel, bar, potholes, boulders, vegetation	250x15	M-H	55	N	110	Stranded pool, potnentially increasing at next rampdown	No (river right) Boat?	IMG_3136, IMG_3137
37 Rkm	50.79291, -12218803	4	37	L	Pool, Riffle, Run	Side channel, bar, potholes, boulders, vegetation	130x20	M-H	55-20	N	45	Log jams and pool, should be observed every rampdown, some areas will isolate next rampdown	Yes, 37 walk in stock site location	IMG_3139, IMG_3140
37 Rkm	50.79291, -12218803	4	37	R	Pool, Riffle, Run	Side channel, bar, potholes, boulders, vegetation	250x10	M-H	115-80	N	110	Dewatered at 80	No (river right) Boat?	IMG_3185, IMG_3186, IMG_3189
35.5	50.79627, -122.17605	3	36	R	Run, riffle	side channel	75x15	L	55	N	55	Potential side channel stranding	No (river right) Boat?	IMG_3143
34.5	50.81094, -122.17479	3	35	R	Riffle, run	side channel	50x10	M-H	55	N	55	Next rampdown will confirm any strandning areas through observations as it is stil flowing	No (river right) Boat?	IMG_3145
34	50.81094, -122.17480	3	34	R	Riffle, run	Side channel, bar, potholes, boulders, vegetation	500X15	M-H	55	N	55	Large side channel with run and pool with large amount of debris	No (river right) Boat?	IMG_3192

Name	GPS Location	Reach	Rkm	Riverside (L or R)	Habitat (pool, riffle, run, etc)	Potential Stranding Site Description	Approx: Flooded area (length x width in meters)	Stranding Risk Rating (Low, Medium, High)	Potential flow @ dewatering	Fish Observed	Initial Dewatering (m3/s)	Observations	Accessability	Picture #
Above Fraser/sample site	50.81102 - 122.17486	3	33	L	pools	potholes	30x5	M-H	65	N	96	dying vegetation, grass, will be isolated next rampdown, conitued monitoring	Yes, road access	IMG_3147
Russel	50.82948, - 122.19781	3	30	L	Riffle, run pool	Multiple side channels, potholes, boulders and vegetation	130x24	M-H	45-15	N	45	Already shallow, will dewater quick along bank, high stranding with so many potholes, concern at throughout next rampdown events.	Road Access	P7140407, P7140408
Russel	50.82948, - 122.19781	3	30	L	Pool	pools, at old parking area	30x10	M-H	100-65	N	80	Old parking area	Yes, road access	IMG_3153
Below Russel	50.83086, - 122.19905	3	30	R	Riffle, run pool	Multiple side channels, potholes, boulders and vegetation	130x25	M-H	80-60	N	80	Dewatered 55	No (river right) Boat?	IMG_3155
29	50.832385, - 122.197980	3	29	L	Pool, runs	side channel and pools	100x10	M-H	55-45	N	45	Dewater 55	Road access	IMG_3336 - IMG_3341
Across from site 26.4	50.85855, - 122.19189	3	26	R	Riffle, run	small side channel	60x5	M-H	45-30	N	35	small channel below grizzly bar	NO (river right) Boat?	IMG_3159
Yalakom Confluence	50.86355, - 122.17330	3	25	R	Pool, Riffle, Run	multiple side channels, and large pools	50x10	M-H	55	N	55	water flowing through log jam creating large side channel on river right	No (river right) Boat?	IMG_3160
Above Camoo Bridge	50.82666, - 122.11093	2	18	L	pools, run	potholes, side channel and isolated pools	70X15	M-H	80-55	N	55	Draining well, without isolated	Yes Camoo Bridge	IMG_3163, IMG_3164
Above Camoo Bridge	50.82666, - 122.11093	2	18	R	riffle, run, pool	Potholes, debris, vegetated, side channel	50x50	M-H	80-55	N	96	Isolated pools throughout durning next rampdown	Yes Camoo Bridge	IMG_3166, IMG_3167
Below Camoo Bridge	50.82666, - 122.11093	1	18	L	pool	potholes, debris	20x10	M-H	80-55	N	55	Deep pothole along cliff edge	Yes Camoo Bridge	IMG_3165
18.5 Rkm/Camoo Creek	50.82269, - 122.09472	1	17	R	run, riffle, pools	Potholes, debris, vegetated, side channel	500X10	M-H	65-55	N	55	Large side channels, vegetation, debris, potholes from mining sites	Camoo Creek (Camoo Road)	IMG_3168
18.5 Rkm/Camoo Creek	50.82052, - 122.08580	1	16	R	run, riffle, pools	Potholes, debris, vegetated, side channel	350x20	M-H	65-55	N	55	Large side channels, vegetation, debris, potholes from mining sites	Camoo Creek (Camoo Road)	IMG_3169
Applesprings	50.81054, - 122.03753	1	11	R	run, riffle, pools	Potholes, debris, vegetated, side channel	500x10	M-H	65-55	Y (a small school of juvenile salmonids observed in small pool area on June 30, 2017)	55	Large side channels, vegetation, debris, potholes from mining sites, some isolated pools already dried up, but maintaining throughout site	Applesprings road	IMG_3173, IMG_3174, IMG_3175

Appendix D – Detailed Summary of Flow Rampdown Events

Table D1 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/s)	End Flow (m^3/s)	Flow Change (m^3/s)	Start Stage (cm)	End Stage (cm)	Stage Change (cm)	Mean 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 D2 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)	End Flow (m^3/s)	Flow Change (m^3/s)	Start Stage (cm)	End Stage (cm)	Stage Change (cm)	Mean 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**	**111.3**	**14.9**	**-96.5**	**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.)**